

**A multi-proxy approach to track ecological change in Gunbower  
Wetlands, Victoria, Australia**

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## ABSTRACT

The wetlands of the Murray-Darling Basin have come under the threat of a drying climate, the over-allocation of water for irrigation agriculture and widespread catchment disturbance. A synthesis of many paleolimnological assessments undertaken in the upper and lower sections of the Murray floodplain, and the Murrumbidgee, reveal considerable ecological change in wetlands from early in European settlement. The wetlands of the Gunbower Forest lie in the middle reaches of the Murray River. They are located on Gunbower Island that is deemed a wetland of international significance under the Ramsar Convention and an icon site under the Living Murray Initiative. Many Gunbower Island wetlands are located in protected forests, while others are within a zone developed for irrigation, mostly dairy, agriculture. This study analysed the sedimentary records of two wetlands within the forest estate and two within irrigation lands intending to compare long term change in the Gunbower wetlands to studies on floodplains both up and downstream, and to assess the relative impact of regional causes of change and that of local land use.

Sediments constitute natural archives of past environmental changes. Sediment records were recovered from four wetlands and radiometric dating and multi-proxy paleoecological techniques were applied to assess how these wetlands have responded to changes in human occupation and other factors, such as climate. Then, extracted sediment cores were taken from Black (core length: 84 cm) and Green (86 cm) Swamps located in the forest, and Taylors (94 cm) and Cockatoo (74 cm) Lagoons were situated amongst dairy farms. In order to reconstruct ecological and water quality changes from the study sites, the cores were analysed using four different analysis techniques, i.e., Itrax-XRF (X-Ray Fluorescence) scanning, Lead-210 ( $^{210}\text{Pb}$ ) dating, Stable isotope and diatom analysis. XRF scanning provided evidence of the elemental composition of the cores. Detrital enrichment in the lower parts of all cores was observed, indicating elevated erosion rates or low water levels. In addition to this, some recent metal pollution was evident with high Cu, Ni and Pb inputs. Stable isotopes provided limited information on the carbon and nitrogen sources. The  $\delta^{13}\text{C}$  enrichment in the lower sections of the cores is correlated with less phytoplankton and organic matter (less nutrients). In contrast, low values of  $\delta^{13}\text{C}$  predicted planktonic enrichment. Moreover, the C/N values of most of the core sequences showed organic matter to be of mostly aquatic or algal origin and rarely terrestrial, suggesting the dominance of internal mechanisms. Lead-210 analyses showed sedimentation rates to be approx. 1 cm/yr in Taylors and Cockatoo lagoon, whereas Green and Black swamp showed 0.6 cm/yr and 0.8 cm/yr, respectively. All four records extend only to the early 20th century, and so it appears the wetlands have become

permanent only since widespread regulation of the river system. The diatom records revealed a general shift from clear water benthic species to an assemblage dominated by river plankton. Changes shown by diatoms and sedimentation rates are relatively consistent across the four sites, and are also closely comparable to other sites analysed across the Murray-Darling Basin. However, the recent increase in the eutraphentic diatom species (e.g., *Nitzschia palea*) in the wetlands of the dairy lands suggested the influx of additional, local sources of nutrients.

This comparative study suggests that there have been changes in the condition of wetlands both in the forest and in dairy country, so the main driver of change is likely to be regional, possibly via the influence of the distributary Gunbower Creek. The intensive dairy activities appear to represent an additional stressor on nearby wetlands. As the main changes are recent it suggests that the condition of the wetlands is continuing to degrade. The records suggest that management to rehabilitate these wetlands needs to focus on regional drivers of change and to establish a more variable pattern of inundation, as well as mitigating local sources of nutrients and sediments.

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## **DECLARATION**

This thesis describes the original work of the author, and has not been submitted previously for a degree or diploma from any university. It contains no material previously published or written by another person except where due reference is made. The thesis is less than 100,000 words in length. Some material from this thesis has been published or presented (see Appendix IV).

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## GLOSSARY OF TERMS

Terms	Description
Abrupt	sudden and unexpected
Allochthonous	Material that has been imported into an ecosystem from outside the aquatic system.
Autochthonous	Originating or formed in the place where found.
GL	gigalitre
ML	megalitre
NTU	Nephelometric Turbidity Unit
Regime shifts	These are large, abrupt, persistent changes in the structure and function of ecosystems.
Raphe	The raphe is a structure that allows diatom cells to move over surfaces.
Stratification	Tendency of the water body to form separate and distinct and thermal layers during warm weather.
XRF	X-Ray Fluorescence

# CHAPTER 1- INTRODUCTION

## 1.1 Introduction and research context

About 6% of the global land area is occupied by wetlands (Junk *et al.* 2013) and they occur in a great variety of forms. Ecologists, managers and government agencies found it hard to consider a regulatory description of wetlands, so, there is no solitary, formal definition of wetlands owing to their complex nature (Mitsch and Gosselink 1993; Dennison 1997). In 1971, the acknowledgement of the ecological importance of the wetlands and the threats of human activities on them saw the launching of the International Convention on Wetlands, signed by 18 nations in the city of Ramsar, Iran (Finlayson *et al.* 2011). In this study, wetlands will therefore be considered under the broad designation provided in Article 1.1. of the Ramsar Convention on Wetlands of International Importance as “areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six metres” (Gardner and Finlayson 2018; pp 12).

Floodplain wetlands are a specific subset of wetland systems. They exhibit significant wetting and drying phases that primarily depend on the hydrological regime of the surrounding shallow lakes or rivers (Townsend 2006). Wetlands play an indispensable role in hydrological and biogeochemical cycles thus harbouring a large part of the world's biodiversity, and providing multiple services to humankind (Junk *et al.* 2013). These services include storm prevention, flood control, water supply, maintenance of the water table, groundwater recharge, nutrient and pollution control in floodplains (Barbier 1993). Costanza *et al.* (2017) identified the services that functioning ecosystems provide humanity and promoted this as a means for society to better recognise the intangible benefits that are offered by nature. However, an analysis of the extent of the world's wetlands shows that many have disappeared in response to anthropogenic pressures and climate change (Chen *et al.* 2018), particularly with the diversion of water for human consumption (Tockner *et al.* 2010; Pittock and Finlayson 2011; Finlayson 2013). Overexploitation of these water resources due to long-term human activity (such as deforestation, land use change, drainage, canalization, etc.) has altered their hydrology and limnological state. These anthropogenic impacts are further exacerbated by climate change. Cumulatively, the effects of climate and non-climatic factors, including human activities, have detrimental consequences on floodplain hydrology and hence the

ecological state of wetlands (Wolfe *et al.* 2011). On a global scale, and somewhat due to our huge dependence on them, many wetlands have disappeared (Millennium Ecosystem Assessment 2005; Davidson 2018), and many freshwater species have become endangered or extinct (Dudgeon *et al.* 2006). According to the Living Planet Index (LPI), since 1970 population sizes of freshwater species have declined by about 81% (WWF 2016), demonstrating a significant loss of biodiversity.

In 1971, the Ramsar Convention on Wetlands was initiated and signed by representatives of 18 nations in Ramsar, Iran (Matthews 1993). The Ramsar Convention is a global measure to stem the declining state of the world's wetlands and it has implications for resident and migratory wetland species (Gell *et al.* 2016). It has been suggested by the Convention that 'limits of acceptable change' would be a useful means to understand impacts on wetlands and to determine the most appropriate management regime for the wetland (Newall *et al.* 2016). In this respect, paleoecological approaches can assist in qualifying the ecological descriptions that have been developed to characterise wetlands. Reid *et al.* (2019) identified that overexploitation, water pollution, flow modification, destruction or degradation of habitat and invasion by exotic species as five leading causes of population declines and range reductions of freshwater organisms worldwide. However, due to increases in the anthropogenic activities over the last decade, it is evident that these threats have escalated and/or evolved, and now new, additional threats are apparent.

Numerous contemporary ecological studies are available on shallow lakes that include experiments; laboratory (Burks *et al.* 2001; Chang *et al.* 2019), mesocosm (Liboriussen *et al.* 2011; Trochine *et al.* 2014) and multi-lake comparisons along environmental gradients (Jeppesen *et al.* 2014) and monitoring/repeat survey studies (Hilt *et al.* 2018). Such investigations have extended our knowledge of lake ecology (Brooks and Dodson 1965; Carpenter *et al.* 1992; Vadeboncouer *et al.* 2003) and have been used to build a range of ecological theories (Moss 1990; Scheffer *et al.* 1993). However, the information is limited in the time extent covered, providing only a snapshot of impacts that seldom extend beyond ten years (Likens 2012). Thus, rarely do contemporary studies show how lake ecosystems change over the longer (decadal-centennial) timescales that are relevant to stressors such as eutrophication, acidification and climate change. Also, where there are forces of internal or external changes of long return interval or infrequent events, a long-term perspective is the principal means by which major events can be anticipated, and their impacts predicted. In such cases, paleolimnology is a different approach which helps to reconstruct change on longer timescales by using various indicators of condition preserved in sediment sequences.

## **1.2 Assessing disturbance through paleolimnology**

Initially, paleolimnology was applied to reveal lake development, based on the observations of lake change over a period of time, from oligotrophic (nutrient-poor) to eutrophic (nutrient-rich) conditions as a consequence of lake infilling with sediments (Engstrom *et al.* 2000). Later recognition of such lake changes drew attention to human induced lake eutrophication and lake water acidification (Engstrom *et al.* 2000). Lake sediments contain substantial information regarding past changes as these sediments are composed of autochthonous and/or allochthonous material. Changes in the catchment or lake itself are preserved in the lake sediments that help to reconstruct local, regional or even global changes using a variety of physical, biological and chemical proxies (Last and Smol 2006). In particular, the development of diatom-pH transfer functions (Birks *et al.* 1990; Birks and Simpson 2013) and the use of lake sediments to identify contamination of lakes by distantly transported air pollutants (Rippey 1990; Norton *et al.* 1992; Bennion *et al.* 2011) were early advancements in paleolimnology. In 1983, the effects of acid deposition on North American lakes, related to burning of fossil fuels, were investigated to examine the relationship between the location of lakes and industries (Whitehead *et al.* 1990). In another study, diatoms were used to analyse water impacts of industrial pollution on Lake Orta in Italy (Ruggiu *et al.* 1998). Paleolimnological assessments provide new insights into the nature of environmental variability in time and space at multiple scales. Climate change causes alterations in precipitation and evaporation, and so influencing salinity, lake levels and drives regime shift in lake ecosystems (Fritz 2008) that can be inferred from paleolimnological approaches.

## **1.3 Timescales of gradual and abrupt changes**

Much paleolimnological evidence from shallow lakes attests to abrupt and gradual changes. For example, Wang *et al.* (2012) revealed abrupt changes in Lake Erhai, China by analysing diatom assemblages. They found increasing variability in diatom assemblages in response to increased nutrient loading. The past diversity change was noticed in North American lakes using diatoms and cladocerans (Gregory-Eaves and Beisner 2011; Velghe *et al.* 2012) and also, diatom species turnover was predicted in Arctic lakes (Smol *et al.* 2005). Boyle *et al.* (2016) demonstrated gradual regime shifts in some constructed wetlands in Norfolk, UK impacted by the introduction of toxic tributyltin (TBT) as an antifouling paint on boats in the lakes in the 1960s. This condition has caused regime shifts, with a substantial loss of submerged vegetation, the replacement of macrophytes by phytoplankton and a change in associated diverse planktonic and animal communities. Furthermore, regime shifts are often the result of multiple stressors, and, in

many cases, the changes in forcing have exposed the ecosystems to conditions beyond the range of natural variability (Collie *et al.* 2004; Randsalu-Wendrup *et al.* 2014; Taranu *et al.* 2018). In essence, it is necessary to understand the past changes to investigate the ecological risks caused by various threats in order to protect the corresponding aquatic ecological environment and human health.

Here, paleolimnology plays a vital role in identifying reference conditions and this long-term research has dramatically improved our understanding of long-term lake ecosystem responses to anthropogenic activities (Bradshaw *et al.* 2005; Davidson *et al.* 2005; Ayres *et al.* 2007; Bennion *et al.* 2011; Bindler 2011). This information on the historical conditions of an ecosystem can help in taking management decisions. For example, The Water Framework Directive (WFD) (2000/60/EC) was established to maintain and improve the ecological health of rivers, lakes, estuaries and coastal and groundwaters by using paleoecological methods. In Europe, Battarbee *et al.* (2011) defined a paleolimnological meta-database for lakes that is derived from the recent (~100–200 years) environmental change to assess the timing and degree of pollution (acidification and eutrophication). This study was conducted to identify the period to represent the pre-impact or reference conditions recommended by the Water Framework Directive (European Directive 2003). Bindler (2011) reviewed heavy metal profile data, mainly of lead but also mercury, and cadmium, from long sediment records covering up to 2,000 years to determine the history of heavy metal pollution in European lakes. He determined the period during which natural background conditions prevailed and presented a discussion on how these may differ from the concept of reference conditions. In addition, in UK lakes, diatom records were analysed to define chemical and ecological reference conditions for better management practices (Bennion and Simpson 2011).

Further studies utilising palaeolimnological evidence have attempted to identify the degradation of many wetlands on account of human activities and that these changes were initiated hundreds, if not thousands, of years ago (Dubois *et al.* 2018). For example, in a study on a Ramsar site in the Everglades, USA, it was reported that hydrological modifications and pollutants adversely affect the ecology of wetlands (Riedinger-Whitmore 2016). Dong *et al.* (2016) have revealed a high historical baseline condition of 50 mg L<sup>-1</sup> of total phosphorus of floodplain lakes within the lower Yangtze River in China. In another study, palaeolimnology has shown the major anthropogenic influences over the last 170 years in the wetlands within the Port Philip and Bellarine Ramsar site, Australia (Reeves *et al.* 2016). Also, palaeolimnological analysis of Murray Darling Basin (MDB) wetlands revealed the historical changes in the ecological character of the wetlands which were largely unknown (Gell and Reid 2014; Gell *et al.* 2020). Hence, these examples



illustrate the benefits of using paleoecology which could improve understanding of the timing and nature of change and provide early warnings of future change. Given the importance of wetlands worldwide, it is important to understand the impact of rapidly changing climates and human activities on water quality and ecology of wetlands.

#### **1.4 Knowledge gaps and current extent of the problem in Murray Darling Basin of Australia and its wetlands**

According to Victoria's state-wide wetlands assessments of 2009-2011, the nature of the MDB wetlands has changed. These wetlands are threatened by altered hydrology, soil disturbance, degraded water quality, reduced wetland form and altered wetland form (Papas and Moloney 2012). Victoria's state-wide wetlands assessments almost covered 6% of Victoria's wetlands, determining their response to these threats. The waters of the MDB Australia have tolerated multiple stressors for more than a century (Gell and Reid 2016) from pollution, water use and climate change. For example, irrigation commenced in the 1880s and river regulation in the 1920s in the form of the construction of many weirs and locks. To fulfill the water requirements for irrigation, river regulation enabled a substantial increase in water abstraction through the 20th century. River regulation led to vegetation clearance that triggered dryland and irrigation salinisation (Macumber 1991; Herbert *et al.* 2015). The impacts of salinisation and water abstraction were so intense that, by the early 21st century, the Murray system was widely recognised as being degraded (Norris *et al.* 2001). In addition, the impacts of natural disasters such as floods and droughts cannot be ignored. Numerous studies (e.g., Reid *et al.* 2007; Davidson *et al.* 2013; Gell and Reid 2014; Gell *et al.* 2020) have shown the impacts of these stressors on the water quality and ecological character of the MDB. The threats to the MDB are of such great magnitude that the waterways of the MDB are today recognised as among 10 Australian ecosystems most at risk from tipping points (Laurance *et al.* 2011) and that many significant wetlands of the MDB have changed. The cause of the problem is contested between pollution and altered water volume.

Further, in many Australian shallow lakes, such abrupt changes were observed, attributed to changed hydrological regimes, increased abiotic turbidity, acidification and salinisation (Ogden 2000; Reid *et al.* 2007). Ogden (2000) documented significant loss of aquatic macrophytes in large, deep MDB wetlands owing to their vulnerability to light attenuation. In a study, Reid *et al.* (2007) suggested that European land disturbance and catchment erosion caused the loss of plants in Hogan's Billabong of the southern MDB. Further, Davidson *et al.* (2013) reported on 10 billabong sites in the northern and southern MDB, observing that most wetlands experienced significant declines in the diversity of benthic

diatom communities across the European settlement boundary. The evidence mounted by analyses of diatom assemblages is also reflected in fossil crustacean records. In a lower Murray River wetland, it was reported that the loss of the diverse, littoral cladoceran communities corresponded with a rise in phytoplankton (Kattel *et al.* 2015). However, the cause behind this abrupt variation remains uncertain.

Many paleolimnological analyses from the Murray River system are focussed on the upstream reaches (Reid *et al.* 2007) or those wetlands below the junction of the Murrumbidgee River (Figure 1.1). These studies have shown that a number of wetlands have experienced increased sedimentation rates and declining aquatic and benthic animal communities on account of increased turbidity (Reid *et al.* 2007; Gell and Reid 2016). Analysis of the regional distribution of responses in wetlands identifies contrasting patterns between the upper and lower systems (Gell and Reid 2014). So far however, few studies have focussed on the middle reaches of the Murray River. Also, several MDB paleoecological records lack dating models and rarely are such studies supported by the additional evidence of stable isotope analysis and XRF stratigraphy data (Gell and Reid; Gell *et al.* 2020).

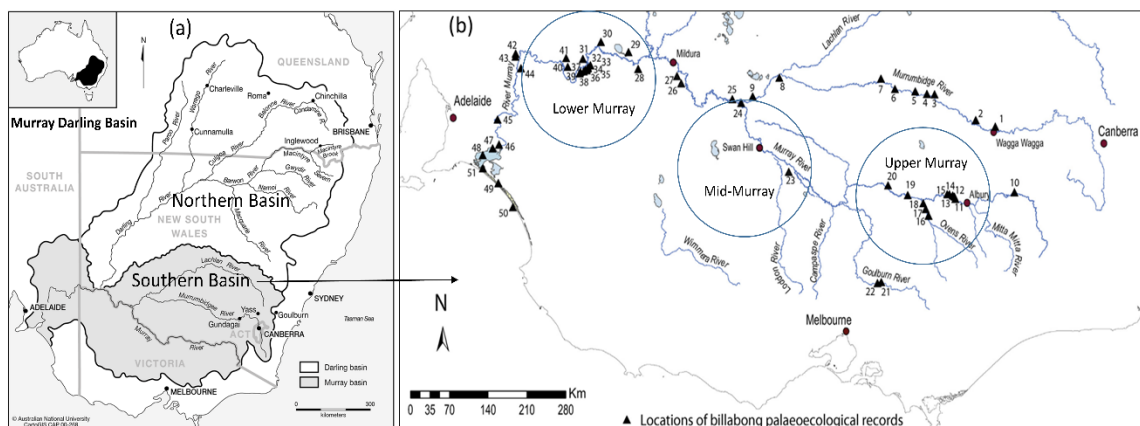


Figure 1.1 (a) Location of the Murray–Darling Basin, Australia (map source: CartoGIS Services, College of Asia and the Pacific, The Australian National University), highlighting Southern Basin (shaded grey). (b) Location of previous billabong paleoecological records of the lower, upper and mid Murray regions

## 1.5 Significance of selecting Gunbower wetlands

Paleoecological reconstructions in this study are based on a multi-proxy approach. By reconstructing the history of Gunbower wetlands in the middle reaches of the Murray River floodplain system, this thesis addresses the geographical gap in spatially distributed paleoenvironmental research in the MDB. The Gunbower Forest wetlands of Northern Victoria are situated midway between the tributary and upland sites and those in the lower reaches (Fig 1. 2). The wetlands in this study were listed as a wetlands of international importance under the Ramsar Convention in 1982 and are listed as iconic sites under the

Living Murray (TLM) Program (Hale & Butcher, 2011). Furthermore, these wetlands are targets for restoration and the state and federal governments have invested in infrastructure to provide environmental flows to the wetlands in an attempt to ensure that their ecological character and biological diversity are maintained and enhanced. However, the site has been influenced by natural and anthropogenic activities, such as climate change, river regulation, nutrient enrichment, and sedimentation for over a century.

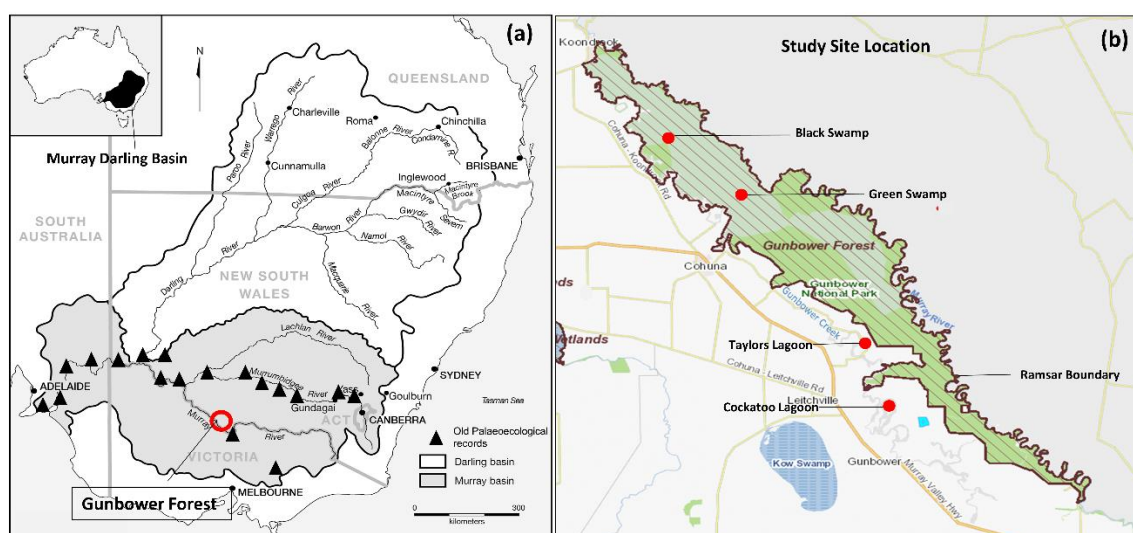


Figure 1.2 Location of the Murray–Darling Basin, Australia (map source: CartoGIS Services, College of Asia and the Pacific, The Australian National University). Black triangle indicates old paleoecological sites and the red circle indicates the Gunbower Forest. (b) Location of the study sites (Black Swamp, Green Swamp, Taylors and Cockatoo Lagoon) in the Gunbower Forest (shaded green), Murray–Darling Basin northern Victoria (near Cohuna). The Ramsar boundary of the study site is shown by the dark brown line around the Gunbower region.

Black Swamp and Green Swamp are located in a protected forest (Ramsar site) while Cockatoo and Taylors Lagoon are within an extensive dairy farming area. These sites are part of a local network of wetlands connected, via Gunbower Creek, to the hydrological system of the Murray River. In contrast to the sites within the Ramsar listing area, the dairy farming sites are situated in an agricultural landscape and so are near to local pollution sources. It is hypothesized that these study sites are under the influence of regional drivers of wetland changes but with more proximal sources. Hence, this study will explore the degree to which the ecological condition of the wetlands at the time of its Ramsar listing was representative of its long-term state, and to assess the causes and any deviation that may have occurred in the wetlands of the Gunbower Forest. Paleoecological evidence in several Ramsar sites elsewhere has shown alteration of ecological character even before they were listed as International Ramsar Sites (Gell *et al.* 2016) suggesting the significance of the study of this kind.

Based on the knowledge gaps, this is the first study in south-east Australia to be conducted on Ramsar wetlands to understand the water quality and ecology using a multi-proxy approach. It is believed that this project will contribute to extend our understanding of the scale of change driven by climates, anthropogenic pressures such as dairy farming, and geochemical processes. Other innovations of this study will be the contrast between land use types which addresses the question of regional v local drivers of change for the first time.

## **1.6 Research questions and objectives**

This study is based on the two hypotheses: 1) water quality of Black and Green Swamp has changed since its listing; and 2) the Gunbower system is under the influence of local drivers of change. These hypotheses will be addressed by key questions that will focus on the changing sources of nutrients to wetlands and, anthropogenic and climate change impacts on geochemical fluxes. The following specific research questions will be addressed by this project:

Question 1. How has the water quality, ecology and condition of wetlands varied and changed in the past?

Question 2. What have been the impacts of dairy pollutants and river regulation on wetlands?

Question 3. What has been the nature of wetland change and how can this knowledge lead to a better understanding of its modern ecological character and function?

To answer these questions, data from multi-proxy approaches (Figure 1.3) will be used to form a localised/regional representation of environmental changes allowing comparisons to be made with other secondary paleoecological records produced within the MDB. This will be achieved by analysing the fossil diatom assemblages and sediment geochemistry in combination with chronological techniques. In brief, diatom species assemblages were enumerated to indicate environmental changes, particularly of water quality and habitat, in the respective water bodies. Sediment cores were scanned through the Itrax core scanner in the Environmental Research facility at Australian Nuclear Science and Technology Organisation (ANSTO). Itrax scanning is a non-destructive technique that provides high resolution analysis of the geochemical nature of the sediments of the cores being analysed (Croudace *et al.* 2006). The Itrax scanner measures geochemical elements within the sediments. The continuous data derived from the scans of the cores reflect changes through time. The X-radiographs help direct further subsampling for other techniques, such as  $^{210}\text{Pb}$  dating, and the magnetic susceptibility measurements aid the

identification of phases of sediment input from the catchment. Furthermore, stable isotope analyses of organic matter preserved in the cores were applied to indicate past trophic conditions.

This multi-proxy study will identify the baseline and derived ecological character of the wetlands and the factors impacting the water quality and ecology of the wetlands and their catchments by combining Itrax scanning,  $^{210}\text{Pb}$  dating, stable isotope and diatom analysis. This research seeks to examine a century of change and evaluate their causes; specifically, it seeks to distinguish between local/direct or regional/systematic drivers of change.

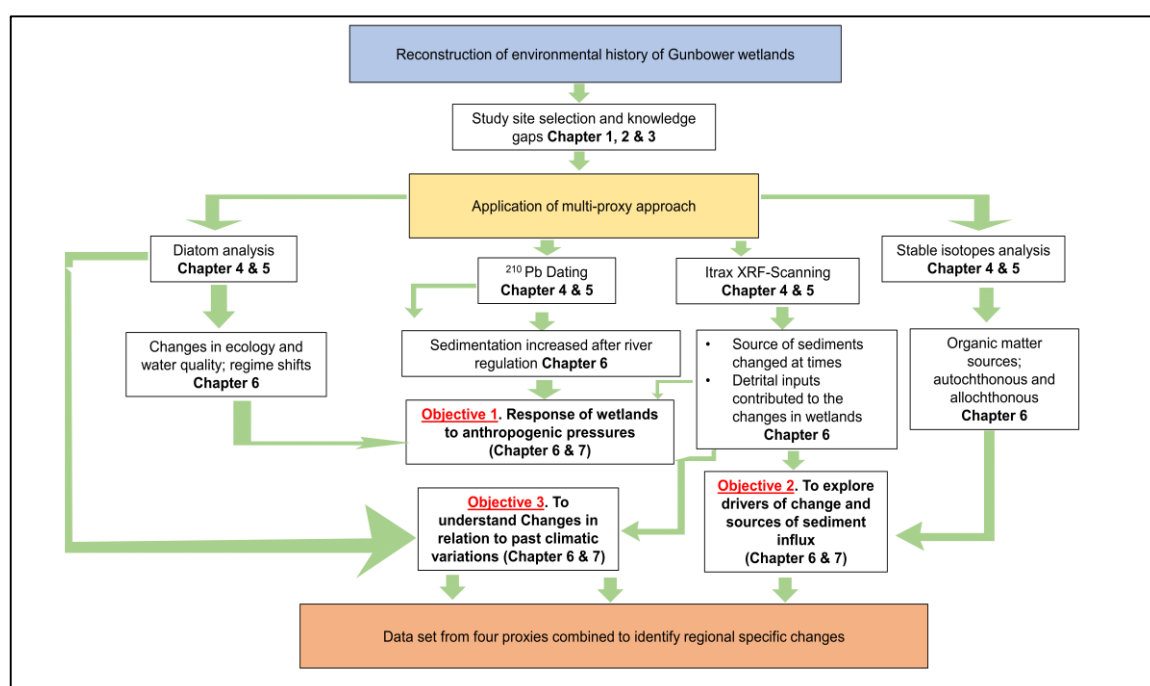


Figure 1.3 Research framework to reconstruct environmental history of Gunbower wetlands. Knowledge gaps and main objectives are highlighted with corresponding chapter numbers.

This study will help in determining the ecological response of seasonal or intermittent wetland systems to climate variability and anthropogenic impacts in south-eastern Australia. It will also contribute towards a better understanding of the relationship between wetland variability and catchment scale mechanisms and how the biological functioning of aquatic ecosystems has evolved over the past years in response to these changes. This research aims to fill a significant gap in our understanding of the impacts of global and regional forces on the water quality and ecology of Ramsar wetlands by providing a longer term view from a paleoecological perspective. To achieve this aim, the following objectives are set:

1. To use multi-proxy data from sediment sequences to establish the response of wetlands to anthropogenic pressures at each wetland over time;
2. To understand changes in wetland conditions in response to past climatic variations;
3. To compare and contrast how the ecology and water quality of four wetlands in different settings have been affected in the last 100 years;
4. To explore drivers of the observed changes in water quality of four wetlands;
5. To combine data sets to identify regionally specific environmental changes;
6. To understand the path that wetlands have taken to their present state to assess whether and how authorities might intervene to restore their ecological character and biological diversity.

## 1.7 Thesis structure

This introductory **Chapter 1** provides the background to the importance of wetlands and their degradation, impacts of natural and anthropogenic changes on wetlands, with emphasis on the situation in MDB wetlands. This chapter briefly states the knowledge gaps, site selection, research questions and aim, and concisely outlines the research methods employed.

The literature review **Chapter 2**, begins by reviewing the current paleoenvironmental literature, focusing primarily on the factors that impact the ecology and water quality of wetlands. The study site **Chapter 3**, aimed at acquainting a reader unfamiliar with Gunbower Forest with the regional setting for this research. In this chapter, the physical setting of the Gunbower Forest and the study sites, including the region's geomorphic evolution, hydrological issues, status under the Ramsar Convention, vegetation and climate have been discussed.

**Chapter 4** begins with a brief introduction the chapter outlining field and laboratory techniques. This is followed by a justification of the selection of study sites used in this study followed by detailed account of methods used in the field to extract sediment cores to isolate the required proxies. The laboratory procedures are outlined, including methods used in the diatom preparation, Itrax, stable isotope and  $^{210}\text{Pb}$  analysis, used in reconstructing history of Gunbower sites. Thereafter, statistical procedures such as indirect ordination, CONSENS (Constrained Incremental Sum of Squares) and analysis by statistical softwares (e.g., Tilia, C2) have been discussed in detail.

In **Chapter 5**, results for each of the four sites are presented. This chapter presents the stratigraphic changes in diatom composition and elements for the sediment profiles,

plotted against the depth in the profile, from each of the four study sites. Statistical analyses of these changes are then presented with zones defined by CONISS. The sediment accumulation rates and age of the cores are determined by  $^{210}\text{Pb}$  analysis while stable isotopes of carbon and nitrogen provide information about nutrient inputs.

**Chapter 6** synthesizes the information collected from the proxies for each of the study sites providing an interpretation of changes in the water quality and ecological character of the sites. In this chapter, comparison is made across the four study sites to infer significant changes in the Gunbower system. Data from all proxies are discussed in a global context and implications for regional and local contamination sources are discussed in detail.

**Chapter 7** represents the key findings of the study and the extent to which the study aims have been achieved. As this study is the first high resolution study conducted in Gunbower Forest, provides some insight for future research.

# **CHAPTER 2- UNDERSTANDING WETLANDS, RIVER-FLOODPLAIN DYNAMICS, FACTORS IMPACTING THEIR ECOLOGY AND RELATED PALEOLIMNOLOGICAL STUDIES - A REVIEW**

## **2.1 Introduction**

The twenty-first century has been characterised by unprecedented environmental changes owing to the increasing rates of global warming and anthropogenic activities (IPCC 2014). These environmental changes include ongoing land clearance and deforestation, declines in natural vegetation, pollution, infrastructure development, degradation, and loss of natural ecosystems. The functioning and ecological conditions of natural ecosystems have been impacted by environmental changes that pose formidable challenges for global water resource management practices (Bhaduri *et al.* 2016; Large *et al.* 2017). In particular, inland water systems have been identified among the most threatened ecosystems (Finlayson *et al.* 2005; Dudgeon *et al.* 2006; Huziy and Sushama 2017). Inland water systems cover approximately 0.8% of the earth's surface and hold just about 0.26% of the total water resources on the planet (Dudgeon *et al.* 2006; Carpenter *et al.* 2011). Earth's total land area is occupied by ten largest countries such as Russia, Canada, United States, China, Brazil, Australia, India, Argentina, Kazakhstan and Algeria (Figure 2.1). Of this land area, 1.1% of Australia's land is covered by inland water systems. The existence of these inland water systems is affected by the surrounding natural and anthropogenic conditions posing further ecological changes.

In this chapter, the literature on the relationship between wetlands, anthropogenic pressures (e.g., river regulation, pollution, eutrophication, anoxia, introduction of exotic species, etc.), and climate change is reviewed to provide a conceptual basis for this study. Firstly, this chapter explores the existing literature on the world's wetlands, their distribution, diversity, and ecosystem services. These discussions led to a review of river-floodplain dynamics, wetland change and, related global paleolimnological assessments are outlined to understand established evidence for the ecological response of wetlands to human activities and climate change. Thereupon, the development of paleolimnological work in the MDB, Australia is outlined to familiarise the reader with key sites and proxies used. The use of diatoms, pollens and geochemical properties as paleolimnological proxies in the MDB is then critically assessed, describing their value as ecological and climate proxies. Finally, the current understanding of the state of MDB wetlands is presented with gaps in the existing literature.



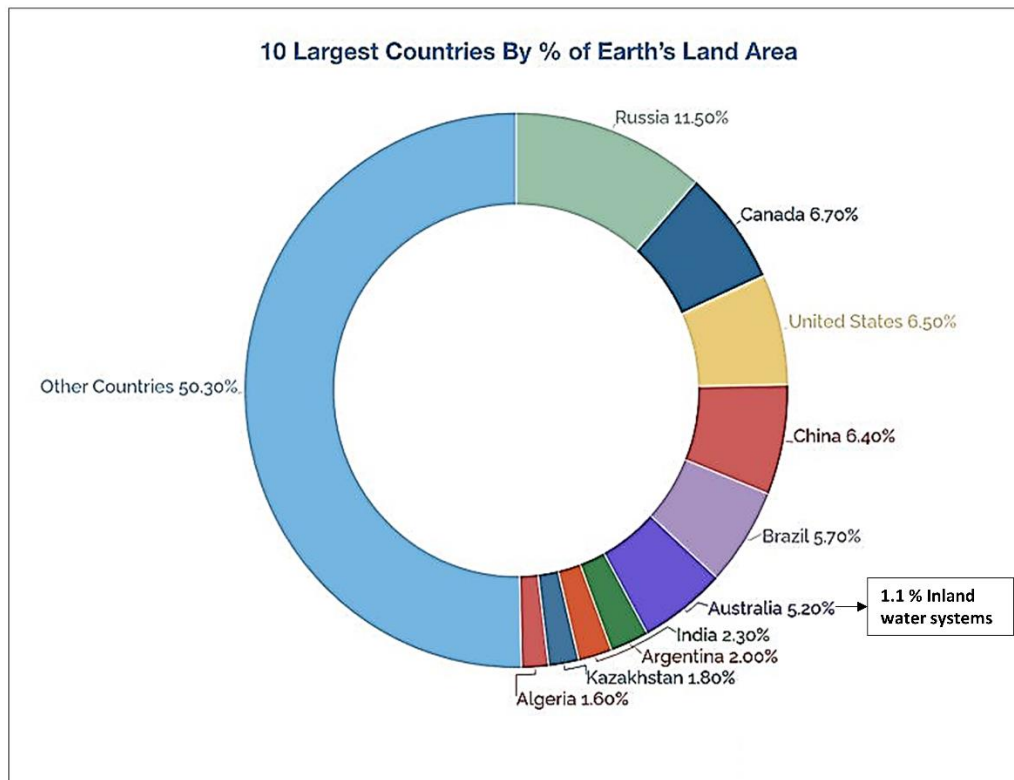


Figure 2.1 Comparative land mass figure of ten largest countries in the world and 1.1% of Australian land mass is occupied by inland water systems (Source: <https://www.worldatlas.com/features/countries-by-area.html>)

## 2.2 Wetlands – distribution and diversity

Based on a recent assessment by Davidson (2018), global inland and coastal wetland area is more than approx. 12.1 million km<sup>2</sup>. Of this area, 54% is permanently inundated while 46% is seasonally inundated. Around 93% of wetlands are inland systems, with only 7% being coastal. Globally, human-made wetlands are small in comparison; reservoirs cover approx. 0.3 million km<sup>2</sup> and rice production 1.6 million km<sup>2</sup>. Since the 1970s there has been a remarkable increase in the estimated global wetland area over time, due to recent advancements in remote sensing and mapping technologies. The largest areas of wetlands are in Asia (32%), North America (27%) and Latin America & the Caribbean (16%) while wetland areas in Europe (13%), Africa (10%), and Oceania including Australia (3%) are smaller (Davidson 2018) (Figure 2.2). Although wetlands are widely distributed across the earth's surface, the condition of many natural wetlands have been progressively declining for centuries in most parts of the world mainly due to anthropogenic activities.

The Wetland Extent Trends (WET) Index provides the most detailed analysis available to date about global wetland status and trends. This index shows a continuing progressive decline of wetland areas (Dixon *et al.* 2016). Globally, there has been a decline of about

35% in both coastal and inland wetlands areas. It is documented that regional declines have been significant in Latin America and the Caribbean (59%), and Africa (42%) but least in Oceania including Australia and North America (20%). Davidson (2014) synthesised several published reports and documents to report on the loss of wetlands. According to the report, in regions for which data exists, 87% of wetlands around the world have been lost since 1700 CE, and this loss mainly occurred in the 20th and early 21st centuries. Since the 21<sup>st</sup> century, and despite our strong dependence on their environmental functions and services, more than 60% of the world's wetlands have vanished due to anthropogenic influences (Xu *et al.* 2019). In Oceania, including Australia, half of wetlands have been lost with continuing high rates (Kingsford *et al.* 2016) owing to anthropogenic activities, leading to the extinction of freshwater wetland-dependent species.

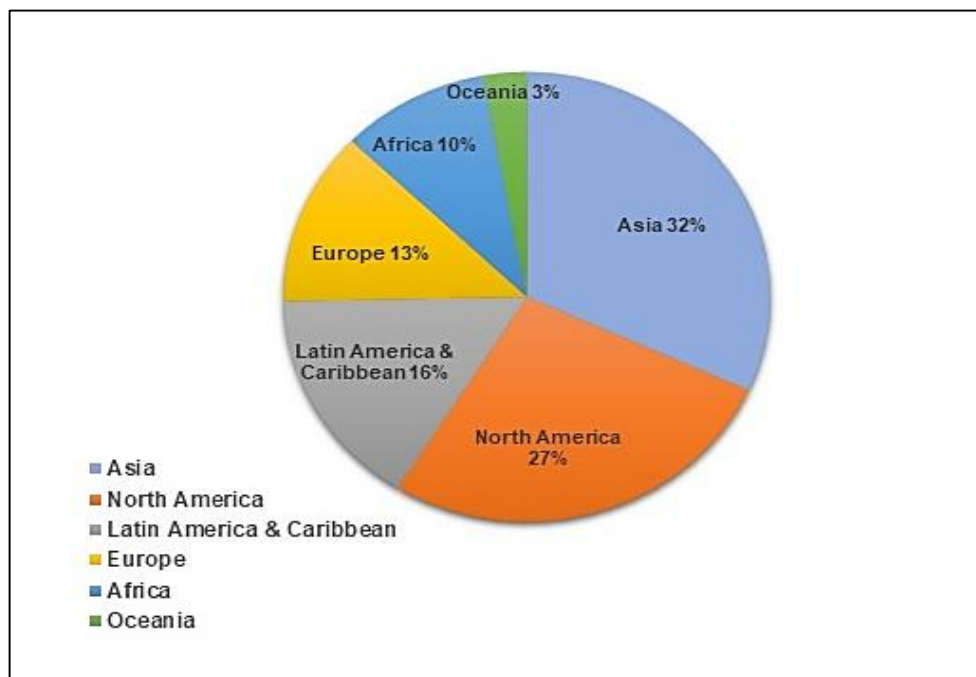


Figure 2.2 Percentage of wetland area based on regional distribution (Source: Davidson 2018)

Recent assessments of wetland-dependent and freshwater species have been undertaken by the International Union for Conservation of Nature Red (IUCN) and The Living Planet Index (LPI) (WWF 2016). In the IUCN assessments of over 19,500 wetland-dependent species were assessed globally out of which one-quarter (25.1%) are considered threatened with extinction. According to the LPI assessment, since the 1970s, approximately 81% of the population sizes of freshwater species have declined globally.

It is considered to be much greater decline than those of species depending on any other ecosystem.

### **2.3 Importance of wetlands and their ecosystem services**

Wetlands are crucial for human livelihoods such that they satisfy the increasing water demand, nutrient cycling, maintenance of biodiversity, and recreational opportunities for people. Wetlands play a significant role by providing hydrological, social, and environmental services which are very beneficial to both natural systems and the human environment (Finlayson *et al.* 2005; Hu *et al.* 2017). An important hydrological service of wetlands is mitigating storm damage and flood disasters, especially in urban areas which are dominated by impervious surfaces with an altered hydrological cycle and large runoff (Carpenter *et al.* 2011). Further, wetlands act as major sources of water for domestic, agricultural, and industrial purposes. Also, wetland waters are utilised for fishing and aquaculture, for extracting and growing agricultural and horticultural products, for transport, and as sources of energy resources (Millennium Ecosystem Assessment 2005; Fritz *et al.* 2018). Environmental and physical functions of wetlands include micro-climate regulation and carbon dioxide sequestration and storage (Finlayson *et al.* 2005; Mitsch *et al.* 2013; Pachauri *et al.* 2014). Wetlands provide natural ecological conditions such as habitats for a diversity of species, especially migratory birds and other animals of high conservation value (Barbier *et al.* 1997; Nyman 2011; Gardner and Finlayson 2018). In addition, wetlands play a significant role in nutrient cycling owing to significant river-floodplain interactions.

### **2.4 River-floodplain dynamics**

Rivers constitute important longitudinal and lateral dynamics of flowing water (Wetzel 2001; Fritz *et al.* 2018) carrying sediments, nutrients, and other materials through the landscape and into other aquatic systems. This interaction creates a linkage between the land, lakes, wetlands, and rivers. Thus, rivers and streams are known as the 'circulatory systems' of the environment (Wetzel 2001). The floodplain is described as the zone where a lateral exchange of water, nutrients, and organisms occur between the river and its floodplain (Thoms 2003; Fritz *et al.* 2018). The interactions between rivers and floodplains are of utmost importance. The flood pulse concept is among the most comprehensive approaches to understanding river-floodplain interactions (Junk *et al.* 1989). According to this concept, rivers and their floodplains are considered an integral part of a single complex system and are strongly associated with each other through hydrological and ecological processes (Junk *et al.* 1989; Mitsch *et al.* 2008; Higgs *et al.* 2020). This

relationship allows floodplains to exchange matter with the river that causes increased concentrations of macro elements influencing biochemical processes. In these ecosystems, various abiotic (soil) and biotic (vegetation) elements of an ecosystem are connected through surface water and groundwater (Figure 2.3) (Juliano *et al.* 1999). Simultaneously, a spatial connection exists between the valley and riverbed along the river continuum (Kiedrzyńska *et al.* 2008). These connections are crucial for maintaining the function and integrity of floodplain-river ecosystems (Zalewski and Kiedrzyńska 2010).

Floodplains are complex habitats; ecosystems that differ in the hydrological connectivity which affects the ecosystem dynamics. Floodplains respond to the spatial and temporal variability of river discharge and flooding which influences ecological stability and rates of biogeochemical cycling (Kattel and Gell 2012a; Reid *et al.* 2016). This variation in lateral connectivity across the floodplain creates a distinct zonation of habitats. These differences in the connection of floodplain lakes with the river channel determine the availability of nutrients and the extent of the river to control processes such as primary productivity and decomposition (Tockner *et al.* 2000; Kiedrzyńska *et al.* 2008; Higgsion *et al.* 2020). At the same time, ecosystems isolated from the nearby river for most of the year develop a lentic character that promotes autochthonous organic matter cycling. This means that river (lotic) systems have floodplains that sometimes act as lentic systems. An increased flow rate throughout these water bodies is beneficial to organisms in transporting food, oxygen, nutrients, and particulate and dissolved organic matter (Dong *et al.* 2012a). However, these interactions are becoming largely depleted by many factors such as climate change and anthropogenic pressures.

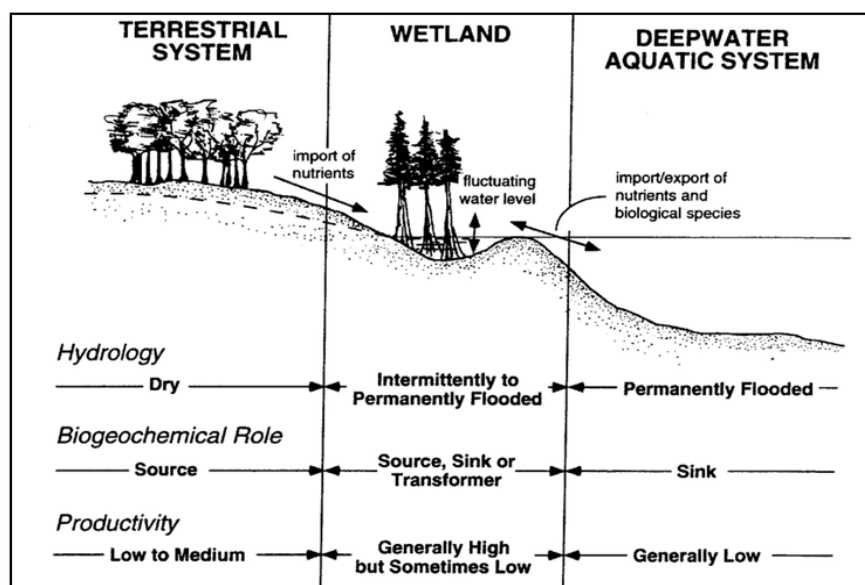


Figure 2.3 Interactions between terrestrial system, wetland and deep aquatic system (Reprinted from: Juliano *et al.* 1999)

## 2.5 The impacts of climatic factors and non-climatic factors on wetland ecosystems

It is widely accepted that inland water systems such as rivers and lakes are important for human livelihoods and they provide a variety of ecosystem services. This has led to many human activities (construction of dams and levees) around rivers and lakes, impacting their surrounding landforms such as wetlands (Poff *et al.* 1997; Thoms 2003; Tockner *et al.* 2008; Junk *et al.* 2013; Huziy and Sushama 2017). It is highly recognised that wetlands are vulnerable to human activities and are one of the most threatened environments mainly from human and climate-induced changes (Junk *et al.* 2013).

In this section, the major factors that impact the water quality and ecosystem dynamics of wetlands have been discussed. These impacts are categorised into two types: natural/climatic and human-induced/non-climatic impacts (Figure 2.4). Human impacts include river regulation, pollution (e.g., agricultural, dairy farming, chemical, and metal pollution), eutrophication, and the introduction of exotic species while climatic drivers encompass changes in e.g., temperature and the hydrological cycle etc. (Table 2.1). An ecosystem is a dynamic complex of biotic (plants, animals, and microorganisms) and abiotic (i.e., non-living) factors. Knowledge of these factors enables us to explain patterns of evolution and ecological drivers of wetland changes.

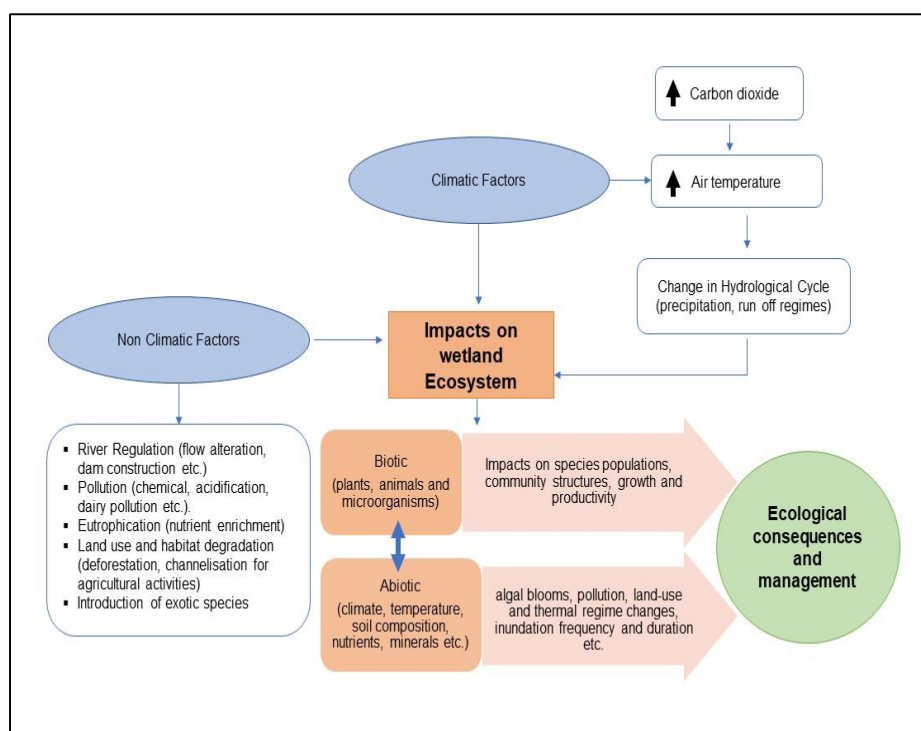


Figure 2.4 Conceptual approach to understand the impacts of climate factors and non-climatic factors on wetland ecosystems (Adapted from: Lloréns 2008)

**Table 2.1 Impacts of climate change and human induced changes on wetland and their ecological consequences**

Factors	Impacts	Ecological consequences	References
<b>Climate-induced changes</b>	Change in hydrological cycle, declining water levels	Declining population of species, community dynamics, plant growth, habitat degradation, loss of fishery potential and endemic species	Coops 2003; Kingsford <i>et al.</i> 2011; Zhang <i>et al.</i> 2012; Wang <i>et al.</i> 2013; Yang <i>et al.</i> 2016
	Extreme events of flood and droughts, changing water levels	Impacting the extent of suitable habitat for species and water quality	
	Alteration of river channels causing erosion	Changes in community structures, productivity, and algal blooms	Dudgeon 2002; Palmer <i>et al.</i> 2008; Pachauri <i>et al.</i> 2014; Atkinson <i>et al.</i> 2019
	Unpredictable flood events	Transport of excessive nutrients causing eutrophication; increased phytoplankton biomass	Paerl <i>et al.</i> 2016; McCullough <i>et al.</i> 2012
	Elevated temperature and rising carbon dioxide concentrations	Production of harmful algal blooms	Visser <i>et al.</i> 2016
<b>Human induced change</b>	River regulation, dam and weir constructions	Habitat destruction as the greatest global threat to biodiversity	Leadley <i>et al.</i> 2010; Atkinson <i>et al.</i> 2019
		More frequent flooding of low-level wetlands; reduced flooding of high-level wetlands	Maheshwari <i>et al.</i> 1995; Walker and Thoms 1993; Kingsford 2000; Hering <i>et al.</i> 2015
	Drastic changes in natural flow regimes	Impacts on aquatic communities	Jian <i>et al.</i> 2014; Huang <i>et al.</i> 2016
	Alterations in connectivity of rivers and floodplains	Impacts on nutrient cycling, threat to ecological integrity, sediment dynamics, and biotic diversity	Poff <i>et al.</i> 1997; Bunn & Arthington 2002; Naiman <i>et al.</i> 2002; Junk <i>et al.</i> 1989; Poff and Zimmerman 2010; Kennedy and Turner 2011; Zeiringer <i>et al.</i> 2018
	Inundation length	Changes in the community composition, growth and regeneration of the floodplain vegetation	Bren 1988
	Pollution (from chemicals, fertilisers, salinisation, sediment runoff, heavy metals, endocrine disturbance)	Lake acidification Impacting the health, growth and reproduction of aquatic organisms Eutrophication inducing algal blooms; Regime shifts	Pham <i>et al.</i> 2008; Du Laing <i>et al.</i> 2009 Hotchkiss <i>et al.</i> 2008 Khan <i>et al.</i> 2014 Zhang <i>et al.</i> 2012; Kattel <i>et al.</i> 2018
	Low dissolved oxygen	Hypoxic conditions, fish kills, impacts on photosynthesis of aquatic plants	Khan and Ansari 2005, Ansari <i>et al.</i> 2011
	Land use, deforestation, agricultural activities, dairy farming	Habitat loss due to stream channelization and conversion of wetlands to croplands	Wilcock 2013; van Soesbergen <i>et al.</i> 2019
		Deforestation causing soil erosion and increased sedimentation accumulation rates, declining water quality	Singer and Bissonnais 1998 Wang <i>et al.</i> 2008
	Over use of fertilisers	Loss of biodiversity, shifts in food chains, impairment of fisheries, and outbreaks of invasive species	Richter <i>et al.</i> 1996, Stein & Flack, 1997; Tilman, 1999
	Introduction of exotic species	Biological invasion, native biodiversity loss and structural complexity affecting habitat structure, biogeochemical cycling and consequently, food webs	Dudgeon <i>et al.</i> 2006; Kamenova <i>et al.</i> 2017; Reid <i>et al.</i> 2019

## 2.5.1 Understanding global climate change

Global climate has continually changed with natural variations in the earth's orbit, and in solar radiation, and these variations have taken place over time scales of millions of years down to a few years. These natural variations continue to exist today but at a very slow rate and their influence is rather small to explain the rapid warming seen in recent decades (IPCC 2014). It was suggested by Ruddiman (2003) that carbon dioxide emissions began

almost 6000 years ago. Over the past century, climate change impacts have been exacerbated by human activities such as the burning of fossil fuels, leading to the increased rate of global warming (IPCC 2014). Levels of carbon dioxide in the atmosphere over the past 800,000 years and temperature anomalies from 1880-2020 have been shown in Figures 2.5 and 2.6, respectively (NASA 2011). The records of 800,000 years of average global carbon dioxide levels fluctuated between about 170 ppm and 300 ppm. However, the situation changed rapidly upon the burning of fossil fuels in the industrial era with the recent carbon dioxide concentration of 412.5 ppm. This long-term reference makes it clear that carbon dioxide emissions and variations in the global temperature were low before the 19th century (Lüthi *et al.* 2008).

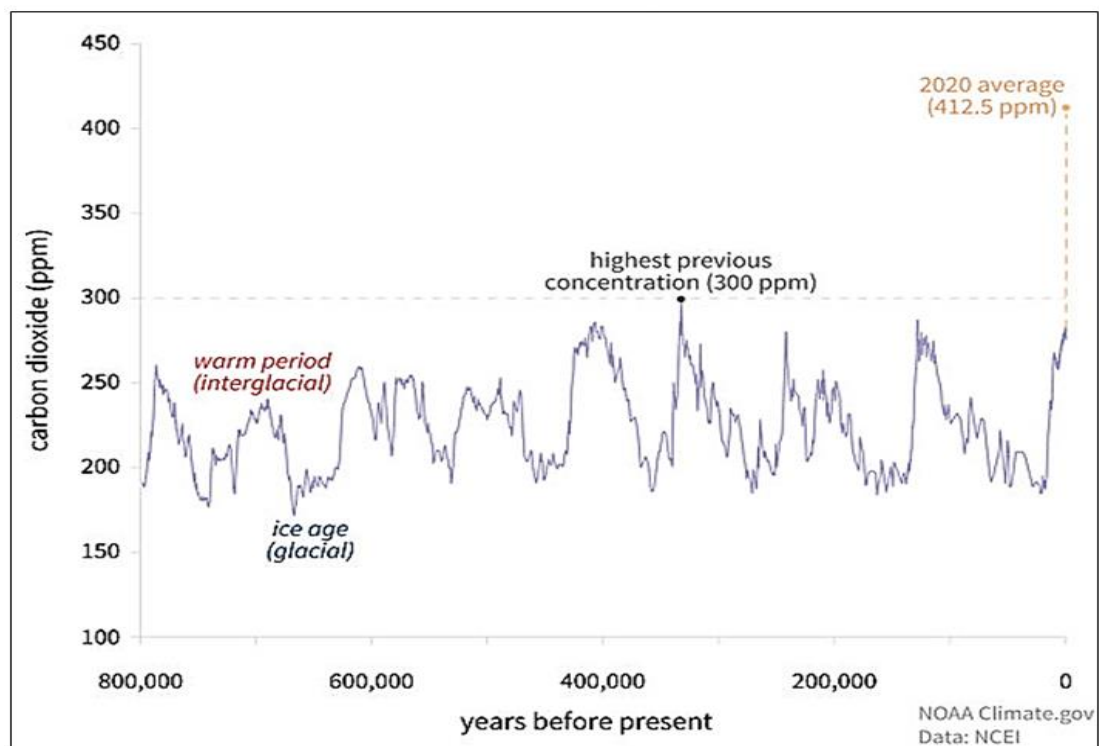


Figure 2.5 Levels of atmospheric carbon dioxide years before present. In 2020, carbon dioxide levels were 412.5 ppm (Source: NOAA 2020a)

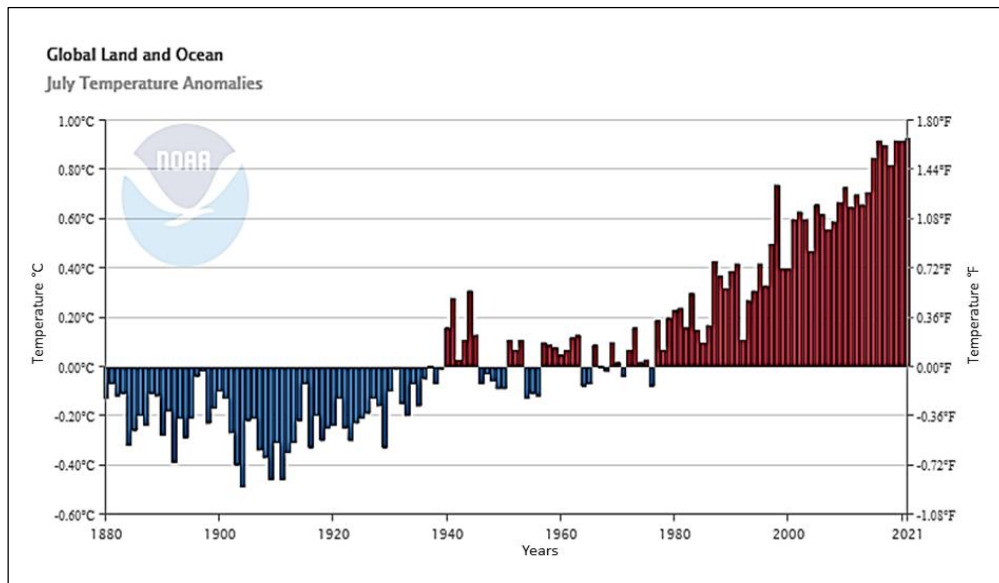


Figure 2.6 Comparison of yearly temperature to the twentieth-century average (red and blue bars) from 1880-2020 (Reprinted from: NOAA 2020b)

In Figure 2.6, global land and ocean temperatures increased slowly during the first half of the observational record in the late nineteenth and early twentieth centuries (Ruddiman 2003; Ruddiman *et al.* 2020). According to Fig 2.6, in 19th century, the temperature increased by an average of 0.05 °C per decade. However, over the next 7 decades, the rate of warming averaged 0.14 °C per decade. In essence, since the 1950s, the levels of carbon dioxide and temperature rates have been rapidly increasing.

Natural climate cycles vary over a range of time scales. Hydroclimatic variability arises from orbital (Milankovitch) forcing that generates glacial and interglacial cycles. Natural climate cycles will likely continue at its slow pace while emission of significant amounts of greenhouse gases into the atmosphere will be observed (IPCC 2021). These cycles drive high-amplitude variations in sea-level, temperature, and rainfall over millennia. Numerous global climate change models have predicted major changes in global climate owing to increases in temperature due to anthropogenic carbon dioxide emissions (IPCC 2021). These will lead to atmospheric circulation changes that will have significant impacts on the hydrological cycle.

## 2.5.2 Understanding climate of southern hemisphere and Australia

Based on the information from the instrumental records, it is revealed that the climate of southern hemisphere is influenced by a range of ocean–atmosphere processes operating in the Indian, Pacific, and Southern Oceans (Verdon-Kidd *et al.* 2014) that can bring prolonged periods of wet and dry conditions. These processes include: El Niño–Southern



Oscillation (ENSO), the Indian Ocean dipole (IOD) and Southern Annular Mode (SAM) (Pui *et al.* 2012; BOM 2019). Much of Australia lies across the high pressure belts of mid-latitude, thereby many arid and semi-arid regions exist in the country. Rainfall in Australia is highly variable with much inter-annual variability such that periods of drought and flooding are commonly observed (BOM 2019). The ocean-atmosphere oscillations involved in rainfall variability, droughts and floods in Australia are outlined in the following sections.

#### **2.5.2.1 *El Niño–Southern Oscillation (ENSO)***

ENSO has a big impact on rainfall variability in Australia (Myers *et al.* 2007). The Southern Oscillation Index, which is calculated by the difference in sea-level pressure between Tahiti and Darwin, is a common representation of ENSO fluctuation (Barr 2010; BOM 2019). The winds move from east to west over the surface of the tropical Pacific Ocean in the neutral state, bringing warm, humid air to Australia and the western Pacific in general while keeping the central Pacific Ocean chilly. El Niño is the negative phase of ENSO and is connected with warm ocean water in the equatorial Pacific Ocean's central and east-central regions. The consequence of prolonged El Niño phases for Australia are extended periods of severe drought (BOM 2019). However, La Niña relates to the positive phase of ENSO and refers to the extensive cool ocean water off the central and eastern tropical Pacific Ocean. These La Niña conditions strengthen the south-east trade winds, which drive and improve rainy conditions across Australia's eastern and northern regions (BOM 2019).

#### **2.5.2.2 *The Indian Ocean dipole (IOD)***

The variation in sea surface water temperature between the eastern and western sections of the Indian Ocean is known as the IOD (Cai *et al.* 2008). Many countries such as Australia, Indonesia, and Papua New Guinea are among those affected by the IOD's activities in the Indian Ocean Basin. The mechanism of IOD is similar to ENSO and it plays a major role in controlling rainfall variability over Australia (Cai *et al.* 2012). Sea surface temperatures rise in the western Indian Ocean during a positive phase and winds blow from the east, resulting in less rainfall in Australia due to the prevailing offshore winds while the opposite of this happens during a negative IOD phase (BOM 2019).

#### **2.5.2.3 *Southern Annular Mode (SAM)***

SAM is a north-south migration of the westerly wind belt that occurs in the middle and high latitudes of the southern hemisphere (Marshall 2003). These oscillations are one of the

main rainfall drivers in southern Australia. Further, on the decadal timeframe, the Interdecadal Pacific Oscillation (IPO), and the closely-related Pacific decadal oscillation (PDO), influence rainfall variability and the impacts of ENSO in large parts of the Southern Hemisphere (Power *et al.* 1999; Garreaud *et al.* 2009). The strong westerly wind belt compresses towards Antarctica during periods of positive SAM. Consequently, wind pressures across southern Australia are weaker than average and restrict the spread of cold and rainy weather across the region (Karpechko *et al.* 2009). According to reports, positive SAM was the primary cause of Australia's Millennium Drought, which lasted from 1997 to 2010 (BOM 2019). However, the strong westerly wind belt spreads towards the equator during a negative SAM, causing an increase in rainfall and storms across southern Australia.

It is evident from the foregoing discussion on climate change that in the next few decades and beyond, due to global warming, irregularity in temperatures is likely to be observed. This will cause increases in atmospheric moisture content, extreme heatwaves, and fewer frosts. Moreover, shifts in rainfall will occur such as increased rainfall in most tropical and high-latitude regions while dry conditions are likely to prevail in many subtropical and mid-latitude regions. Furthermore, the increase of extreme climate events can significantly influence the tropics and subtropics by increasing the number of intense cyclones, larger floods, and more extreme droughts and fires as a consequence of the increasing strength of El Nino and La Nina events (Rojas *et al.* 2019). Such changes will promote variability in hydrological conditions causing changes in freshwater ecosystems and associated wetland conditions such as some permanent wetlands will become seasonal or intermittent and some will be permanently flooded.

### **2.5.3 Understanding impacts of climate change on freshwater ecosystems**

Freshwater ecosystems such as wetlands have changed over millennia in response to this hydroclimatic variability (Pittock 2018). Hydrological modifications of wetlands include variability in the nature of the hydro period, and magnitude and severity of extreme weather events. For instance, in the USA, under the influence of the Atlantic Multidecadal Oscillation (AMO), fewer rainfall events occurred that led to the Midwest droughts in the 1930s and 1950s (Enfield *et al.* 2001). The discharge of the Mississippi River varied by 10% between positive and negative phases while inflows to Lake Okeechobee, Florida varied by 40%. These variations drive stratification, fluvial or aeolian sediment transport, nutrient release through erosion or fire events impacting lake's water quality. Furthermore, in south-eastern Australia, a long drought period of about 13 years was recorded at the beginning of the 21st century, but also at the beginning of the 20th century and in the

1840s. The recent prolonged dry conditions, that were linked with a positive IPO, had a remarkable impact on the river and wetlands of the MDB (Gergis *et al.* 2012). During the drought period, water levels declined to expose acid sulphate sediments resulting in acidification conditions. So, downstream river floodplains re-acidified because sulphur stored in the upper areas of the littoral zone re-oxidized, causing a decline in the population of many species (Kingsford *et al.* 2016).

#### **2.5.3.1 Climate change impacts and their ecological consequences**

Wetlands are extremely vulnerable to climate change owing to the primary importance of the hydrological regime that plays a significant role in controlling the ecological characteristics of wetlands (Junk *et al.* 2013). Climate change, mainly through alterations in precipitation and temperature regimes, is increasing the incidence of extreme flood and drought events, storms and hurricanes (IPCC 2021). This will, in turn, have substantial impacts on the water volume and so, hydrological conditions of wetlands.

The water volume in a wetland directly influences ecosystem functioning by determining the extent of suitable habitat for littoral and planktonic species and many aspects of water quality. The seasonal dynamics of the water volume of a wetland is determined by regional precipitation and runoff regimes. Heavy precipitation events are likely to increase by the middle of the 21st century, causing increased annual average runoff and water availability by 10-40% at high latitudes and in some wet tropical areas. In contrast, a decline in runoff by 10-30% is anticipated in some dry regions at mid-latitudes and in the dry tropics (IPCC 2021). Therefore, a change in climate that alters the existing hydrologic regime has the potential to greatly modify habitat suitability for many species and to cause significant ecological changes.

Over the past century, episodes of extreme events have increased in Australia, North America, Japan, and in some parts of Europe (IPCC 2021). These variations in global climate affect the hydrology of wetlands with periods of both extreme drought and flooding. Rivers need floods to sustain habitats and to support biological productivity and biodiversity and so they are beneficial for floodplain wetlands. Further, flood events play a significant role in primary and secondary production, growth, and development of the native plant, animal, and nutrient dynamics (Junk *et al.* 1989; Atkinson *et al.* 2019). Initially, during the onset of a flood event, as water levels are high, primary production is inhibited. At later stages, when water levels are lower but the water is rich in nutrients, is when the floods most subsidise aquatic food webs (Alford and Walker 2013). However, unpredictable flood events can transport excessive nutrient loads causing eutrophication of waters or they have negative consequences on primary productivity resulting in shifts

to unfavourable species (Paerl *et al.* 2016). For example, following flood events, the phytoplankton biomass in Lake Winnipeg increased and the phytoplankton community shifted to include more cyanobacteria (McCullough *et al.* 2012). Consequently, the water quality of Lake Winnipeg declined owing to the excessive production of harmful algal blooms. Therefore, increased primary production post-flood is considered a positive outcome but if production is excessive then flooding results in declines in ecosystem services. Further, Holocene records of floodplains in the USA illustrate that, in arid regions, the intense floods cause widening of channels where sparse riparian vegetation is unable to maintain bank stability (Carpenter *et al.* 1992).

Climate change induced flooding alters river channels, erosion, nutrient, and sediment transports in wetland ecosystems (Palmer *et al.* 2009; Arnell *et al.* 2015). Given the sensitivity of many wetlands to the annual hydrological cycle, even small alterations in the annual pattern, such as changes in the timing and intensity of hydrological events, may impact floodplain-river interactions. Many MDB wetlands of Australia suffer from decreased river connectivity causing ecological issues. These ecological consequences of hydroclimate variability on floodplain lakes include alteration of river channels causing erosion, shifts in community structures, productivity changes, and increased algal blooms (Pachauri *et al.* 2014; Atkinson *et al.* 2019; Reid *et al.* 2019). Therefore, soil erosion and increased sediment transport impact water quality and aquatic communities of the floodplains of the MDB in Australia (Gell *et al.* 2009).

#### **2.5.3.2 Role of temperature and carbon dioxide in cyanobacterial blooms**

Another key factor regulating climate change is carbon dioxide which is considered the primary gas responsible for recent rises in atmospheric temperature. Elevated concentrations of atmospheric carbon dioxide and temperature have been found to be responsible for the development of harmful cyanobacterial blooms in eutrophic waters. For example, Lake Volkerak, a large eutrophic lake in The Netherlands, has experienced a rise in cyanobacterial blooms and the condition was stimulated by elevated temperature and rising carbon dioxide concentrations (Visser *et al.* 2016). A similar condition was observed in Lake Taihu in China, where nutrient enrichment together with increased lake water temperature has led to the succession of cyanobacteria, such as *Microcystis*, causing toxic water conditions (Wang *et al.* 2014). As a consequence, the water quality of many lakes, reservoirs, and brackish ecosystems across the globe has declined with the production of harmful algal blooms.

Many biological activities are controlled by temperature such that a change in the thermal regime can directly regulate growth and habitat preference posing a risk to biodiversity

(Bellard *et al.* 2012). For example, many organisms are adapted to a specific range of temperatures, so increased temperatures could exceed the thermal optima for some cold-water species, mainly in summer (Lloréns 2006). The capacity of species to tolerate temperature is different, so, temperature changes can produce shifts in species composition that, in turn, can affect the overall metabolism and productivity of wetlands. It is expected that global warming will shift the potential geographic ranges of species (populations) to poleward, or upward (in highlands) regions. Similarly, the equatorward (or lower-elevation) part of the present geographic range of many species will become unsuitable. The geographical distribution ranges of species largely depend upon climate variables such as warming, precipitation, and moisture. Under unfavourable climatic conditions, species shift to suitable places to increase their survival and reproduction (Bellard *et al.* 2012) or, in many cases, become extinct locally.

Although there is relatively limited evidence of current extinctions caused by climate change, the available studies suggest that climate change could surpass habitat destruction as the greatest global threat to biodiversity over the next few decades (Leadley 2010; Atkinson *et al.* 2019). This suggests that climate change is likely to have major impacts on the world's wetlands, their flora and fauna as well as the delivery of the many ecosystem services which they provide.

#### **2.5.4 Anthropogenic impacts and their ecological consequences**

Natural aquatic systems have been under the impact of anthropogenic pressures for millennia at local and regional scales. The direct (e.g., through river regulation, species introductions) and indirect (e.g., through altering lake catchments) anthropogenic pressures on aquatic systems are undisputable (Hering *et al.* 2013). Given the objective of this research, this review focuses on the impacts of regulation and pollutants on wetlands but first briefly outlines the ecological changes associated with anthropogenic activities in freshwater ecosystems.

##### **2.5.4.1 River regulation and its associated impacts**

Rivers are regulated to serve human needs such as flood protection, transport, irrigation, drinking water supply, and hydropower generation. River regulation, water abstraction, channelisation particularly, through the construction of large headwater dams, has altered the flow regime of rivers worldwide (Dynesius and Nilsson 1994; Ligon *et al.* 1995; Graf 1999; Kingsford 2000). Globally, there are over 45,000 dams, over 15 m high, impacting over half of the world's rivers (Nilsson *et al.* 2005). The construction of rivers has led to a reduction in the connectivity of rivers to their surroundings. Thereby, nutrient cycling,

sediment dynamics, and biotic diversity (Junk *et al.* 1989; Poff *et al.* 1997; Kennedy and Turner 2011) are impacted because flow modifications reduce flood peaks, flooding frequency, and duration, and change the nature of dry periods (Poff and Zimmerman 2010).

Globally, about 50% of all rivers demonstrated a reduction in their connectivity, in large rivers, unbroken natural connectivity exists only in remote areas of the Arctic, Amazon Basin, and to some extent in the Congo Basin (Grill *et al.* 2015). Statistically, 63% of the world's very long rivers (length >1000 km), representing 41 % of the total river volume, is no longer free-flowing. For example, free-flowing rivers are mostly absent from the mainland of the United States, Europe, Mexico, and the Middle East, except in parts of India, China, southern Africa, southern South America, and most of southeast Asia and southern Australia (Grill *et al.* 2015). These alterations of flow regimes are considered as a serious issue and a continuing threat to the ecological integrity of river– floodplain ecosystems (Bunn and Arthington 2002; Naiman *et al.* 2002; Poff and Zimmerman 2010; Zeiringer *et al.* 2018). The impact of such river alterations is aggravated by climate change, which potentially enhances the flow intermittency (Reynolds *et al.* 2015). This disturbance in the natural flow regime further induces changes in the species distribution and interaction that is likely to contribute to the decline of resilient riverine communities (Strona and Lafferty 2016) and ecosystem services (Tonkin *et al.* 2018).

Quantifying the effects of regulation on a river system's wetland flooding regime requires an understanding of the relationship between river flow, wetland flooding, and natural variability (Tockner *et al.* 2010). The interruption in the flow regimes and flooding frequency has significantly influenced large river systems. For example, rivers are threatened by dams under construction or planned for construction, especially in developing regions, such as the Yangtze River Basin in China and the Tigris and Euphrates River Basin in the Middle East (Millennium Ecosystem Assessment 2005). The construction of Three Gorges Dam has caused drastic changes in the natural flow regimes of the Yangtze River Basin and impacted the aquatic communities of the river (Jian *et al.* 2014; Huang *et al.* 2016). The dramatic loss and degradation of many individual wetlands are observed in the Tigris and Euphrates River Basin in the Middle East. A noticeable decrease in the surface area of the Mesopotamian marshes (located between the Tigris and Euphrates Rivers in southern Iraq) from 15,000–20,000 km<sup>2</sup> in the 1950s to less than 400 km<sup>2</sup> in 2005 was observed owing to excessive water withdrawals and construction of dams. Further, channelisation and levees reduce overbank flows leading to an alteration in flooding frequency. Along channelled rivers, flood plains only receive surface flooding from the river during major flood events. For example, in regulated rivers such as the

Austrian Danube, floodplain inundation is short and the increase in inundation area is abrupt. This has caused a loss of vegetation, increased erosion and has altered water levels.

The water level is an important factor in determining the vegetation pattern, community dynamics, and plant growth in a wetland. A decline in water level significantly influences vegetation and dependent aquatic communities (Yang *et al.* 2016). In dam-regulated rivers, relatively small water level fluctuations are often observed in downstream areas (Magilligan and Nislow 2005) and these could pose detrimental impacts on the ecosystems of river-connected lakes in various ways. For example, if a lake bottom is exposed to air for an extended duration, it can cause potential wetland degradation (Zhang *et al.* 2012b; Wang *et al.* 2013). In the past two decades, several environmental perturbations were observed in Greek lakes and wetlands. These include a progressive dramatic reduction in water level leading to cultural eutrophication, habitat degradation, loss of fishery potential, and endemic species (Coops *et al.* 2003; Jeppesen *et al.* 2015). Moreover, freshwater biodiversity is affected by flow regulation and habitat degradation (Dudgeon *et al.* 2006). Upon dam construction, the water levels are interrupted, the conditions which were favourable to lotic benthic species are replaced by those suitable for lentic organisms (Hirsch *et al.* 2017). Consequently, the population of phytoplankton is favoured leading to declining water quality. However, due to increasing water levels, the area of land submerged is increased, leading to significant impacts on the native flora and diverse floodplain habitats.

#### **2.5.4.2 Understanding impacts of pollutants on freshwater ecosystems**

Pollution in freshwater ecosystems mainly started to appear in the 1950s owing to the anthropogenic activities being the major cause behind the release of pollutants into these water bodies (Millennium Ecosystem Assessment 2005). The major pollutants are categorised into organic (e.g., pesticides, herbicides) and inorganic pollutants (e.g., nutrients, fertilizers, human and animal waste, suspended sediments from runoff, and heavy metals) (Kingsford *et al.* 2016). These pollutants have the potential to degrade water quality and harm the biodiversity of freshwater ecosystems. Anthropogenic activities (e.g., chemical industries) are also responsible for the emission of harmful chemicals into the atmosphere that can alter the pH levels of freshwater ecosystems via acid rain events (Kumaraswamy *et al.* 2020). Most of the freshwater aquatic plants and organisms survive best in water with a pH of 7.0-9.2. The growth of plants declines in low pH conditions which further decreases nourishment to some aquatic birds. At low pH the number of freshwater shrimps, crayfish and clams start to decline and the bacteria that decompose

leaf litter also begin to perish impacting the nutrient source of plankton. An emerging water quality concern is the impact of personal care products and pharmaceuticals, such as contraceptive pills, painkillers, and antibiotics, on aquatic ecosystems. There is growing evidence of pharmaceutical pollution (personal care products, endocrine disruptors) where river areas are near large animal or human populations (Jeffries *et al.* 2010), leading to impacts on health, growth and reproduction of aquatic organisms (Hotchkiss *et al.* 2008).

Further, salinity is another determining factor of water quality. Dryland salinity occurs when deep-rooted native plants are removed or replaced with shallow-rooted plants that use less water (Figure 2.7). However, leakage of irrigation channels and inefficient irrigation practices usually result in mobilisation of stored salts into the root zones causing irrigation salinity (Figure 2.8). These large volumes of irrigation water create imbalance in vegetation and water supply affecting soil composition through salt leaching, raising the water tables that brings salts to the surface (Slinger and Tenison 2005). Therefore, increased salt levels cause salinisation of soils and wetlands and also increases the threat of acidification (Du Laing *et al.* 2009) that reduces species richness in freshwater ecosystems. For example, the Coorong and Lower Lakes wetland in south-eastern Australia are Ramsar-listed significant sites that experienced significant acidification, following drawdown (Kingsford *et al.* 2011). Acidification mobilizes heavy metals that can significantly impact freshwater biota sometimes leading to genetic abnormalities (Kumaraswamy *et al.* 2020).

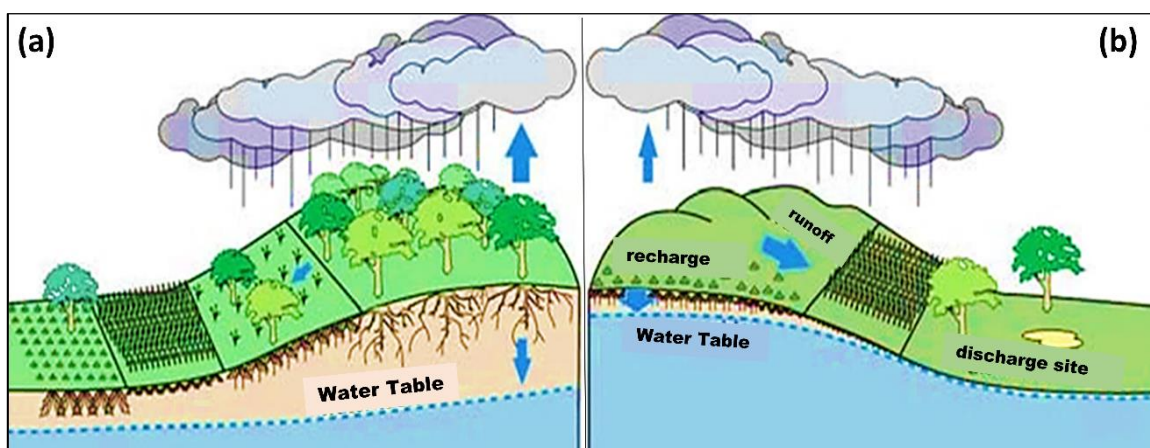


Figure 2.7 An overview of causes of dryland salinity. Impacts of vegetation on water tables shown by a hydrologically (a) balanced system which shows deep- rooted trees and native vegetation using most of the water that enters soil ensuing reduced leakage and, (b) an unbalanced catchment demonstrating removal of vegetation and increased leakage to groundwater system leading to increased water table that brings salt to the soil surface and root zones (Reprinted from: Slinger and Tenison 2005)



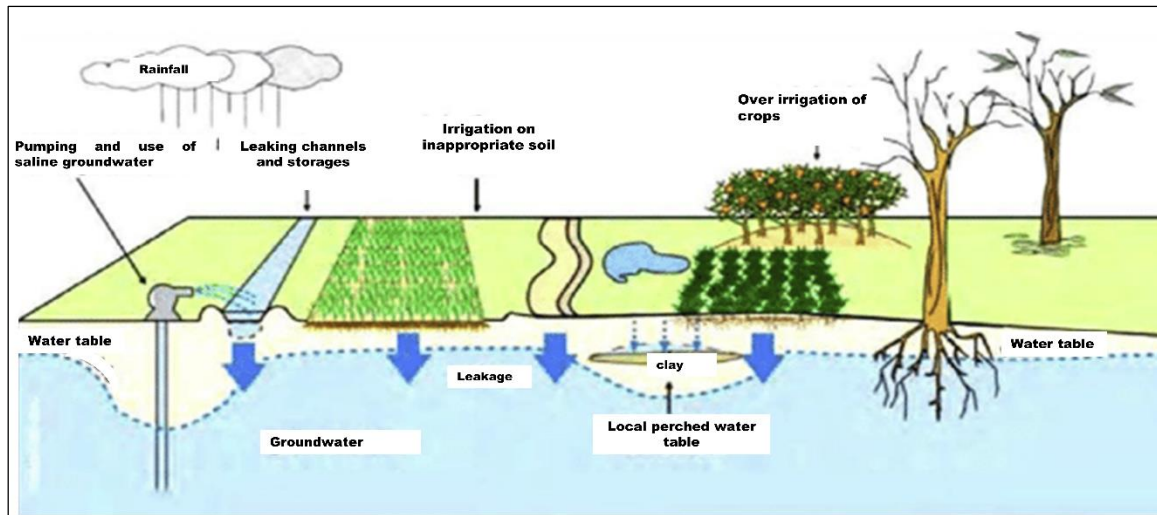


Figure 2.8 Irrigation salinity- water levels increased due to excessive application of water through irrigation channels (Reprinted from: Slinger and Tenison 2005)

#### 2.5.4.3 Eutrophication and anoxia

Eutrophication is the condition when a water body becomes overly enriched with nutrients inducing the growth of algae (Khan *et al.* 2014). Nutrient loading (primarily Nitrogen and Phosphorous) is a natural process occurring in wetland ecosystems. However, shifts from oligotrophic (clear water, low nutrient) to productive, eutrophic conditions (elevated nutrient supply) and cultural eutrophication of aquatic ecosystems are recognised as severe anthropogenic impacts (Smith and Schindler 2009; Alexander *et al.* 2017; Li-kun *et al.* 2017). The watersheds dominated by agriculture and human settlement have seen significantly increased nutrient levels of rivers, streams, and lakes (Benoit and Fizaine 1999; McFarland and Hauck 1999; Cuffney *et al.* 2000; Berka *et al.* 2001; Wang 2001). The elevated nutrient concentrations drive the growth of filamentous and floating algae which form scum layers following death. This, in turn, restricts light penetration and impacts on the photosynthesis of aquatic plants (Khan and Ansari 2005; Ansari *et al.* 2010). The resultant death and decay of aquatic plants lead to foul-smelling and turbid water and loss of oxygen. For example, in Lake Erie which is one of the largest of the Great Lakes in North America, eutrophication due to mainly phosphorus enrichment, resulted in huge blooms of floating blue-green algae and attached green algae (King *et al.* 2017). This reduced light penetration into the lake and reduced photosynthesis and oxygen production by other phytoplankton groups.

In addition, nutrient inputs from agricultural activities and other human activities are considered to be driving hypoxia in the Gulf of Mexico (Randall and Goss 2008; Rabalais and Turner 2019) and many deep lakes worldwide (Jenny *et al.* 2016). The function and

services of aquatic ecosystems are threatened by the spread of hypoxic conditions. Since the 1980s, in Lake Taihu, China, increased chemical oxygen demand, total nitrogen, and total phosphorous levels have been observed resulting in the corresponding decline in littoral macrophytes, increased sediment resuspension, poor water quality and increased biomasses of phytoplankton and smaller zooplankton (Zhang *et al.* 2006; Guan *et al.* 2011). Similar research has been undertaken on the shallow Lake Osmar in Brazil (Moi *et al.* 2020) which emphasised that species richness and zooplankton were highest during the clear water state.

Ample literature indicates changes in the natural and ecological properties of wetlands with the consistent conclusion that water quality and habitat quality of these systems are significantly impacted by eutrophication (Piceno and Lovell 2000; Álvarez-Cobelas *et al.* 2001, Guntenspergen *et al.* 2002, Liston *et al.* 2008). Consequently, many researchers (eg., Reddy and DeLaune 2008; Vadas *et al.* 2005, Ansari *et al.* 2010) have made considerable efforts in understanding the role of high levels of phosphorous in favouring eutrophication that drives regime shifts in floodplain lakes. In the 1950s, the phosphorous concentration in the Yangtze River floodplain lakes exceeded 100 µg/L owing to increased land-use activities. This has caused a regime shift, where a large number of lakes with macrophyte-dominated states were transformed into algal-dominated states (Zhang *et al.* 2012a; Kattel *et al.* 2018).

Davis and Koop (2006) have reported on the eutrophication of Australian inland rivers and estuaries while recent studies investigating effects of eutrophication have rarely been addressed for Australian wetlands. Although some information about nutrients and eutrophication is available for MDB wetlands (Gell *et al.* 2009) there is a spatial gap in the existed information. It is this important gap that this study seeks to fill.

#### **2.5.4.4 Land use and habitat degradation**

In many parts of the world, land-use changes have directly affected freshwater ecosystems. Tockner and Stanford (2002) reported that up to 90% of the floodplains in the river basins of Europe and North America are altered by land-use changes with serious consequences in floodplain functionality. Land-use changes, including the increasing extent and intensity of agricultural activities are considered as the primary driver of habitat and biodiversity loss and indirect cause of impacts on the water quality of freshwater ecosystems (van Soesbergen *et al.* 2019). One intensive form of agriculture is dairy farming (Wilcock *et al.* 2013). Significant alterations in land use changes, such as from sheep and beef cattle grazing to more intensive dairy farming, are known to impact the

water quality and ecology of streams and rivers (Larned *et al.* 2004; Howard-Williams *et al.* 2010).

The development of drainage channels and clearance of vegetation for agricultural purposes, poses the risk of increased runoff, soil erosion, and sedimentation accumulation rates (Li *et al.* 2014) and have direct and indirect impacts on Australian inland systems (Finlayson *et al.* 2005; Kingsford *et al.* 2016). Direct impacts are associated with loss of habitats displaced by crop or pasture lands while indirect effects are linked with water quality issues. The construction of dams affects variability in water levels which in turn influences the availability of aquatic habitats such as pelagic or littoral environments (Logez *et al.* 2016). However, indirect effects are caused by phosphorus, nitrogen and other contaminants of agricultural runoff, as well as the hydrologic alteration in the form of altered volume and timing of runoff. These factors impact the water quality and ecological condition of the water bodies. The Sky Lake and White River in the Mississippi River system, USA have experienced increased erosion and sedimentation caused by catchment modifications and hydrological alterations and these are considered to be the key drivers of water quality problems and habitat degradation in the Mississippi River Basin (Wren *et al.* 2008; Bhattacharya *et al.* 2016).

#### **2.5.4.5 Introduction of exotic species**

Biological invasions of freshwater ecosystems are global in scale (Dudgeon *et al.* 2006), particularly of freshwater fish species and aquatic plants. Freshwater ecosystems are vulnerable to biological invasions because of high levels of isolation and endemism (Vander Zanden and Olden 2008; Walsh *et al.* 2016). Therefore, the introduction of exotic species into lakes and wetlands can significantly impact their ecosystem structure and functioning owing to the new disturbed regime and high floodplain productivity (Vander Zanden and Olden 2008). Exotic species compete with native species which often results in the loss of native biodiversity and structural complexity affecting habitat structure, biogeochemical cycling, and consequently, the nature of food webs. Displacement of habitats and subsequent extinction of native populations are reported as some of the foremost impacts of introduced species in many large river floodplain lakes (Reid *et al.* 2019).

It has been documented that river basins of the Northern Hemisphere support the highest number of non-native fish species (Leprieur *et al.* 2008). More than 50% of the biota of the Hudson River in the USA comprises introduced species, mostly from Europe, where 10% of the populations have significant ecological impacts on native populations (Nilsson and Berggren 2000). Human activities are exacerbating the establishment of non-native

species by disturbing natural landscapes (Leprieur *et al.* 2008). Due to a lack of competition with native species, non-native species can spread at a faster rate in large river systems (Koehn 2004). It is evident that the impacts of invasion on native populations have been increasing, including the global distribution of aquarium fauna from electronic purchases (Reid *et al.* 2019).

In regulated environments of the Southern Hemisphere, the nature of the invading species in susceptible habitats is also becoming unpredictable (Bunn and Arthington 2002). For example, since European settlement, alien species such as European perch (*Perca fluviatilis*) and common carp (*Cyprinus carpio*) have successfully become established in the Murray River Australia. This has led to declining growth and development of native fish populations (Koehn, 2004). Moreover, several endemic species, including Murray hardy head (*Craterocephalus fluviatilis*), Macquarie perch (*Macquaria australasica*), and Murray cod (*Maccullochella peelii peelii*) have become critically endangered or vulnerable on account of invasive species (Hutchison and Armstrong 1993). Also, in a large river system of South Africa (Orange-Vaal River System), water regulation has maintained the natural flow regimes favouring the alien aquatic vegetation (e.g., *Myriophyllum* spp., *Azolla* spp.), thereby reducing the light penetration, water movement and oxygenation followed by the displacement of the native vegetation bed (Ashton *et al.* 1986; Wasserman *et al.* 2011).

According to the above discussed causes of wetland changes, it is revealed that modern floodplain wetlands are extensively regulated, polluted, habitats with low connectivity. Since the 1950s, they have become increasingly vulnerable to the accumulation of pollutants (Millennium Ecosystem Assessment 2005), invasive species (Thomaz *et al.* 2015), and nutrients (Smith and Schindler 2009) generated by human activities (Saulnier-Talbot 2016). In the 1970s, the concept of wetland mitigation was first developed in the USA which was later adopted by Australia. This provides a framework to avoid damage to the natural environment caused by development activities (Burgin 2010). Thereupon, several regulations were introduced to conserve the natural environment and ecosystem services. Despite these apparent on-going mitigation attempts of conserving the environment, the loss of species, habitat and water quality degradation persist and are exacerbated by climate change. Cumulatively, multiple threats and alterations can cause combined effects on freshwater ecosystems. These impacts, along with their ecological consequences, have been summarised above (see Table 2.1). Hence, to establish an understanding of the detrimental impacts of human and climate change, it is appropriate to explore the interaction of those impacts with the processes that support the structure

and functionality of the ecosystem (Poff *et al.* 1997). This can be achieved by adopting paleolimnological assessments.

## **2.6 The value of paleolimnology as a technique for assessing wetland values**

Paleolimnology is the study of the physical, chemical, and biological information that is preserved in lake and wetland sediments (Whitmore and Riedinger-Whitmore 2014; Gell *et al.* 2018b). It plays a vital role in reconstructing natural and anthropogenic changes in the past when instrumental records are absent. The environmental and ecological conditions in lakes and rivers change on a wide range of time scales. However, obtaining long-term data about these changes is one of the greatest challenges faced by many ecologists, environmental scientists, and limnologists. Without such data, it is difficult to understand the past changes that are occurring in the aquatic systems and the likely drivers of ecosystem changes. A record of past conditions is preserved in the sediments of lakes and river systems that accumulate in an organised manner with older deposition occurring deeper in the sediment core and the most recent material to be found at the surface of the core. Paleolimnological techniques can often be used to reconstruct information that is preserved in such sediment sequences. In the past three decades, various advancements in the paleolimnological approaches have been seen such as retrieving and sectioning of sediment cores, provision of robust geochronological information as well as major developments in interpreting stratigraphic changes in a statistically robust manner. Anthropogenic disturbance may be determined based on long-term records of environmental change and a range of natural conditions such as reference or background conditions. Once the baseline conditions are established from the past, non-degraded environmental condition that occurred before any human interference from which the direction and magnitude of subsequent human-induced change in a given ecosystem through time is measured (Figure 2.9). Radiometric and dating technologies can be applied to establish the chronology of the cores (Skilbeck *et al.* 2017) to provide dates at depth in sedimentary profiles. The nature of exploration of long-term environmental and ecological change in a particular region can be determined by referring to a range of paleolimnological indicators.

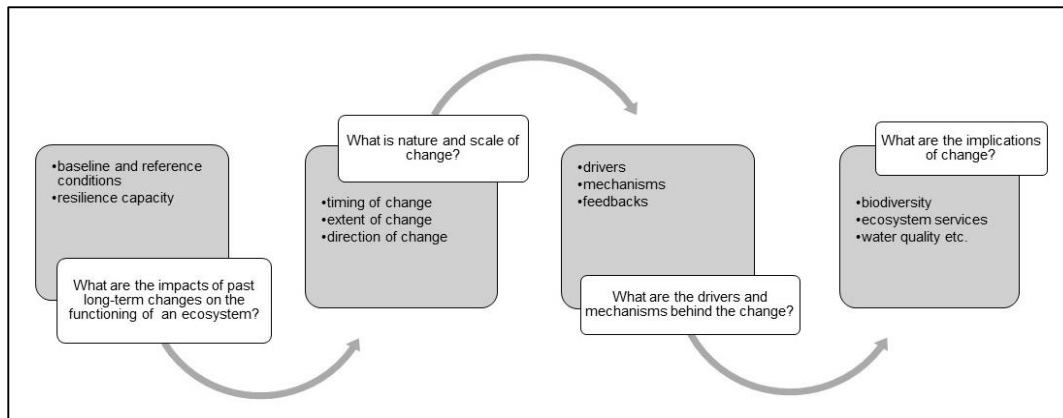


Figure 2.9 Application of the paleolimnology approach to inform management and conservation of ecosystems

## 2.6.1 Paleolimnological indicators of past conditions

The ideal paleolimnological indicators are abundant, diverse, and are well preserved in sediments providing information of past physical, chemical, and biological changes. In paleolimnology approaches, a wide range of proxies are used to reconstruct change in freshwater systems and these are listed in Table 2.2. This section outlines the proxies (such as elements, sediment accumulation rates and, stable isotopes of carbon and nitrogen, diatoms, pollens, ostracods, cladocerans) used in paleolimnological assessments. Also, a brief overview of the proxies used in this study – pollen, diatoms, Itrax elements,  $^{210}\text{Pb}$  dating and sedimentary proxies of total organic content, is presented in the subsequent paragraphs to familiarise the reader with the variation in their characteristics and application in paleoecological reconstruction.

**Table 2.2 Summary of physical, chemical and biological proxies used in paleolimnology studies (Adapted from: Gell *et al.* 2018b)**

	Proxy/Indicator	Measure	Inference
<b>Physical</b>	Particle size	Silt, clay and sand	Fluvial energy
	Sedimentation accumulation rate	Lead-210 and Cs-137	Erosion Inputs
	Colour, varves, laminations	Visual inspection	Oxygen conditions
<b>Chemical</b>	Organic content	Loss-on-ignition	Productivity
	C and N isotopes of bulk sediments and biological remains	$\delta^{13}\text{C}$ ; $\delta^{15}\text{N}$	Organic matter inputs and C & N biogeochemistry
	O isotope of carbonaceous or shells	$\delta^{16}\text{O}$ , $\delta^{18}\text{O}$	Temperature
	Elements	Ti, Ca, Si, K, S, P etc.	Detrital inputs, source of sediments
<b>Biological</b>	Diatoms	Percentage of species assemblage, indices; transfer functions	Salinity, nutrients, pH, turbidity, water source, habitat structure
	Pollens	Species; presence of exotics taxa	Vegetation, dating
	Cladocerans	Species assemblage	Habitat, food-web structure
	Ostracods	Species assemblage; shell chemistry	Salinity, anoxia
	Chironomids	Species assemblage	Anoxia
	DNA	Presence/absence or relative change in abundance of a given taxon	Extinctions; invasions; population dynamics and diversity

### **2.6.1.1 Physical Indicators (colour, texture and composition)**

Examination of the physical appearance of sediment provides information about the colour changes, texture, composition, and presence of laminae etc. A variety of other techniques, such as radiography and X-ray, often reveal information about the physical structure and composition that is not visible to the naked eye. Also, particle size analysis of sediments provides insights to determine the processes and source of detrital inputs. Sediment sequences also archive magnetic signals that can be used in interpreting past erosion events (Maher 2011). In addition, determination of sediment accumulation rates plays an important role in constructing chronology of the sediment cores.

### **2.6.1.2 Chemical indicators (elemental analysis, stable isotopes of C and N)**

Stratigraphic analysis can be useful to reconstruct anthropogenic environmental changes such as catchment disturbance and metal pollution and the transport of other elements to the lake ecosystems (Guyard *et al.* 2007; Niemann *et al.* 2013). These changes are reflected in the inorganic elements that accumulate in lake sediments. In particular, concentrations of chemical elements, such as iron (Fe), titanium (Ti), and phosphorus (P), potassium (K) in sediments and other archives can be used to determine past changes in lake productivity, erosion, and land use activities. The intensity of erosion is influenced by changes in precipitation and streamflow as well as changes to the land such as vegetation clearance. Changes in aquatic productivity may reflect precipitation-related fluctuations in nutrient input from the land. In addition, land use changes induced by agriculture or urbanization can cause deposition of elements released from fertilizers, sewers, and other systems. These elemental depositions can be detected by Itrax-XRF (X-Ray Diffraction) scanning. This core-scanner provides a continuous record of relative elemental composition, with variations corresponding to environmental and sedimentological changes (Croudace *et al.* 2006). It provides a stratigraphic analysis of the sediments by providing insight into the geochemical composition of the core sequences.

An important application of XRF core scanning is to elucidate changes in the relative amount and nature of the material that is derived from the lake catchment. Detrital inputs are indicated by the lithogenic elements Al, Si, K, Ti, Fe, Rb, and Zr because they are conservative in most geochemical environments (Boës *et al.* 2011). Furthermore, XRF scanning can also be used to determine in-lake processes such as redox conditions and their response to climate and environmental changes (Aufgebauer *et al.* 2012). A variety of factors can affect redox conditions such as changes in water depth, biological productivity, rapid sediment deposition as well as climatic drivers such as temperature, rainfall, and wind regime. Under the influence of reducing conditions, the solubility of Fe

and Mn increases, with Mn being more affected (Boyle 2002), so an increase in Fe/Mn ratios can signal the onset of anaerobic conditions. These high values of Fe/Mn may indicate hypoxic conditions in bottom waters during enhanced stratification, or anoxic conditions that arise from organic decay following enhanced biological productivity linked to changing nutrient input. For example, at Lake Potrok Aike in Patagonia, increased lake mixing conditions are interpreted by low Fe/Mn ratios, possibly as a result of lower lake levels or increased wind speed (Haberzettl *et al.* 2007).

XRF scanning can also be used to make inferences about the biological components of lake sediments. The ratio Si/Ti has been used to estimate biogenic silica (Brown *et al.* 2007; Stansell *et al.* 2010), where increasing ratios represent enhanced diatom productivity. Moreover, XRF scanning has significantly contributed to the development of paleoclimate records from lake sediments, such as long records of Quaternary climate variability (Scholz *et al.* 2007; Melles *et al.* 2012), identification of rapid, abrupt climate shifts (Kylander *et al.* 2011) and inter-annual climate variability (Metcalf *et al.* 2010). High-resolution analysis of magnetic susceptibility is used to trace amounts of terrigenous input which occurs during periods of high sedimentation as a result of increased erosion in wet periods (Marshall *et al.* 2011).

The stable isotopes of carbon (C) and nitrogen (N) provide an alternative means to detect early environmental changes in aquatic ecosystems (Wang *et al.* 2015). Ratios of stable isotopes from the same element can be measured from archive material to infer a wide range of information about past environments. Stable isotope analysis of nitrogen and carbon ( $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$ , respectively), and the C/N ratio, have been widely used to elucidate the distribution and pathways of organic matter within an ecosystem (Meyers and Ishiwatari 1993; Talbot 2001; Vermeulen *et al.* 2011; Kattel *et al.* 2020). The elemental ratio of total organic carbon (TOC) to total nitrogen (TN), expressed as C/N or TOC/TN, has been widely used as a proxy to determine sources of organic matter (OM) in a lake, as these can be allochthonous or autochthonous (Meyers 1997). Plants of relatively simple characteristics such as freshwater phytoplankton (autochthonous input) have isotopic signatures published in the literature. They typically have  $\delta^{13}\text{C}$  values from  $-35.0\text{‰}$  to  $-25.0\text{‰}$  (Boutton 1991) and  $\delta^{15}\text{N}$  values around  $5\text{‰}$  (Wada and Hattori 1990) and C/N values between 1-10 (Meyers 1997). However, any value beyond these limits likely refers to allochthonous or anthropogenic inputs. It is beneficial to gather evidence about the dynamics of organic matter due to its significant role in the functioning of aquatic ecosystems. Organic matter influences carbon and nitrogen cycles through the movement of dissolved organic carbon and dissolved organic nitrogen (Inamdar *et al.* 2012) and alters aquatic trophic states (Zhang *et al.* 2010). Therefore, eutrophication is considered



to be the result of the input of terrestrial organic matter in aquatic ecosystems (Massicotte *et al.* 2017).

Brenner *et al.* (1999) reported on the past trophic status and stratigraphic shifts in the stable isotope records from four Florida lakes using isotopic signatures to assess baseline trophic conditions. Moreover, stable isotopes of carbon and nitrogen have been used to compare the ratio of planktonic and benthic algae to understand water turbulence and temperature (Doi *et al.* 2010; Wang *et al.* 2013). Perga (2011) examined diagenetic changes in  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  values of cladoceran exoskeletons using both laboratory and field experiments. It was investigated that most changes took place during settling in the water column, while microbial breakdown of organic exoskeletons in sediments selectively liberated isotopically light C and N compounds. Seasonal changes in the communities were tracked by  $\delta^{15}\text{N}$ , whereas  $\delta^{13}\text{C}$  values reflected the community during stratification. This study suggested that C and N isotopes might be subject to different physiological processing within Cladocera and highlights that post-depositional, and biotic processes drive variability in stable isotope values.

### **2.6.1.3 Biological Indicators (ostracods, cladocerans, diatoms and pollens)**

Ostracods are bivalved crustaceans with two calcite valves. They can be found in most aquatic environments, e.g., in continental, estuarine, marine, and hypersaline waters (Holmes 2003). Shell calcification takes place quickly in response to variable environmental conditions. Thus, the chemistry of ostracod shells reflects the chemical conditions of the water such as temperature, salinity, dissolved ion composition and hydrology (Mischke *et al.* 2008). Hence, they can be used as a proxy to track a wide spectrum of environmental variables.

Trophic states, type of vegetation, and other ecological conditions can be determined by the remains of aquatic invertebrates such as zooplankton and chironomids (Kattel 2012b). Cladocerans play a central role in lacustrine food webs and are sensitive to changes in resource availability (bottom-up effects) and predation (top-down effects). For instance, over the past 200 years, in Lakes Chaohu, Liangzi, and Zhangdu of China, a major transition of cladoceran species occurred. Large-sized cladoceran species changed from *Daphnia pulex* and *Daphnia hyalina* to small-sized *Bosmina coregoni* (Kattel *et al.* 2015). Likewise, chironomid remains enable the dynamic history of lake benthic communities to be reconstructed (bureau *et al.* 1995). In addition, recent developments in DNA extraction methods and analysis from sediments can reveal the nature of catchment change and predict the colonisation timing of fish taxa (Olajos *et al.* 2018). DNA analysis can even differentiate between native or exotic species (Stager *et al.* 2015).

Pollen grains and spores are morphologically distinct and are widely used in paleoenvironmental studies. The study of pollen grains and spores is known as palynology (Jones and Jones 2001), which has many applications besides paleolimnology, such as in archaeology and forensic science. Plants have ability to produce large numbers of pollen grains, that are well preserved in sediments. Although most pollen grains and spores are from terrestrial vegetation aquatic macrophytes also produce pollen that are morphologically different. Pollen grains can be identified to the genus level, but sometimes even to the species level owing to their distinct morphological structures. Hence, the composition of past forests and other vegetation provides important information on past climate and soil development, as well as the succession of different plant species. Pollen grains have unique features that assist in dating sediment cores. For example, in north-eastern North America, the arrival of European settlers, and their related activities of clearing forests and initiating European style agriculture, resulted in the increase in pollen grains from ragweed (*Ambrosia*). In addition, plant macrofossils (Birks 2002) and algal pigments (Leavitt and Hodgson 2001) have also been extensively used in paleolimnological reconstructions.

Diatoms are abundant in aquatic systems and are powerful bioindicators (Gell *et al.* 2018b). They belong to a group of microscopic unicellular algae that form part of the class Bacillariophyceae (Smol and Stoermer 2010). The prominent feature of diatoms is their siliceous cell walls that allow diatoms to be preserved, being highly resistant to changing conditions. Diatoms are composed of two valves making up a frustule, which range in size from ~2-200µm (Smol and Stoermer 2010) and are present in almost all aquatic habitats, and, due to their successful preservation, they are often abundant in ocean, lake, and wetland sediments (Battarbee *et al.* 2001; Smol and Stoermer 2010). More than 10,000 diatom species exist out of which a large proportion are ecologically sensitive, making them a particularly valuable paleoenvironmental proxy (Battarbee *et al.* 2001; Smol and Stoermer 2010). The ecological sensitivity of diatoms is due to their rapid reproduction and short lifespans (roughly a few days to a week). This feature allows diatom communities to respond rapidly to environmental stresses and change (Stevenson *et al.* 2010). Identification of diatoms is based on their morphology such that diatoms are broadly classified into centric and pennate forms (Battarbee *et al.* 2001). Thereafter, specific identifications are made depending upon the more specific shape, length, breadth, frequency, and pattern of striations, and presence of a raphe (Battarbee *et al.* 2002). Therefore, a precise environmental interpretation can be achieved from this high-resolution identification as species are segregated based on their different environmental preferences and tolerances (Reid and Ogden 2009; Liu *et al.* 2012b). Diatoms respond to

a large range of environmental variables, including temperature, nutrient concentrations, pollution levels, pH, and herbivory, with each species having a specifically defined set of thresholds for many of these attributes (Battarbee *et al.* 2001; Stevenson *et al.* 2010). The tolerance of each diatom species to different environmental variables can be established from training sets. Diatoms have been used to infer important indicators of water quality conditions such as pH, salinity (Gell 1997), acidity (Battarbee and Charles 1987; Tibby *et al.* 2003), and nutrients (Hall and Smol 2010). Diatom communities, and their change through time, can thus be used to determine local climate and environmentally-driven changes within a waterbody (Dong *et al.* 2012b).

#### **2.6.1.4 Radiometric techniques to determine the chronology of sediment cores**

A critical factor in interpreting lake sediment histories is to establish an accurate depth-time profile (i.e., geochronology) for the sediment core. Otherwise, it would not be possible to reconstruct environmental changes in a proper temporal perspective. There are various radiometric approaches available that can be selected based on the site location, sediment type, and the approximate age of the sediments themselves. Radiometric techniques that are most widely used to date lake sediments are radiocarbon-14 ( $^{14}\text{C}$ ) (Xu 2015) and lead-210 ( $^{210}\text{Pb}$ ) dating (Appleby *et al.* 1988; Du and Walling 2012). For older sediments,  $^{14}\text{C}$  dating is most commonly used, as the half-life of  $^{14}\text{C}$  is  $5730 \pm 40$  years. It is useful for dating sediments on millennial time scales up to approximately 45000 years of age. The  $^{210}\text{Pb}$  dating method is appropriate to understand recent histories of lakes. The half-life of  $^{210}\text{Pb}$  is about 22.3 years, and so it is ideal for dating sediments over the last century or so. Besides, to further refine the chronology of cores, other isotopes, such as cesium-137 ( $^{137}\text{Cs}$ ) can be used. This isotope is a by-product of nuclear explosions that became more common in the mid-1950s leading to a rise in  $^{137}\text{Cs}$  that can often be seen in sediment profiles at this time (He and Walling 1996; Du and Walling 2012).

#### **2.6.1.5 Applications of paleolimnology in freshwater ecosystems**

Paleolimnological approaches with the use of appropriate paleo-indicators, have been extensively applied to detect the impacts of humans on aquatic systems. For example; in the 19th century, the release of urban wastes into receiving water caused a considerable increase in hypoxic conditions in many lakes worldwide (Jenny *et al.* 2016). Also, nutrient enrichment in many lakes (e.g., in Europe, China, Central America and Australia) triggered eutrophication and caused regime shifts (Wang *et al.* 2012; Dubois *et al.* 2018). Lake acidification from acid rain impacting its water quality (Battarbee and Charles 1987), and

the impacts of heavy metals on lake ecology (Sayer *et al.* 2006) were documented using paleolimnological approaches. It is evident from these long-term records that substantial change to freshwater ecosystems occurred in response to natural and anthropogenic pressures which are well captured using paleoecological advancements well before detection by contemporary ecological studies.

This approach can be used to establish management targets and facilitates the scientific understanding of the complex dependency among people, climate and the environment, scientifically (Battarbee and Bennion 2011; Gell *et al.* 2013). It is an increasingly greater challenge to manage natural systems under a changing environmental condition where catchment histories are unknown (Gell *et al.* 2013). Knowledge without extensive historical information is likely to lead to misinterpretation of the drivers of change, the prospects of recovery, and the likely future path (Finlayson *et al.* 2015; Gell *et al.* 2016). Here, paleo-records provide evidence of past environments of a particular region over a long period of time, thereby providing knowledge of the nature of the response to environmental change. Paleolimnology assessments are used to quantify human-influenced changes in freshwater ecosystems (Battarbee and Bennion 2011) and for defining reference conditions and restoration targets (Bennion *et al.* 2011). The total P was inferred from fossil diatom assemblages in nine lakes across Europe in a project that used modern analog matching methods to identify contemporary reference sites (Bennion and Simpson 2011). These European showed a range of slight to major changes in vegetation over time. Foster *et al.* (2011) used paleolimnological methods to estimate pre-disturbance deposition rates and sediment yield to quantify maximum feasible sediment reduction in rivers of England and Wales. Where paleoecological approaches can reveal these changes, the integration of modern and paleoecological approaches can place modern data in the context of this evidence for change.

### **2.6.2 Paleolimnological assessments to understand floodplain lakes**

Many paleolimnological studies have focused on isolated lakes as, in these settings, many influences are controlled allowing for clearer inferences on attributing documented change. Lakes in fluvial settings are influenced by atmospheric input, inputs from the river as well as internal biological activities and chemical processes. Understanding pollution sources, eutrophication, turbidity, and other water quality issues are important concerns in fluvial and wetland ecosystems (Bennion *et al.* 2014). Worldwide, increasing deterioration of rivers and their impact on floodplain ecosystems is a reason for concern. It is believed that floodplain lakes play a key role in the exchange of nutrients, pollutants and other materials between the river and the floodplain. Whilst floodplain lakes represent

more complex settings for paleoecological studies, sediment records can be used to provide evidence of variations in water quality, changing sedimentation rates and aquatic or regional vegetation communities that are unavailable by other means.

Paleoecological approaches have been extensively used to report on the causes behind wetland changes such as variations in the water levels and quality, floodplain connectivity, sedimentation rates, and aquatic vegetation. Water level fluctuations add a dynamic aspect to the species/depth relationship. These fluctuations result in shifting patterns of wetland vegetation types. The history of the watering regime of a floodplain wetland can be inferred by reference to indicators that respond to water depth (Casanova 2011; Campbell *et al.* 2018) to understand water level fluctuations. For example; deeper water conditions would have a high proportion of regional pollen relative to local (Walker 2000). However, organic-rich sediments and local aquatic flora and fauna would be seen in shallow water conditions.

River connectivity and inundation periods have been inferred from diatom analysis in floodplain lakes of Australia. For example, upon floodplain inundation, river diatom plankton (*Aulacoseira* spp.) may enter floodplain lakes (Gell *et al.* 2002) while littoral taxa dominate floodplain lakes when not connected with a river. This is clear evidence for the level of river connectivity and inundation. In another study on Oxbow lakes of Lower Mississippi Valley, during 1968-2008, ecological evolution was observed in response to hydrological variability and anthropogenic changes (Bhattacharya *et al.* 2016). Diatom assemblages shifted from predominantly benthic (*Gomphonema parvulum* and *Meridion circulare*) to primarily planktonic assemblages (*Aulacoseira granulata* and *Cyclotella meneghiniana*) and revealed higher hydrological connectivity.

Changes in the proportion of diatom species in sediment cores have indicated an important change in the water quality of floodplain lakes. Diatom assemblages in Taibai Lake in China have undergone a marked temporal change over the past ~200 years. The lake was dominated successively by *Aulacoseira granulata* (planktonic), *Gyrosigma acuminatum* (periphytic), and facultatively planktonic *Nitzschia palea* (Liu *et al.* 2012a). Since the early ~1950s, epiphytic diatom taxa declined quickly and disappeared after ~1970 to be replaced by planktonic species tolerant of high turbidity and high nutrient concentrations (Yang *et al.* 2008). This shows that planktonic species thrive in turbid or light limiting conditions.

Furthermore, the impacts of humans on sediment influx in floodplain wetlands can be understood by sediment dating. Dating provides long records of sedimentation rates to distinguish between natural changes and human-induced changes. The history of

sedimentation concerning land use changes has been studied in three lakes of Ishikari Wetland in Northern Japan (Ahn 2018). It was reported that the lakes experienced higher sedimentation rates after the 1940s due to enhanced sediment influx from agricultural activities. Similarly, Xu *et al.* (2017) used paleolimnological records to estimate pre-disturbance conditions in 14 typical lakes of Yangtze River Basin of China and a sediment rate of 0.08 g/cm<sup>2</sup>/yr would be a feasible sediment reduction target. In southeast Australia, the past transitions in the ecological regime of Kings Billabong were analysed using multi-proxy approaches (Kattel *et al.* 2017) to better understand the nature of change in the Billabong.

In the light of the above paleorecords it can be said that paleoecological approaches can be better used for future management of complex floodplain lake settings. It was identified that conditions of lake ecosystems have been changed due to anthropogenic pressures. The nature and magnitude of these impacts have also been intensified. However, due to the lack of continuous monitoring of floodplain lakes, it is difficult to understand their environmental changes and ecological development over time (Hausmann *et al.* 2011). Finally, this study seeks to use paleoecological analysis to better understand historical floodplain changes.

### **2.6.3 The Ramsar Convention on wetlands and their paleoecological records**

The Convention on Wetlands of International Importance, known as the Ramsar Convention, came into force in 1975 when it was adopted as an international mechanism (Matthews 1993; Finlayson *et al.* 2011). This Convention is responsible for the local, regional, and national actions and international cooperation and thereby promotes the conservation and wise use of all wetlands. The term “wise use of wetlands” was introduced to maintain the ecological character of wetlands by implementing ecosystem approaches, within the context of sustainable development (Finlayson *et al.* 2011). According to the Ramsar Convention the ecological character is defined as the biological, physical, and chemical components of the wetland ecosystem. These components contribute in maintaining the wetland and its products, functions, and attributes.

Today, the Ramsar Convention is supported by over 170 signatory nations (Matthews 1993) that have listed over 2400 wetlands globally. All signatory nations consent to the wise use of all their wetlands, but there is particular recognition of duty to those listed. The Ramsar Convention requires signatory nations to identify, at the time of listing, the natural ecological character of the wetland as part of the Ramsar Information Sheet. The condition described is typical that at, or shortly before, the site is nominated, although the

Convention does not preclude the nation from changing the description if new evidence becomes available. Aware of the variability wetlands may experience, the Convention also encourages nations to identify the Limits of Acceptable Change (LAC) within which management is to maintain the ecological condition. A simplified concept of limits of acceptable change is presented in Figure 2.10 as a range within which the ecological condition of a wetland may vary under natural conditions.

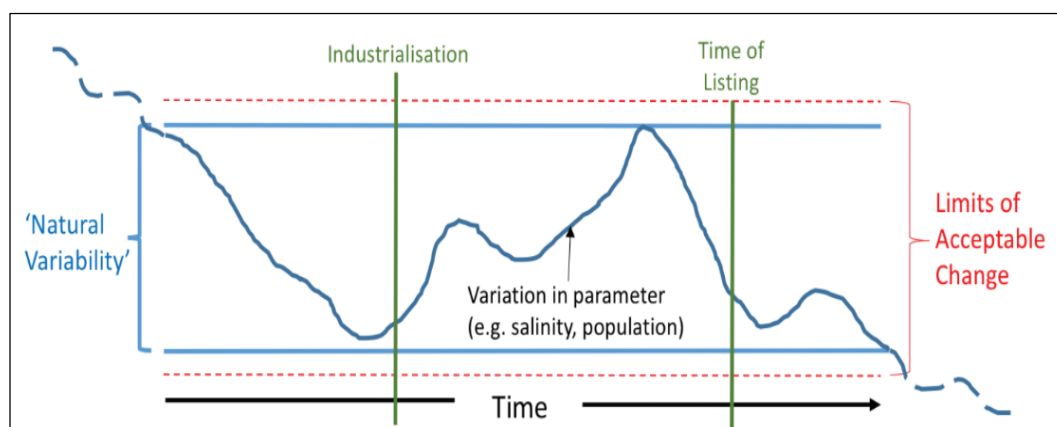


Figure 2.10 Comparison of natural variability and limits of acceptable change. The context for limits may change through time due to recent extent of natural variability over time (Reprinted from: Gell *et al.* 2018b)

The selection of internationally recognised wetlands is based on nine selection criteria that relate to their ecology, botany, zoology, limnology and/or hydrology, and cultural significance. These criteria are used to characterise a wetland under the Ramsar Convention (Table 2.3). While the retention of the listing criteria is most critical, the exceedance of LAC would suggest that the condition of the wetland has parted from its Ecological Character Description (ECD). In such cases, the nation is required to notify the Convention and rectify the degradation. In the absence of restoration, the nation is required to dedicate a substitute, equivalent wetland, although this has never been requested (Shine and De Klemm 1999).

The existing paleoecological records on Ramsar sites and other wetlands provide information that, in response to variability, the ecological character of a Ramsar site has changed after its international recognition. For example, the Florida Everglades site has been impacted by pollutants and hydrological change that has caused adverse change in the ecological character of the site (Riedinger-Whitmore 2016). In another example, the long sediment records from the Peace-Athabasca Delta site, the lakes show variations in lakes in response to infrastructure development which preceded listing. Moreover, the lakes experienced reduced flooding in response to a drying climate (Wolfe *et al.* 2008). Identification of this type of change can create conflict when considering the natural

ecological character, limits of acceptable change and declining condition of any specific wetland. Such information from paleoecological records is beneficial for management to protect the natural value of wetlands. For example, in the mid-1800s, a relatively high nutrient reference condition (TP of 50 µg/L) was suggested as an appropriate target for lake management for the shallow lakes in the Lower Yangtze River Basin (Dong *et al.* 2016). However, since the 1950s, the lakes have undergone considerable change with increased levels of nutrients to well above reference conditions. As the reference condition was known, the impact of anthropogenic pressure can be estimated for wise management practices.

**Table 2.3 Ramsar criteria to nominate a site (Reprinted from: Hale and Butcher 2011)**

Number	Basis	Description
<b>Group A. Sites containing representative, rare or unique wetland types</b>		
<b>Criterion 1</b>		A wetland should be considered internationally important if it contains a representative, rare, or unique example of a natural or near-natural wetland type found within the appropriate biogeographic region.
<b>Group B. Sites of international importance for conserving biological diversity</b>		
<b>Criterion 2</b>	Species and ecological communities	A wetland should be considered internationally important if it supports vulnerable, endangered, or critically endangered species or threatened ecological communities.
<b>Criterion 3</b>	Species and ecological communities	A wetland should be considered internationally important if it supports populations of plant and/or animal species important for maintaining the biological diversity of a particular biogeographic region.
<b>Criterion 4</b>	Species and ecological communities	A wetland should be considered internationally important if it supports plant and/or animal species at a critical stage in their life cycles, or provides refuge during adverse conditions.
<b>Criterion 5</b>	Waterbirds	A wetland should be considered internationally important if it regularly supports 20 000 or more waterbirds.
<b>Criterion 6</b>	Waterbirds	A wetland should be considered internationally important if it regularly supports one percent of the individuals in a population of one species or subspecies of waterbird.
<b>Criterion 7</b>	Fish	A wetland should be considered internationally important if it supports a significant proportion of indigenous fish subspecies, species or families, life-history stages, species interactions and/or populations that are representative of wetland benefits and/or values and thereby contributes to global biological diversity.
<b>Criterion 8</b>	Fish	A wetland should be considered internationally important if it is an important source of food for fishes, spawning ground, nursery and/or migration path on which fish stocks, either within the wetland or elsewhere, depend.
<b>Criterion 9</b>	Other taxa	A wetland should be considered internationally important if it regularly supports one percent of the individuals in a population of one species or subspecies of wetland-dependent non-avian animal species.



Given ongoing change in wetlands, whether through climate variability and geomorphic evolution, or wetland ontogeny, the Ramsar sites will likely vary within the LAC, or even exceed them. Paleoecological records almost inevitably reveal past conditions at variance from those at the time of listing and so challenge both ECDs and LACs (Gell *et al.* 2018b). As Ramsar Convention encourages restoration, the identification of past, more diverse states would justify a change in the LAC and encourage efforts to recover these conditions. Paleoecology therefore, provides better options to understand wetland variability and management that accommodates change, rather than seeking stasis.

Thereby, better understanding of wetland characteristics and the potential impacts of river change on wetland condition enables policymakers and wetland users to develop and implement policies, legislation and management practices that lead to the appropriate use of wetlands while maintaining their biodiversity (Bunn *et al.* 1997). Such interventions require better knowledge of the processes that influence the physical, chemical and biological characteristics of the wetlands and ongoing changes due to land use practices in the vicinity of sensitive wetlands and the influence of impacts transported to the wetland via the river (Piyankarage *et al.* 2004). The wise use of wetlands, and adherence to national and international conventions, requires consideration of the broader catchment-scale drivers of change, as well as those proximal to the site itself. Globally there are few studies on the ecological character of intermittent wetland systems at Ramsar sites. Along the Murray River, Ramsar sites include the Barmah Forest, Hattah Lakes, the Riverland, Banrock Station and the Coorong and Lower Lakes.

## **2.7 Murray Darling Basin wetlands**

Among the large rivers of the world, the Murray Darling Basin (MDB) is Australia's largest river system that hosted a large population of indigenous Australians prior to European invasion (Pardoe 1988). Early settlers used the catchment for sheep and cattle grazing. For the Europeans, the rivers were sources of transportation of agricultural produce into the market. During the 1850s, gold digging led to the clearance of large areas of vegetation and was responsible alone for an estimated transport of 400 M m<sup>3</sup> sediment load into the MDB (Davies *et al.* 2018). In late 19th century, the irrigation technologies were developed to increase food production to fulfil the demands of a growing population of Australians. However, climate variability significantly impacted the agricultural production and transportation, so river regulation was introduced by construction of weirs and dams in the Murray Darling Basin.

The MDB has experienced many drought and flood events in the past that are associated with the El Niño Southern Oscillation, as well as the Indian Ocean Dipole. Based on global

climatic assessments, recent drought and flood events across the MDB provide an insight into the heightened variability that this basin is experiencing (Mills *et al.* 2013). In recent decades, in the Murray River, a rise of 1°C in temperature has been measured and this is predicted to cause approximately 15% reduction in annual flows (BOM 2019). The southern basin is expected to suffer from reduced spring and autumn rainfall while the northern catchments will experience dry periods (BOM 2019) but extreme floods when tropical cells penetrate higher latitudes. The Millennium drought brought extreme conditions from 1997-2009 such that rivers within the MDB received 40 % less inflow than was the long-term average (van Dijk *et al.* 2013). This drought event was interrupted by an extreme wet phase through 2010-11 and had severe implications on the MDB whereby systems such as the Loddon River floodplain was inundated for several months. More recently an extended drought through NSW contributed to the drying of the lower Darling River leading to mass fish kills (Vertessy *et al.* 2019). This too was concluded with a high rainfall event which brought the Darling River water through to the Murray River. While river flows and wetland inundation in this catchment is substantially influenced by river regulation and overexploitation, shifts in climate will tend to exacerbate the extremes being felt in recent decades.

Regulation of the MDB over the past 100 years, has resulted in considerable implications for ecosystem structure and function. The construction of weirs and dams typically interrupt longitudinal connectivity and forms water reservoirs where large volumes of water are lost due to evaporation. For example, downstream flows of the Murray River have been reduced due to the construction of dams (Mallen-Cooper and Zampatti 2018). Dams interrupt sediment movement downstream, restricting the exchange of energy between the river and its riparian zone during flood events (Kondolf *et al.* 2014). Dam building has also been known to obstruct upstream migration of biota and altered the thermal environment, nutrient dynamics and sediment loading (Schmutz and Moog 2018). Furthermore, the high level of water abstraction in the MDB since the 1960s has accentuated changes in flow and water volume as principal drivers of change (Jones *et al.* 2002; Grafton *et al.* 2014). These changes are particularly related to direct catchment disturbances such as agricultural activities and have been exacerbated by an extended drought regime causative of unprecedented low levels in rivers and wetlands. Hence, irrigation, climate change and regulation are considered as major drivers of change in the MDB (Mackay and Eastburn 1990; Grafton *et al.* 2014).

The MDB has been under the influence of anthropogenic stressors for over more than 150 years (Davies *et al.* 2018). Monitoring of the water quality of the Murray River commenced in 1951 and little or no pre-regulation water quality data, that could provide reasonable

understanding of the principal phase of water resource development (Gell and Reid 2016), exists. Therefore, it is hard to understand the baseline ecological condition and historical variability patterns of MDB. Whilst, ecological evidence has attested to the degraded state of the system and the cause of the problem is contested, it is argued that either pollution or water volume could be driving changes in the MDB (Gell and Reid 2014). The historical evidence of human impacts on the aquatic systems of the MDB, and the relatively short duration of field monitoring, reveal that there is likely much change that has occurred that is obscured by time (Gell and Reid 2014). Where paleoecological approaches can reveal these changes, the integration of modern and paleo approaches can place modern data in the context of this evidence for change. The benefit of this longer-term perspective includes the capacity to understand the influence of low frequency cycles, to identify past critical transitions in state that may preclude restoration, to understand natural baseline conditions (Finlayson *et al.* 2016), and to better understand the trajectory of change (Gell *et al.* 2018b).

Numerous studies have investigated paleo-environmental change using biological proxies in inland freshwater wetlands of south-eastern Australia (Thoms *et al.* 1999; Gell *et al.* 2005a; Kattel *et al.* 2015). However, paleoecological research of this type has rarely been carried out in arid and semiarid environments to the extent that it has in more humid regions. This is because under the influence of frequent drying cycles, the preservation of biological indicators in sediment deposits is often very poor (Gell *et al.* 2009). In the MDB, many studies are conducted on wetlands in the Murray-Murrumbidgee region, in the south. Rivers and floodplain wetlands of this region tend to receive more frequent flooding than those in the typically drier Barwon-Darling region, in the north (Brandis *et al.* 2011). Most of this research has used diatoms as indicators of salinity, pH and water level changes (Gell *et al.* 2002; Tibby 2003; Tibby 2004; Gell *et al.* 2005a; Gell *et al.* 2009; Reid and Ogden 2009), or diatoms and invertebrates to infer vegetation (e.g., macrophyte) and land-use changes (Ogden 2000; Reid *et al.* 2007).

The paleolimnological records from the southern MDB are well documented and constitute one of the most intensive records of change in a network of floodplain lakes across the world (Gell and Reid 2014; 2016; Gell 2020). Over 60 wetland sites in the southern MDB have been examined using fossil diatom records supported by various dating techniques (Gell and Reid 2014; 2016; Gell 2020). Among the other pressures, increased sedimentation was observed in 55% of wetlands, raising the concern of complete sedimentation in the shallower systems (Gell *et al.* 2013). The record from each wetland has shown the impact of European settlement, with 80% impacted by sediment, 48% by nutrients, and 34% by salinisation (Gell and Reid 2014). It is evident from the available

literature that elevated pollution loads began mostly since regulation in the 1930s but also from as early as the 1880s (Gell and Reid 2016).

Furthermore, the impacts of river regulation and climate change have been observed on the vegetation of MDB wetlands. There is evidence for the widespread replacement of submerged aquatic plants (Reid *et al.* 2007) by floating *Azolla* and emergent *Typha* (Gell 2012) and the associated shift in diatom algae and cladoceran microfauna from littoral taxa to pelagic (Kattel *et al.* 2015). The coincidence of documented changes and the spatial variability of sediments across wetlands (Grundell *et al.* 2012) implicate the rivers as sources of sediments and nutrients. Although the watering of wetlands can stimulate breeding and recruitment of critical fauna (Kingsford and Auld 2005), inundation can impact water quality and defer the recovery of ephemeral species afforded by periods of drying (Jensen 2002).

In this respect, the study of Murray River wetlands and their ecosystem functions is of particular interest. Gell and Reid (2014; 2016), updated the synthesis of Gell *et al.* (2009) paleo-records across the southern Murray Darling Basin yet few studies have been undertaken in the mid-Murray system. Most of the studies include physical, chemical and biological indicators such as diatoms, moisture and organic content, sediment texture, magnetic susceptibility, pollen and pigments. However, several available records lack robust multi-proxy assessments as used in this study. Therefore, this research aims to use multi-proxy paleoecological assessments to characterise wetland changes that are located in mid-Murray region.

## **2.8 Management**

Freshwater management policies are common around the globe, focusing on the protection, mitigation of threats and rehabilitation of wetlands but varying in their effectiveness. For example, the Clean Water Act (United States), the Water Framework Directive (European Directive 2003), and the Water Act (Australia) are introduced to protect these ecosystems from a variety of human activities. It is well known that although such protections and management policies have improved resource conditions, freshwater ecosystems continue to face impacts from numerous ongoing and emerging threats. The most common threats to wetlands include water resource development via flow regulation, habitat loss and degradation, pollution, invasive species, and climate change. Management and conservation options for these threats have been summarised in Table 2.4.

**Table 2.4 Conservation and management options of wetland ecosystems in relation to major threat  
(Adapted from: Kingsford *et al.* 2016)**

Threats	Management and conservation options
<b>Water resource development</b>	<ul style="list-style-type: none"> <li>• Suitable legislation needed to assess environmental impacts, including ecosystem services and long-term costs for proposal and ongoing renewal of dams</li> <li>• Environmental flow protection</li> </ul>
<b>Habitat loss and degradation</b>	<ul style="list-style-type: none"> <li>• Natural resource management legislation and assessment to protect flow and ecological processes</li> <li>• Protection of flow regimes of free-flowing rivers or wetlands</li> </ul>
<b>Pollution (<i>Nutrient, chemical, salinity, pharmaceutical etc.</i>)</b>	<ul style="list-style-type: none"> <li>• Regulations to ensure that water quality is protected</li> <li>• Management of environmental flows to reduce impacts of pollutants</li> <li>• Reduction of nutrients run-off from adjacent terrestrial systems with buffers</li> <li>• Controlled disposal of pharmaceuticals into rivers and wetlands</li> </ul>
<b>Sedimentation and erosion</b>	<ul style="list-style-type: none"> <li>• Practicing sustainable land use methods Construction of buffer zones to slow down the run-off to protect the wetland from sediments and other contaminants</li> <li>• Use of flocculants, increasing flows to reduce water residence time and introducing organisms that consume nutrient induced algal blooms</li> </ul>
<b>Invasive species</b>	<ul style="list-style-type: none"> <li>• Introduce programs to reduce distribution and species richness</li> <li>• Introduce programs to mitigate the transfer of plant species between wetlands</li> <li>• Provide education programs on the impacts of invasive species</li> </ul>
<b>Climate change</b>	<ul style="list-style-type: none"> <li>• Increased environmental flow</li> <li>• Increase area of wetland for conservation management to allow for inclusion of more refugia</li> <li>• Building of structures (e.g., walls) may be required in some regions to protect wetlands</li> </ul>

It is well known that the condition of floodplain wetlands is impacted by local effects, but is also strongly driven by the nature of its river. The restoration of the ‘natural flow regime’ is seen as the key to the restoration of the river ecosystems (Sparks 1995; Poff *et al.* 1997; Robinson and Uehlinger 2003). In certain situations, headwater dams have been removed to restore a more natural flow regime in the river (Hart and Poff 2002). However, this action is economically, socially, and politically impossible in most rivers. A partial return to the natural regime with managed dam releases has therefore been proposed to imitate aspects of the pre-regulation flow regime (Sparks 1995; Poff *et al.* 1997; Robinson and Uehlinger 2003; Newson and Large 2006). The optimal use of dam releases for environmental recovery requires a detailed understanding of the natural function of the

river system, including an understanding of the natural flow regime, floodplain inundation patterns, and the relationship between fluvial and floodplain ecosystem processes (Tockner *et al.* 2000; Thoms 2003).

### **2.8.1 Point and not-point pollution control**

It is more straightforward to control point source pollution in comparison to diffuse pollution. Generally, point source pollution is controlled by government regulations and guidelines (Kingsford *et al.* 2016). Whilst diffuse sources of pollution require education programs or buffers to reduce pollutant run-off from adjacent lands. The application of advanced municipal wastewater treatment and source reduction can potentially mitigate emerging contaminants. To some extent, pharmaceutical pollution can be controlled by reducing the use of emerging contaminants such as antibiotics and endocrine disruptors. However, reductions in human pharmaceutical usage are unlikely, so better protection and conservation of biodiversity could occur through effective management strategies such as using improved disposal methods of unused medications and advanced wastewater treatment facilities.

Sedimentation and erosion can be controlled by practising sustainable land-use methods. Soil conservation plays a significant role in controlling erosion and sedimentation influx. A buffer of natural vegetation around wetlands can be established and maintained. These buffer zones aid in slowing down the run-off and act as filters to protect the wetland from sediments and other contaminants. Further, enhanced concentrations of nutrients cause eutrophication in water systems with various ecological implications. To control these nutrients, preventative measures may include reducing or removing external nutrient loads (Paerl *et al.* 2011) or treating lake sediments with chemicals to suppress internal nutrient recycling (Molot *et al.* 2014). Potential mitigation measures include chemical controls (e.g., flocculants), physical controls (e.g., increasing flows to reduce water residence time), and biological controls (e.g., introducing organisms that consume nutrient induced algal blooms) (Rastogi *et al.* 2015).

Invasive species control programs are focused on reducing the distribution and abundance of those species that degrade ecosystems. It is necessary to educate communities about the dangers of invasive species. The potential for control measures depends on the application of techniques to detect non-native species. Invasive species of lakes, rivers, and wetlands can be identified from the fragments of deoxyribose nucleic acid (DNA) which is present in water in the form of faeces, cells, secretions, etc., and is transported through drainage channels (Reid *et al.* 2019). These fragments of DNA can be isolated from organic matter in water samples, sequenced and decoded and assigned

to known species using DNA barcoding (Elbrecht and Leese 2017). Thus, they offer an integrated way to assess the species richness in freshwater ecosystems (Deiner *et al.* 2016) in a way that aids in application of chemicals under integrated management plans to control biological invasion.

Furthermore, to manage the impacts of climate change, wetland conservation has to deal with challenges such as rising sea levels and altered flooding regimes which are often worsened by river regulation. One measure may be to build structures around wetlands to protect them from rising water levels. Theoretically, it is rather difficult to build walls owing to the related construction expenses. Therefore, impacts of climate change on wetlands needs to be identified (Dudgeon *et al.* 2006; Reid *et al.* 2019) by the improved monitoring of their ecological status. Monitoring of ecological responses to environmental change is critical in assessing the performance of a management action. Continuous monitoring over long periods is required to understand the dynamic nature of riverine environments so that modifications to management strategy can be made where required.

Based on current rates of extinction, habitat degradation, and emerging challenges, it is evident that freshwater ecosystems have already faced pressures greater than any other ecosystem. In the future, these threats will intensify as the exploitation of freshwater ecosystems grows to satisfy human demand. At the global, local, and regional level, various conservation scientists, ecologists, and environmentalists are working in freshwater ecosystems to provide evidence for the ecosystem changes to firstly arrest the documented decline, and to protect and restore the world's lakes, reservoirs, rivers, streams, and wetlands. These potential actions indicate that the world's freshwater ecosystems and their biota can be protected but only if the measures are taken at local, national, and global scales in the face of increasing pressure.

## **2.9 Summary: research gaps and avenues for future research**

The issues addressed in this chapter have provided a conceptual basis for addressing various factors affecting wetland ecosystems. The literature review revealed that although wetlands remain invaluable elements of the environment, they are subjected to diverse forms of degradation in response to climatic and non-climatic factors. Although literature provides some evidence of declining wetland conditions, there remains gaps in terms of the environmental changes and ecological development over time, owing to the general lack of continuous monitoring of floodplain lakes. Hence, there is need for further investigation to add new insights into and conceptualise the wetland changes using robust high resolution paleoecological assessments. As discussed in the Chapter 1, the aim of this research is to explain and understand the drivers of wetland changes. This research

explores key issues in relation to the changed condition of Ramsar wetlands, drivers of these conditions, and impacts of these conditions on the water quality and ecology of wetlands. The multi-proxy approach is best suited to addressing these issues, as the study site, Gunbower Forest lacks historical monitoring data. The detailed description of study area and its characteristics are elaborated in the next chapter.



## **CHAPTER 3- THE STUDY AREA AND ITS CHARACTERISTICS**

The study area comprises the northern plains of Victoria through which several rivers flow and drain towards the Murray River. Within this region is situated in the Gunbower Forest, and its wetlands that lie on the island formed between the Murray River and its distributary Gunbower Creek. This chapter is divided into three sections that describe the Murray Darling Basin (MDB) (3.1), Gunbower Island (3.2) and the Summary of the chapter (3.3).

### **3.1 Murray Darling Basin**

The Murray-Darling Basin (MDB) has a catchment area of approximately 1.06 million square kilometres (Mackay and Eastburn 1990). The combined length of the two major rivers, the Murray and the Darling, is approximately 5500 km making it one of the world's longest river systems. It supports a population of over 2 million people and, through the provision of agricultural products, many more outside the catchment boundary (Craik 2005). The geographical orientation of the MDB means the climate varies from cool, humid conditions in the eastern uplands, dry temperate conditions in the southern Mallee, subtropical conditions in the northern areas and hot, arid and semi-arid country in the west (Craik 2005). The north of the Basin is occupied by semi-arid ephemeral river systems, while river systems are highly regulated in the south receiving waters from the Australian Alps, which are seasonally snow-covered (Figure 3.1). Therefore, the MDB is a complex system and provides water for economic, social and environmental purposes. The major commercial activity in the MDB is agriculture, which produces 40% of Australia's gross value production (Alexandra 2018) and 70% gross value of the nation's irrigated agricultural production (Australian Bureau of Statistics 2012).

The Murray-Darling Basin is very old however, the recent land formation occurred over the last 60 million years (Connell and Grafton 2011). The Basin is formed by tectonic activity associated with the Australia's rifting from Antarctica followed by its subsequent northward drift (Brown and Stephenson 1991). Proterozoic and Paleozoic rocks elevated to form an upwarp at the basin margins producing a 300,000 km<sup>2</sup> saucer-shaped intracratonic structure surrounded by low mountain ranges. While the Basin is huge, the sedimentary rocks are relatively thin (Connell and Grafton 2011), averaging 5,200 m in the north, east and south and attaining a maximum thickness of little more than 500 m in the west-central depocentre (Brown and Stephenson 1991). The surface of the western part of the basin is mainly occupied by the subdued and well-vegetated east-west-

trending Mallee dunefield. The east of the Basin is dominated by alluvial plains with occasional low sand dunes to the north and east of many of the larger paleochannels.



Figure 3.1 Location of Southern (Murray River) and Northern (Darling River) Basin in Australia, with contributing rivers to their flow (Source: MDBA, n.d.)

During the Cenozoic period, marine incursions from the southwest in the Mallee region occurred repeatedly, representing the culmination of prolonged sedimentation records (Brown and Stephenson 1991). This is followed by a retreat of the sea from the Mallee region in the Pliocene that was accompanied by the uplift of the Pinnaroo Block in the southwest. A natural dam was constructed on the Murray River that led to the formation of the freshwater Lake Bungunnia (33000 km<sup>2</sup>). On the Riverine Plain, the lower Shepparton Formation was deposited by large rivers that fed the lake. Then after 0.8 Ma, a permanent lake channel became established and Lake Bungunnia drained, rapidly fragmenting into several smaller basins of which Lake Tyrrell in Victoria is a remnant. Sometime after the demise of Lake Bungunnia, the Basin transitioned to more arid conditions. The dune fields became established in the Mallee by about 0.5 Ma (Zhisheng *et al.* 1986). However, in the upper Shepparton Formation, the fluvial environment

remained predominant and persisted on the Riverine Plain with buried soils indicating pauses in deposition (Butler 1958). A shift to more arid conditions was witnessed in the Late Quaternary period, which was different from the aridity of the Late Pliocene. Moreover, the landscape evolution of the Riverine Plain was affected by the occurrence of repeated global glacial-interglacial cycles of climatic change. The evidence of fossil lakes, dune fields and large channel systems clearly testifies to these impacts (Bowler 1986). The last post-glacial rise in sea level drove a reduction in stream gradient which saw a braided system deposit the Monomon formation, which, over the last ~ 7000 years, has been overlain by the Coonambidgel formation through which the present Murray River meanders.

### **3.1.1 Settlement history**

Indigenous Australians have inhabited Australia for more than 60,000 years (Clarkson *et al.* 2017) and the Basin itself for at least 45,000 years (Bowler *et al.* 2003). Prior to European settlement in the mid-1800s, the Murray-Darling Basin system was one of the most highly populated regions (Pardoe 1988). It is evident from the existence of shell middens and remains of aquatic origin that the rivers, lakes and wetlands of Murray-Darling Basin are associated with Aboriginal people who exploited the MDB's water resources and biota (Bowler *et al.* 2003; Humphries 2007). In addition, the sophisticated rock shoots and funnels of the Brewarrina Fisheries are further evidence of Aboriginal existence and influence in the Basin (Mathews 1903; Dargin 1976). There is archaeological evidence at Lake Mungo in New South Wales that Aboriginal people had settled there at least 40,000 years ago (Bowler *et al.* 2003). The excavations along the lower Murray revealed 8,000 years of habitation, as well as clear technological development (Humphries 2007). Scatters of clay balls used for fireplaces are evident on Gunbower Island, showing this site as important to indigenous populations through the Holocene.

Before the arrival of Europeans, whalers and sealers had visited the shores of southern Australia to harvest the large colonies of seals found on the offshore islands (Sinclair 2001). They introduced diseases such as smallpox which acted to decimate indigenous populations. In 1924, Hume and Hovell became the first European to see the Murray River. However, Charles Sturt first explored the Murray River and witnessed the effects of these diseases in 1829-36, as did later visitors to the river (Sinclair 2001). The next onslaught on indigenous populations came with British settlement in 1836 and thereafter. Aboriginal people (and the native fauna) were driven out, and many were placed in 'missions', where their traditional cultural systems and practices were eroded and replaced

by European provisions and laws. While there was some cooperation between explorers and settlers, and the indigenous societies, the many massacre sites attest to the violent nature of the contact period (Birna *et al.* 2013).

In the 1850s, Aborigines replaced European workers on many Murray stations because European workers showed little interest in navigating the Murray River due to the Victorian gold rush. During the gold years, the land was cleared for gold digging that encouraged widespread sheep grazing and caused mobilisation of the polluted soil of the gold rush period (Davies *et al.* 2018). By the end of 1850s, most of the river frontage land was occupied by graziers and loggers supplying timber to river steamers (Wachtel 1982). European settlement had a substantial negative effect on the local aboriginal communities. By the early 20<sup>th</sup> century, mostly all the Aboriginal inhabitants were either displaced or killed. Irrigation and river regulation commenced in the 19<sup>th</sup> century and 20<sup>th</sup> century respectively, and this enabled a substantial increase in water abstraction through the 20<sup>th</sup> century. During the period of early settlements, there were no wastewater treatment options, so, rivers were considered as highly degraded (Davis *et al.* 1902; Gell *et al.* 2019). Occupation of the MDB by the first Australians but more so their replacement by Europeans, had significant implications for the aquatic systems (Dubois *et al.* 2018).

### **3.1.2 Hydrology and climate**

The Murray Darling Basin lies in the vicinity of the region of greater atmospheric variability, the mid-latitude zone (Bayon *et al.* 2017) and is predicted to be greatly influenced by climate change (Pittock 2018). While there exists a diversity of climates across the Basin the southern Basin is subjected to the seasonal waxing and waning of the winter westerlies and the mid-latitude high-pressure belt. Typically, these bring winter rains and hot, dry summers, with the dry season occasionally punctuated by intense rain from cut-off low-pressure systems. The system is subjected to the large-scale variations in atmospheric circulations (e.g., IPO, ENSO) that bring droughts, and floods at regular intervals.

In the last few decades, various studies have applied global climate models to identify potential changes in the Basin's climate and estimate their implications on water availability (Austin *et al.* 2010; Smith and Chandler 2010; MDBA 2019). It was suggested (BOM 2019) that predicting uncertain events such as *El Niño* and *La Niña* events is challenging, and the capacity to predict future temperatures and rainfall patterns is limited. However, the predictions of changing temperature and rainfall patterns are consistent. Therefore, climate models characterise the key processes driving warming and rainfall trends (Bureau of Meteorology 2019). Based on climate change projections, in the

northern Basin a small increase in total annual rainfall can be anticipated (Ekström *et al.* 2015; Bureau of Meteorology 2019). However, in the southern Basin a long-term decline in the winter and spring rainfall is predicted to occur through the rest of this century (Hope *et al.* 2017, Bureau of Meteorology 2019).

The southern MDB is hydrologically variable with an erratic pattern of high and low flows (Finlayson *et al.* 2011; Gergis *et al.* 2012). The rainfall anomaly data for the MDB (Figure 3.2) illustrates periods of low rainfall (drought dominated regime) and high rainfall (flood dominated regime) events. Over the past century, and longer, substantial changes in the climate have been documented such as dry and wet periods at decadal and multi-decadal scales. In addition, there is evidence of a series of significant droughts (e.g., Federation drought (1898-1903), World War II drought (1939-45) and floods (e.g., the 1950s) that represent extremes at a centennial scale (Figure 3.3) (Verdon-Kidd *et al.* 2014). Large floods and long droughts are linked to the *El Niño/La Niña* Southern Oscillation and other climate influences that make the MDB among the regions with the greatest variability in the world (Pittock 2018). In the latter part of the 20<sup>th</sup> Century, the river flow regime was dominated by low inflows (3980 GL/yr), owing to intensive regulation and abstraction, in combination with a drought dominated phase (1997-2009). However, high inflows were recorded after the flood phase of 1946-1976 and 2010-11 because of the flooding events associated with an extreme, short, wet phase influenced by the Indian Ocean Dipole (MDBA 2012).

In addition to the expected long-term change, when short-term hydroclimatic variability is considered, such as the extreme 2002–2010 drought in the MDB, the impacts on wetland biota may be substantial, such as the mass fish kill on the Darling River (Vertessy *et al.* 2019) during the 2017-20 drought across NSW. Moreover, paleoclimatic evidence illustrates that wetlands across the MDB (Mills *et al.* 2013) are substantially influenced by catchment perturbations and river regulation which overwhelm the measurable impact of recent climate change and variability.

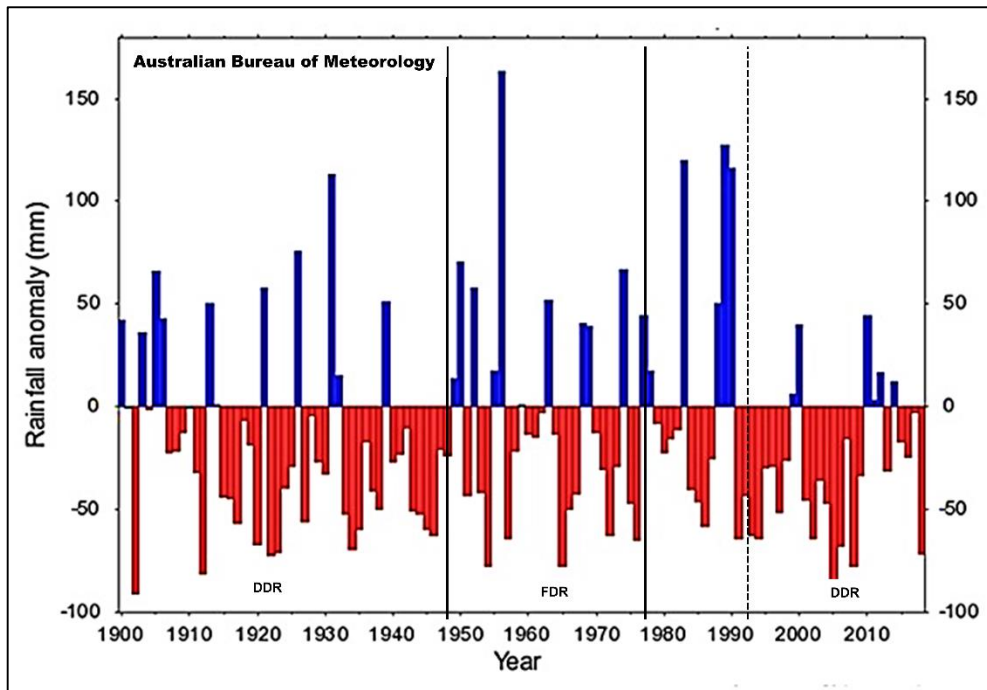


Figure 3.2 Rain Fall anomaly data for MDB showing Flood Dominated Regime (FDR) and Drought Dominated Regime (DDR) from year 1990-2019 (Source: Bureau of Meteorology 2019)

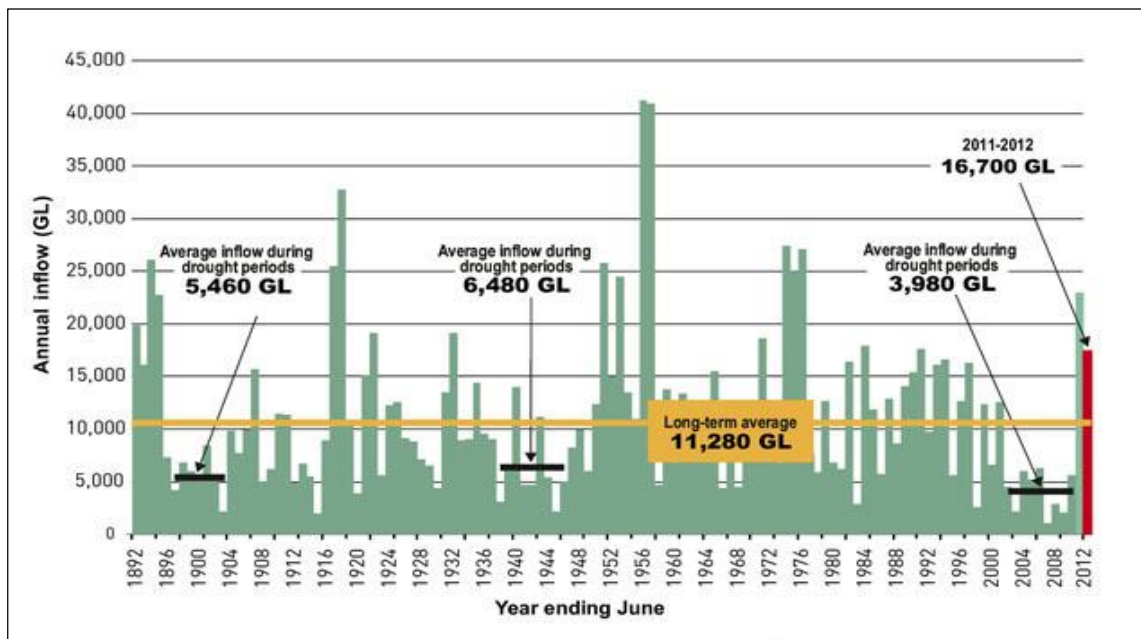


Figure 3.3 Murray River Annual Inflow (GL) during 1892-2012; long-term average flow is highlighted in yellow colour; drought flows in black colour and flows after 2010-2011 are shown in red colour (Source: MDBA 2012)

### **3.1.3 Onset of regulation (1920s) and its impacts on the river system**

Regulation of MDB waterways began with the construction of weirs on its tributaries in the late 19<sup>th</sup> century. The Goulburn Weir, among the earliest, was constructed in 1887 to support the irrigation industry. The main phase of regulation began from the 1920s with intensive development of regulators along the Murray River with the construction of 11 low-level weirs to provide gravity diversion for irrigation and water supply. Thereupon, construction of two large dams: Hume and Dartmouth, commenced in the headwaters of the Murray in 1936 and 1979 to create storages of 1540 GL and 4000 GL, respectively. Due to this diversion, the mean annual discharge of the Murray River to the sea declined by 61% from 12,233 GL to 4723 GL (Mackay and Eastburn 1990; Prosser 2011). Under the influence of river regulation, many wetlands have switched from clear water and macrophyte dominated state to phytoplanktonic turbid systems (Reid *et al.* 2007).

To summarise, numerous studies conducted on the Murray Darling Basin have reported on its poor ecological condition owing to the impacts brought about by European settlement (Gawne *et al.* 2011). These impacts include water diversions, a changed water regime, and hydrology (Mills *et al.* 2013). Cumulatively, these impacts have significantly altered biodiversity and ecological processes (Gawne *et al.* 2007, 2011). Furthermore, native fauna and flora have been greatly impacted along 40% of the total river length in the Basin and these are expected to further decline. It is reported that, prior to regulation, the river and wetlands were well connected, allowing the river biota to thrive (Young 2001).

In the past, climate change has greatly influenced the hydrology of inland Australian rivers (Finlayson *et al.* 2013; Mills *et al.* 2013). Today, the characteristics of the rivers are maintained by the current climate and the hydrology of their catchments, but rivers are also subjected to immense anthropogenic pressures such as water resource developments, river regulation and water extraction (Bunn *et al.* 1997; Davies *et al.* 2010). The ecological character of many wetlands across Australia has deteriorated in recent decades as a consequence of these pressures. Those of the MDB are anticipated to change further on account of the above-mentioned changes (Newall *et al.* 2016). As discussed in Chapter 2, there is lack of information about the wetlands of mid-Murray region. So, in this research the iconic site, Gunbower Forest has been chosen to better understand the change in the ecological character of its wetlands.

## **3.2 Gunbower Island**

Gunbower Island has an area of approx. 20,000 ha and is located between Torrumbarry in the south-east and Koondrook in the north-west (Figure 3.4). The land is occupied by a

state forest (9,343 ha), national park (8,892 ha) and Murray River reserve (1,666 ha) (Hale and Butcher 2011). Gunbower Island is located between the banks of the Murray River and Gunbower Creek, in north-central Victoria, Australia (Cooling *et al.* 2002; Mendham and Curtis 2015). Gunbower state forest provides a breeding habitat for endangered colonial waterbirds, fish, reptiles and other species such as frogs (Cooling *et al.* 2002).

The Island has a very shallow elevation gradient limiting the drainage of wetlands and slowing flow. So, subtle variations in topography create a diverse and rich floodplain wetland system. It comprises wetlands with water regimes varying from near permanent to temporary and has a strong ecological connection with Murray River catchment (Cooling *et al.* 2002). The forest's characteristically flat landscape is made up of small sandy rises and is dissected by a diversity of wetlands and waterways. Soil patterns show clay loams in areas of lower elevation are intersected with sandy loams on the higher rises. The region's climate is typically hot and dry in summer and cold and wet in winter (BOM 2020) that supports native vegetation of the Forest.

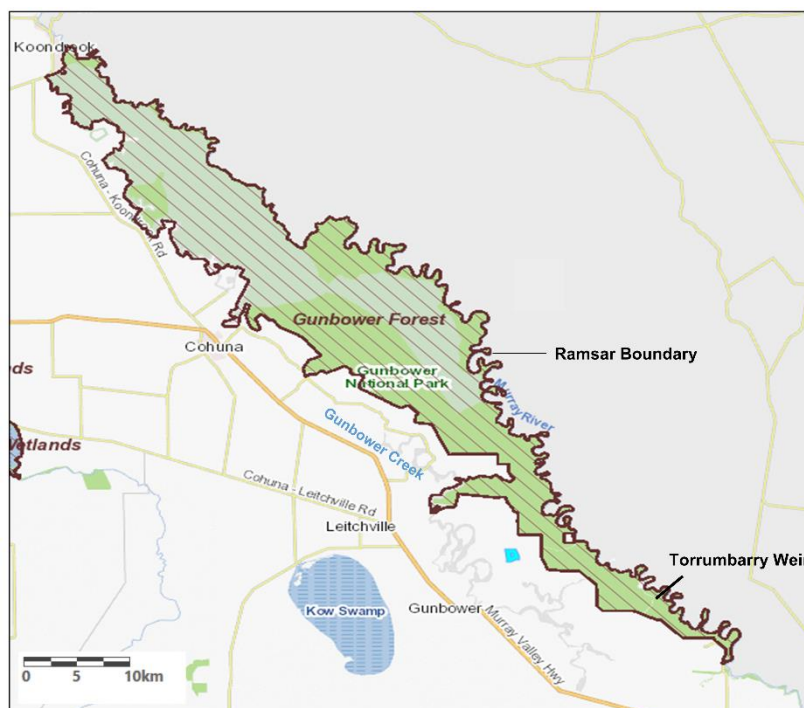


Figure 3.4 Location of Gunbower Island (Gunbower Forest and Gunbower National Park) between the banks of the Murray River and Gunbower creek. The Ramsar boundary of the study site is shown by the dark brown line around the Gunbower region.

Gunbower Forest is known as the second largest forest of red gum (*Eucalyptus camaldulensis*) in Victoria but the island also supports other types of vegetation such as Black Box woodland (*Eucalyptus largiflorens*) and Grey Box woodland (*Eucalyptus*



*microcarpa*). River Red Gum is the canopy dominant in all forests and in the wetter woodland communities. Community structure and understorey composition vary with flood regime, which in turn is a product of geomorphic setting. The driest portions of the floodplain support Black Box woodland which grades into a river red gum woodland at its wetter end, with a sparse, shrubby understorey and groundcover of grasses and herbs with increasing soil moisture (Northern Central CMA 2015). Better-watered locations support a taller river red gum forest with an understorey of moisture-loving grasses, herbs and sedges (Roberts and Marston 2000). The more channelised areas of the lagoons and the creek are generally dominated by species such as Hornwort, Yellow Bladderwort (*Utricularia australis*) and Water Milfoil (*Myriophyllum* spp.) as well as the small floating fern Pacific Azolla (*Azolla filiculoides*). In the slow flowing or still areas of the system, invasive species such as Pale-Yellow Water Lilly (*Nymphaea mexicana*) and Parrot's Feather (*Myriophyllum aquaticum*) are abundant (Ecological Associates 2015). These species are supported by the elevated summer water levels and nutrient levels from surrounding farmland.

### **3.2.1 Ramsar recognition of Gunbower forest**

The Gunbower Forest Ramsar Site is located along the Murray River, approximately 235 kilometres north-west of Melbourne (Figure 3.4) and covers an area of 20,218 ha. The Site includes public land from the River Murray Reserve, Spence Bridge Education Area, State Forest, and Gunbower National Park. Gunbower was found to satisfy four of the nine criteria used to identify wetlands of international importance (discussed in Chapter 2) and was listed as a Ramsar Site of International significance in 1982. Site details for this Ramsar wetland are provided in Table 3.1 (Hall and Butcher 2011). The listing criteria are as follows:

Criterion 1: The Gunbower Forest is unique in nature, being a significant part of the second largest river red gum forest in the Murray-Darling Basin (the largest being Barmah-Millewa Forest). It represents one of the best representatives of the wetland type Xf (freshwater tree-dominated wetlands) in the bioregion. In addition, the Gunbower site forms an extensive area of intact floodplain between the Murray River and Gunbower Creek, and is one of the few such areas with native vegetation in the bioregion (Ramsar Convention 2009).

Criterion 2: Gunbower Forest is an important site and supports at least five wetland-dependent species that are listed as threatened at the national and/or international level. These species include Australasian bittern (*Botaurus poiciloptilus*), swamp wallaby-grass

(*Amphibromus fluitans*), winged peppergrass (*Lepidium monophlooides*), silver perch (*Bidyanus bidyanus*) and Murray cod (*Maccullochella peelii*).

Criterion 4: The Gunbower Forest Ramsar site provides habitat for 66 species of wetland birds, out of which, 48 species have been recorded breeding within the site. The site also supports hundreds of colonial nesting waterbirds during times of inundation and signifies an important habitat for the breeding of native fish (MDBC 2007).

Criterion 8: The site provides migratory paths between habitat in the Murray River, anabranches and floodplains and is considered essential for the recruitment of native fish (King *et al.* 2007).

**Table 3.1 Site details for Gunbower Ramsar wetland (Redrawn from: Hale and Butcher 2011)**

<b>Site Name</b>		<b>Gunbower Forest</b>
<b>Location in coordinates</b>	Latitude: 35° 39' S to 36° 00' S Longitude: 144° 08' E to 144° 30' E	
<b>General location of the site</b>	The Gunbower Forest Ramsar site is located on the Murray River floodplain in the State of Victoria, approximately 30 kilometres northwest of Echuca. Bioregion – Drainage Division 4: Murray-Darling (Australian Water Resources Council 1987).	
<b>Area</b>	19 931 hectares	
<b>Date of Ramsar site designation</b>	Designated on 15/12/1982	
<b>Ramsar Criteria met by wetland</b>	Ramsar criteria 1, 2, 4 and 8	
<b>Management authority for the site</b>	At the time of listing, the site was managed by what is currently called DSE State Forests and Parks Victoria. In June 2010 a portion of the area was designated as a National Park and is currently managed by Parks Victoria.	
<b>Date the ECD applies</b>	1982	
<b>Status of Description</b>	This represents the second ECD for the site, updating DSE 2010.	
<b>Date of Compilation</b>	June 2011	
<b>Name(s) of compiler(s)</b>	Jennifer Hale and Rhonda Butcher on behalf of DSEWPaC.	
<b>References to the Ramsar Information Sheet (RIS)</b>	RIS compiled by Marcus Cooling in 2006.	
<b>References to Management Plan(s)</b>	Department of Sustainability and Environment, 2003, Gunbower Forest Ramsar Site: Strategic Management Plan, Victoria.	

### 3.2.2 Location of research sites- Gunbower Wetlands

In the present study, four wetlands were selected to investigate the changes in the aquatic ecology of the Gunbower system over time. Two of the selected research sites (Black and Green Swamp) are located in the Gunbower Forest area and within the zone recognised under the Ramsar Convention. The other two sites; Taylors and Cockatoo Lagoons are

situated in the dairy farming area in the southern section of the Island. It is evident from reports (e.g., Gell *et al.* 2018a) that the wetlands of Gunbower Forest are considered to be in a degraded condition. However, little information is available on the state of Taylors and Cockatoo Lagoons, which are outside the Ramsar boundary. All of the four selected wetlands receive flows from Gunbower Creek and directly from the Murray River during flood flows. They are therefore, hydrologically influenced by the main River system and its distributary, as well as local influences which includes intensive dairy farming in the case of Taylors and Cockatoo Lagoon.

Black and Green Swamp are located in the lower forest near the Gunbower Creek (Figure 3.5). Black Swamp is a shallow and intermittent open water red gum swamp when inundated and has an approximate depth of 0.5 m. Green Swamp is also a shallow, intermittent wetland of approximate 0.6 m depth. These wetlands provide suitable nesting habitat for small populations of waterbirds, native fish, frogs and they support aquatic meadow vegetation (Northern Central CMA 2015). Considered today to be permanent aquatic habitats, the wetlands are valued as refuges for aquatic biota that can recolonise the forest when flooding returns after dry conditions.

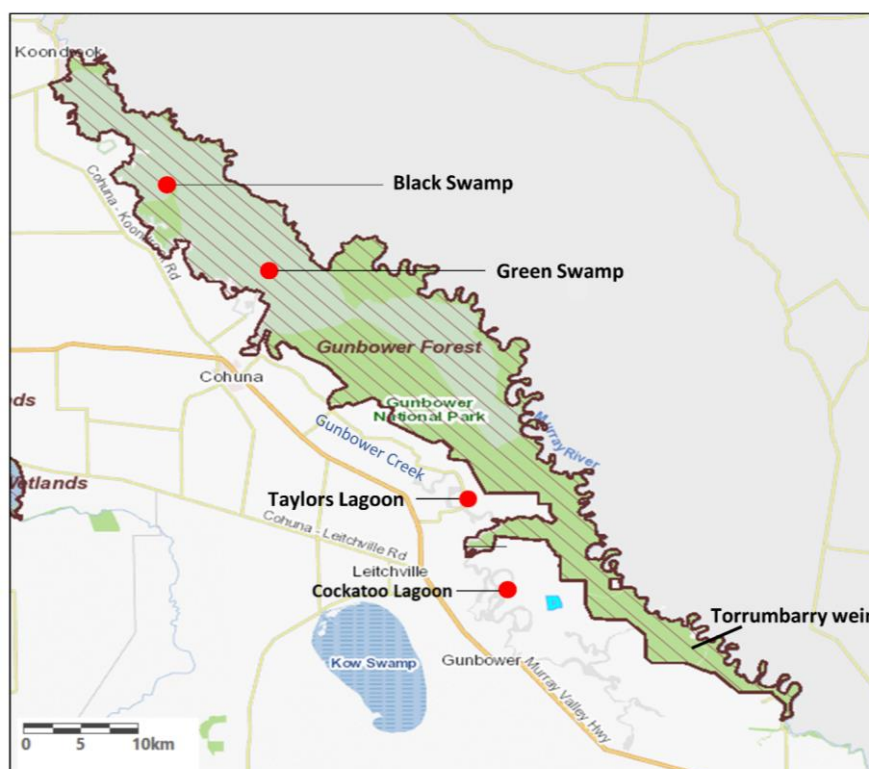


Figure 3.5 Location of the study sites (Black Swamp, Green Swamp, Taylors and Cockatoo Lagoon) in the Gunbower Forest (shaded green) near Cohuna. The Gunbower Region is regulated by Torrumberry Weir.

Cockatoo and Taylors Lagoons (Figure 3.5; McInerney *et al.* 2017) are shallow (<2 m deep), loop-shaped, natural billabongs, regulated by the Thompsons Weir pool and Cohuna Weir, respectively (see Figure 3.7). Cockatoo Lagoon is connected to Gunbower Creek at three points, being the inlet, outlet and at the north-west edge via an excavated channel knows as ‘Jumbo’s Cut’ (constructed to aid in flushing water to reduce blue-green algae blooms). The deeper sections of the lagoon are known to support a series of pool habitats (North Central CMA 2014a). Taylors Lagoon is connected to Gunbower Creek via a narrow-excavated connection at the upstream end and a broad pool at the downstream end (North Central CMA 2014c). Original pictures of Taylors and Cockatoo lagoon are shown in Figure 3.6



Figure 3.6 Original pictures of (a) Taylors and (b) Cockatoo Lagoon, Captured on 12/05/2018.

### 3.2.3 Gunbower creek flows and regulators

Historically, the hydrology of the Murray River and its tributaries were largely managed for water supply, flood mitigation, navigation and hydroelectricity production. In order to offset the subsequent reduced peak river levels, and so inundation events, regulators were constructed on several streams within the Gunbower Forest in the 1930s and 1940s where they leave the Murray River. The hydrology within the site then has been regulated and managed since this time to introduce water to the forest in a number of places. Since it is regulated, the timing of releases does not rely exclusively on the availability of flow peaks in the main river. Gunbower forest receives water from Gunbower Creek that flows in a north-westerly direction, forming the southern boundary. The creek meanders through agricultural land before reaching the remnant Gunbower Forest. The Gunbower Creek is

regulated by five major regulating structures dividing the creeks into a series of fast and slow-flowing reaches. These regulating structures are Gunbower, Thompson, Hipwell, Cohuna and Koondrook weirs (North Central CMA 2015). The location of the regulating structures are provided in Figure 3.7 (Stuart *et al.* 2019).

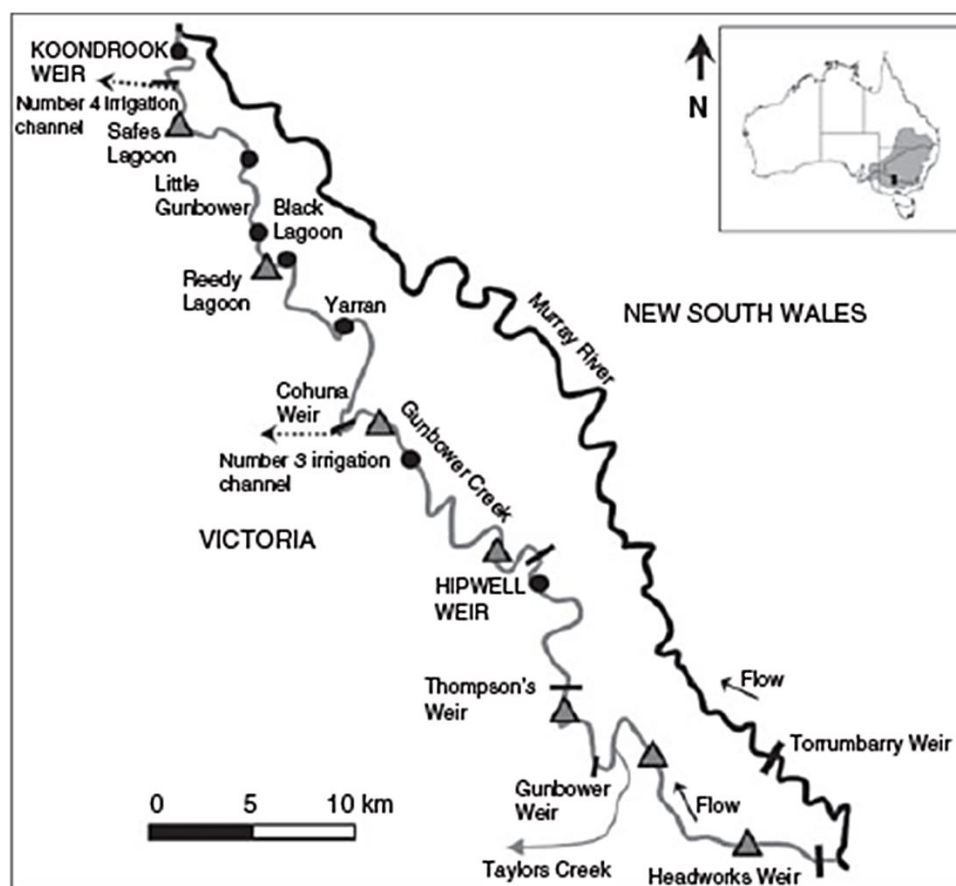


Figure 3.7 Location of Gunbower creek and regulators in north-central Victoria within MDB.

The Gunbower Creek system is an integral part of the Torrumbarry Irrigation Area (TIA). It supports agriculture in various localities including, Kerang, Murrabit and Benjeroop localities as well as the communities of Gunbower, Leitchville and Cohuna. The upper reaches of the Gunbower Creek system support twelve lagoons of different characteristics. These generally include natural and artificial excised cut-off meanders of the original creek line and a series of oxbow lakes (North Central CMA 2015).

The nature of inundation of low-lying creeks and wetlands within Gunbower Forest is a very complex process influenced by numerous creeks, wetlands and hydraulic controls. The wetlands of the lower reaches of Gunbower Creek are Black Swamp, Reedy Lagoon, Little Gunbower Wetland, Iron Punt Lagoon and Whistler Lagoon. All of these wetlands are interconnected by a series of small creeks with various commence-to-flow levels. The Little Gunbower wetland complex is connected to Gunbower Creek via the Little

Gunbower Creek regulator and the River Murray via Barham Cut. However, Black Swamp and Reedy Lagoon are directly linked to Gunbower Creek by regulators.

### **3.2.4 Hydrological change**

The Gunbower Forest receives significant inflows through the inlet channels at river flows of over 15,000 ML/day (Cooling *et al.* 2002). Prior to regulation, overbank flooding events of >30,000 ML/day in the Murray River occurred six to seven years in ten, inundating the Gunbower Forest floodplain and its wetlands (Mallen-Cooper *et al.* 2014). During flooding events, excess floodwater, rich in organic matter, minerals and nutrients drained into the Gunbower Creek before re-entering the Murray River at the downstream end of the system. In 1888, the Irrigation Act was passed by the Victorian Government, after which four Irrigation Trusts in the Cohuna-Torrumbarry regions were introduced (Harris 2005).

In 1923, Torrumbarry weir was constructed to regulate water in the Gunbower forest at a steady flow of 1,650 ML/day, out of which about 700 ML/day is directed into Gunbower Forest and 400 ML/day passing into Gunbower Creek for environmental benefits (Cooling *et al.* 2002). This diversion has changed the frequency, duration and seasonality of the flow regime and altered natural wetting and drying within the creek and lagoons. Moreover, in the case of large natural floods, the National Channel offtake regulator and associated levee system is designed to hold back high-water levels rather than be overtopped (Mallen-Cooper and Director 2018). The sections of the creek are lined with levees, designed to increase the capacity of the creek and reduce the occurrence of overbank flooding. So, there is reduced input of floodplain organic matter, carbon, nutrients and biota into the creek. The watering regime of the creek is now also seasonally inverted with prolonged high flows throughout spring, summer and autumn and low to no flow during winter (Anderson *et al.* 2007). During these days, from spring to autumn, the lagoons are operated at full water (Howard *et al.* 2013). There is a lack of water level records for the Gunbower Forest, so, it is not possible to empirically relate river flows to the extent or depth of inundation of the floodplain. The behaviour of floodplain flows was therefore estimated from the flow thresholds of the main channels entering the forest. Cooling *et al.* (2002) reported pre- and post-regulation flow conditions for Gunbower Forest based on the Murray Darling Basin Commission Monthly Simulation Model (MSM). The model was used to estimate monthly river flows at Torrumbarry for the period from 1891-1990 under pre- and post- regulation conditions. It was estimated that, prior to regulation and diversion, the Murray River levels typically increased from May and reached a peak in August to October while they declined from October to February and remained at low levels until April.

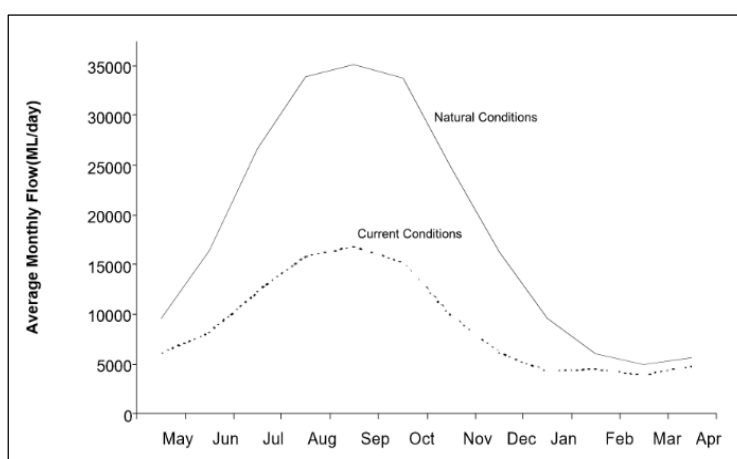


Figure 3.8 Average monthly river flows at Torrumbarry for current and simulated natural conditions (Reprinted from: Cooling *et al.* 2002)

Under regulated conditions, the highest average monthly flows in September were closer to 15,000 ML/day, whereas it was approximately 35,000 ML/day typically before regulation (Figure 3.8). In addition, the average daily discharge data (ML/day) was obtained from the online monitoring of Torrumbarry Weir (1950 to 2020) (Victorian Data Waterhouse 2020a). The records of flows in the river at Torrumbarry weir often increased above thresholds as high as 40,000 ML/day, signifying their contribution in initiating flooding in the Gunbower Forest region (Figure 3.9). Furthermore, regulation of the Murray River and changes to inflow points (e.g., blockages) has created drier conditions across Gunbower forest causing an overall shift towards more terrestrial vegetation types (North Central CMA 2014b). The impacts were exacerbated by ten years of drought (2000-2009) that resulted in a reduction in vegetation condition, floodplain productivity and access for native fauna to food and habitat.

Following the Millennium Drought (2000-2009), Gunbower Forest received floodwaters for three consecutive years (2010-2013) resulting in inundation of more than 50% of Gunbower Forest including substantial areas of Gunbower National Park (Bennetts and Jolly 2017). Floodplain and wetland dependent flora and fauna increased in cover due to these events, with the diversity of understorey species increasing and canopies of River Red Gums showing signs of recovery from drought (Bennetts and Jolly 2017). Waterbird feeding and nesting was also observed and many hundred colonial waterbirds bred in the lower forest (North Central CMA 2014b).



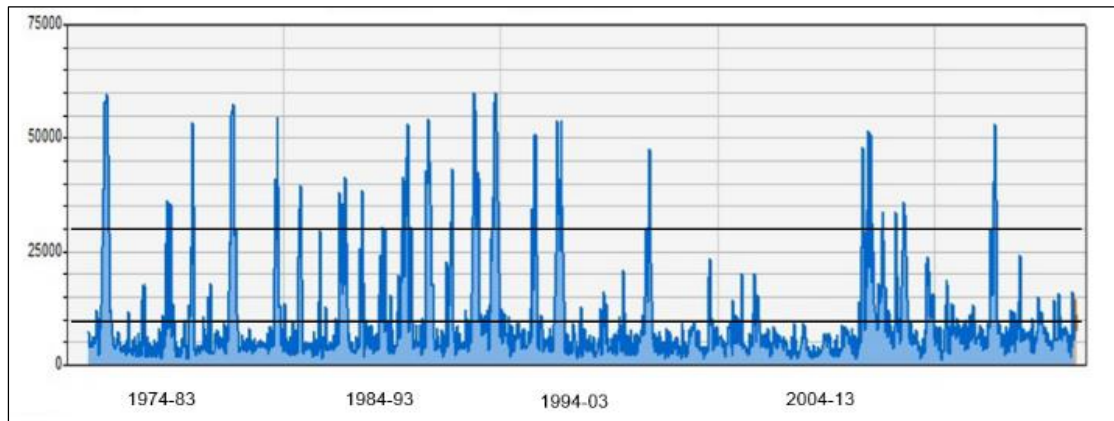


Figure 3.9 Daily discharge (ML/day) in the Murray River downstream of Torrumbarry Weir from 1975 to 2020. Black lines show commences to flow level (13700 ML/day) and threshold for broad scale inundation (30000 ML/day). (Source: Victoria Data Warehouse 2020a)

In simple terms, it is widely argued that river ecosystems are governed principally by the flow regime (e.g., Poff *et al.* 1997). The health, growth and existence of the forest ecosystem are dependent on regular winter-spring inundation from high river flows (VEAC 2008), yet extended inundation can reduce the vigour of trees. In a most recent preliminary study on the long-term condition of Gunbower wetlands, it is reported that prior to regulation, the water was clear, oligotrophic, slightly acidic and fresh while degraded water quality is evident as a consequence of post regulation activities (Gell *et al.* 2018a). It is suggested that river regulation has altered floodplain and river interactions thus influencing water quality.

### 3.2.5 Water quality of Gunbower wetlands and surroundings

Gunbower Creek's water quality is highly variable and is influenced by the Murray River's water quality, the floodplain, and the creek's hydrology. Floodplain inundation is an important factor impacting the water quality in permanent and frequently flooded wetlands, river flow disruptions, and pollution (Woodward *et al.* 2016). The total volume of floodwater has been shown to dilute water chemistry and biota (Gell *et al.* 2002; Walalite *et al.* 2016). Floodwaters can also bring in nutrients, favouring certain algal assemblages (Paerl *et al.* 2016). Further, phytoplankton during peak floods are associated with increases in nutrients and decreases in light (Talbot *et al.* 2018).

The flooding of forested floodplains and wetlands may trigger 'Blackwater events' (Howitt *et al.* 2007; Hladyz *et al.* 2011). These are described as flood occurrences that bring elevated levels of dissolved organic carbon (DOC) in the water column, responsible for bacterial metabolism during spring which in turn drives consumption of dissolved oxygen. The transfer of organic carbon from floodplains to the river channel encourages the



nourishment of the riverine food webs. Hence, Blackwater events are assumed to be natural and are considered important in maintaining productivity of river and floodplain environments (Junk *et al.* 1989). However, if there is a long gap between flood events, the plant matter including dead leaves build-up on the floodplain instead of being washed into the river and bacterial decomposition of plant matter occurs. When rain or flood event occurs, the build-up of dead and decaying matter flows into the river and bacteria continue to break down the leaves and other plant matter. This microbial respiration consumes lots of oxygen, so there is less oxygen for fish and other aquatic organisms to breathe. Dissolved oxygen concentrations can fall below the tolerances of fish and other aquatic fauna (Howitt *et al.* 2007; Small *et al.* 2014). Generally, dissolved oxygen levels below 4 mg/L are considered to impose stress on aquatic biota and in many cases fish kills have occurred even below 2 mg/L (Howitt *et al.* 2007).

In September 2010 and April 2011, extensive hypoxic Blackwater events occurred in the southern MDB. These events impacted the majority of the lowland river-floodplain systems. Blackwater events were triggered by unseasonal, post-drought inundation (from rainfall) influencing both forested and agricultural regions (Whitworth *et al.* 2012). Between March and May 2012, another, unseasonal (summer and autumn) natural flooding and several hypoxic Blackwater events were observed across the MDB. However, the time and magnitude of these events were not as impactful as that recorded in 2010–11 (Whitworth and Baldwin 2012). In 2010, areas of the Gunbower forest floodplain witnessed Blackwater events causing declines in dissolved oxygen (less than 1 mg/L). As a consequence, oxygen concentrations in Gunbower Creek decreased substantially (Hale and Butcher 2011).

According to online monitoring data, water quality in the main channel of the Murray River is generally fresh with salinity below 400  $\mu\text{S}/\text{cm}$  (1992 to 2008; Victorian Data Warehouse 2020). Turbidity is moderate with a median value of 18 NTU and a ninetieth percentile of 40 NTU downstream of Torrumbarry Weir (Victorian Data Warehouse, 2020). River water salinity is mostly fresh, but in times of low or no flow can be considered oligosaline with electrical conductivity rising above 2000  $\mu\text{S}/\text{cm}$ . Similarly, turbidity varies from low (less than 10 NTU) to relatively high in April (almost 100 NTU; Figure 3.10). Furthermore, it is evident from the pH readings that water quality in the system varied from neutral to alkaline but was episodically acidic (Victorian Data Warehouse 2020; Figure 3.10).

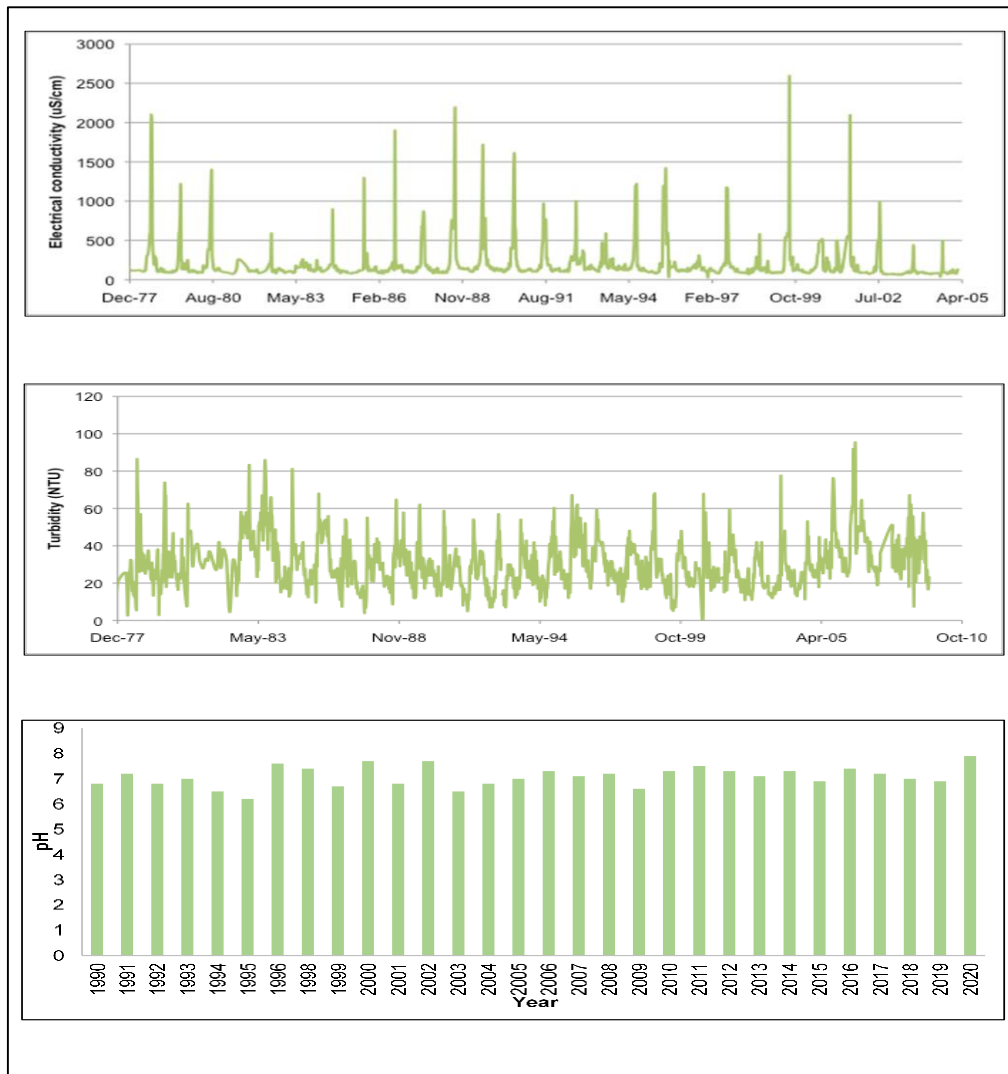


Figure 3.10 Electrical conductivity, turbidity in Gunbower Creek from 1977-2005 and pH fluctuations from (1990-2020) (Source: Victorian Data Warehouse 2020)

Under the Living Murray intervention monitoring program, water quality measurements were taken at a number of locations in the Gunbower Creek and lagoons located in the dairy area (North Central CMA, 2015). It was reported that values of turbidity and nutrients increased throughout Gunbower Creek and regularly exceed the ANZECC (2000) guidelines. This is caused by the slow degradation to the submerged clayey banks of the creek, which also aligns with an increase in phosphorus associated with the fine clay fraction that causes turbidity (Anderson *et al.* 2007), and inputs from the surrounding agricultural land including farm drainage, runoff and stock access. These conditions contribute significantly to eutrophic conditions which are often indicated by the presence of *Azolla* spp., and Pale-Yellow Water Lily (*Nymphaea* spp.). Furthermore, in the past, occurrences of blue-green algae were also noticed in Gunbower Creek and lagoons indicating degraded water quality (North Central CMA, 2015).

In the light of the above short-term water quality information, it can be concluded that the water quality has changed and has become degraded since regulation. During the last 15 years, Gunbower Forest has experienced both drought and flood. Specifically, the period between 2005 and 2014 included one of the longest and driest periods on record (the Millennium Drought, 1997-2009), with unusually low autumn rainfall, followed by one of the strongest *La Nina* episodes (2010-2011), which resulted in widespread above average rainfall (Figure 3.11) (BOM 2019) and flooding across the Murray Darling Basin (Prosser 2011). These dry conditions have likely exacerbated the cases of poor water quality. However, due to a lack of monitoring after the previous large drought, the precedence of these events is unknown.

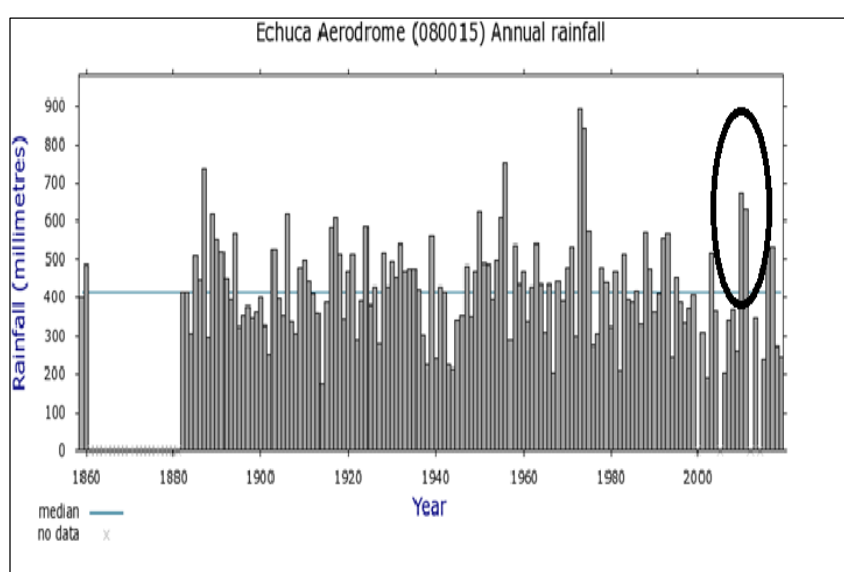


Figure 3.11 Annual rain fall at Echuca between 1860-2019 (Source: BOM 2019). Black oval is highlighting rainfall evenmts of 2010-2011

### 3.2.6 Human disturbance

Land use in the catchment surrounding the Gunbower Creek System is dominated by irrigated agriculture. Irrigation started in the Gunbower district with the formation of irrigation trusts in 1886 with water being directly pumped from the Murray River and Gunbower Creek (Mallen-Cooper and Zampatti 2018). The Gunbower Scheme involved several important works to improve the reliability of supply from the Murray River. This included the construction of headworks, the deepening and widening of the intake of Gunbower, Taylors and Deep creeks and the construction of the National Channel in 1890 (Mallen-Cooper *et al.* 2014). This resulted in the excision of a number of natural meanders in the upper Gunbower Creek, forming a series of lagoons now known as Heppell, Unregulated, Splatts, Turners and Upper Gunbower lagoons (North Central CMA, 2013c).

Dairy is the dominant agricultural industry among other agricultural activities that include irrigated pasture, cattle and sheep, stone fruit, vegetables and orchards. The area also supports pig farming and sand dune extraction enterprises (specifically at Gum and Taylors Lagoons respectively) as well as lifestyle properties and hobby farms (North Central CMA, 2013a, b). Furthermore, un-managed recreational activities can adversely affect the ecosystems of wetlands. Vehicle tracks, for instance, can compact the soil, impacting flora and fauna, and increasing the supply of fine sediments to wetlands.

### **3.3 Summary**

The Gunbower Forest is a large floodplain island that hosts a diverse suite of distributary creeks and shallow wetlands. The main drivers of recent change in the Gunbower wetlands include the impact of river regulation and catchment disturbance on flooding regime, season of flow, inundation permanency, turbidity and nutrients. Fewer floods occur under the present regime and often occur for a shorter duration, meaning that less water reaches the forest and its plants and animals. Prior to this regulation, Gunbower Forest was naturally flooded around seven out of every 10 years and the present lack of flooding has impacted negatively on many wetland and floodplain ecosystems. In contrast, while subjected to floods *per se* less frequently, 87 % of wetlands are now permanently inundated as their inlets connect to the river below the regulated level and the permanently elevated nature of Gunbower Creek limits the drainage of water.

The issues discussed in this chapter have provided limited evidence of ecological changes in Gunbower wetlands owing to anthropogenic pressures and climate variations. Further, due to lack of continuous monitoring, causes and timing of these changes are largely unknown. So, to improve the understanding of drivers of the observed changes and to achieve the main research aims as discussed in Chapter 1, a multi-proxy approach has been employed. The research approach as well as site selection, sediment sampling, data collection and analysis methods used in this research, are elaborated in the next chapter.

## CHAPTER 4- FIELD, LABORATORY AND STATISTICAL TECHNIQUES

### 4.1 Introduction

This chapter provides an overview of the research design and methods. As discussed in Chapter 1, the study is designed to investigate the ecological response of seasonal or intermittent wetland systems to climate variability and anthropogenic impacts in south-eastern Australia. Three main objectives are: 1) to establish the response of wetlands to anthropogenic pressures at each wetland over time; 2) to understand the wetland change in relation to past climatic variations; 3) to explore drivers of change and sources of sediment influx into the wetlands. This chapter will give a background and overview of the techniques used in this study and how they were applied to the sedimentary sequence at each site.

To address the objectives and research questions, a research framework (Figure 4.1) is developed to reconstruct past water quality trends and ecological conditions to understand trajectories of wetland change, responses from past events and any historical legacy that may help explain recent observations. In section 4.2, the field techniques employed, including site selection for four study sites through to sediment coring methods, are outlined. Section 4.3 focuses on laboratory techniques such as diatom sample preparation, species identification, Itrax scanning,  $^{210}\text{Pb}$  dating and stable isotope analysis. The final section 4.4 is concerned with the statistical techniques used to explore the relationships among and between indicators to underpin an understanding of water quality and wetland ecosystem changes.

Sediment cores were collected from four wetlands across Gunbower Forest. Thereupon, sediment cores were subsampled and analysed internally at Federation University Australia laboratories for diatom analysis. Subsamples were taken for external analysis of other proxies and this was completed at the Australian Nuclear Science and Technology Organisation (ANSTO) using facilities such as Isotope Ratio Mass Spectrometry (IRMS) (Meyers and Ishiwatari 1993; Talbot 2001), Itrax core scanning (Croudace *et al.* 2006) and  $^{210}\text{Pb}$  dating (Fink *et al.* 2004).

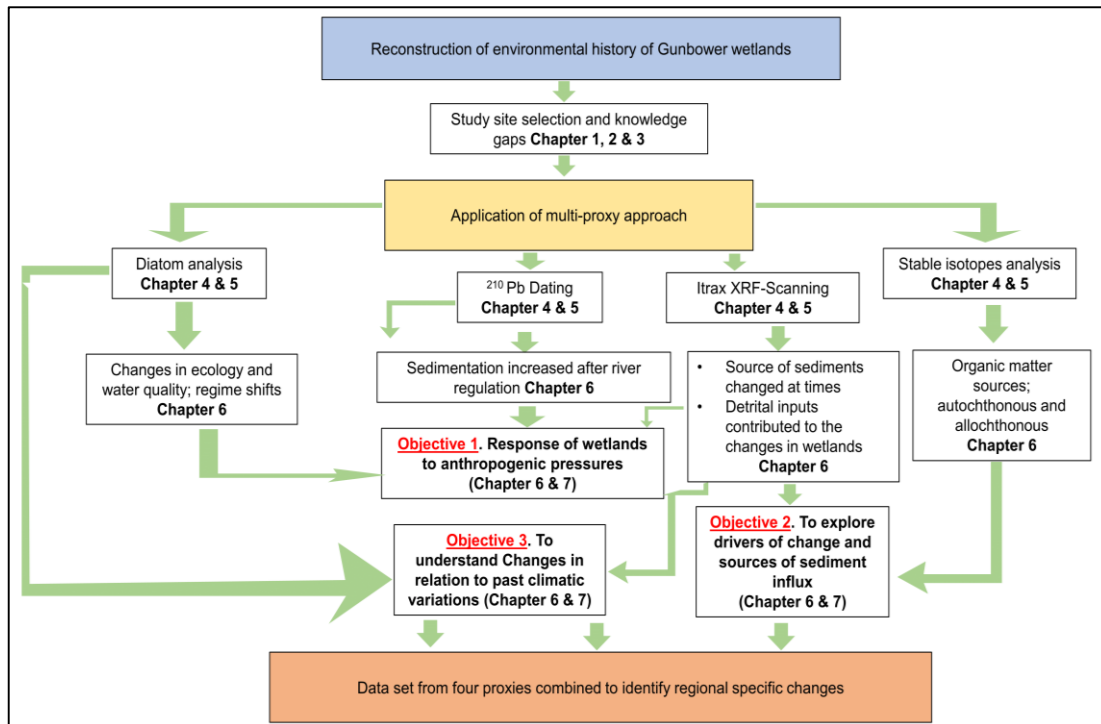


Figure 4.1 Research framework to reconstruct environmental history of Gunbower wetlands

## 4.2 Field techniques

### 4.2.1 Selection of study sites

The aim of this project was to examine the ecological responses of wetlands to climatic fluctuations and human disturbance within the temperate climate transition zone of northern Victoria, Australia by analysing wetland sediment sequences. In particular, the study goal was to characterise the wetlands in the context of regional or local drivers of change, and to determine if water quality has changed due to climate change and catchment inputs. Specifically, this study sought to assess whether wetlands have changed in response to changes to factors such as flow regime or are they associated with local land-uses such as dairy farming. It is known that in some locations around the Murray River, the water quality has been significantly changed after river regulation and impacts are exacerbated with climate change and land use of the surrounding catchment (Gell *et al.* 2007). Consequently, study sites were sought that would reflect and record both local and regional influences. So, sites were selected to characterise the nature of wetland change in contrasting locations. Two wetlands were selected from those located in the middle of the forest surrounded by relatively undisturbed natural vegetation, and two wetlands were selected from those situated in dairy farm regions and so are likely to be more directly influenced by local human interference. Simply, as all four sites were likely similarly influenced by regional influences, receiving water from both the River

Murray and Gunbower Creeks, any clear difference between sites in the two local settings would reveal the effect of local forces relative to those derived from the region.

Site selection is critical to achieve these aims and, as such, several criteria were applied to identify locations that would be the most suitable in the reconstruction of ecological and water quality changes. These criteria are outlined below and summarised in the following sections:

#### **4.2.1.1 Selection criterion 1**

Ramsar site; firstly, the Gunbower Forest was selected as it is recognised as a site of international significance in 1982 under the Ramsar Convention. Part of the significance of the site relates to human occupation as it has been subjected to human use for thousands of years, with abundant evidence for intensive Aboriginal occupation that includes earth mounds, clay balls and widespread canoe trees (Pardoe *et al.* 2014; McConachie *et al.* 2020).

When the Gunbower Forest was listed as a wetland of international significance in 1982 under the Ramsar Convention, the state and federal governments identified its 'natural ecological character'. The Gunbower Forest was shown to satisfy four of the nine listing criteria, those being listed as 1, 2, 4 and 8 (see Chapter 3). The government is obliged to report to the Ramsar Secretariat if a listed wetland 'has changed, is changing or may change in the future (Davidson 2016). So, the status of the site, and the requirement to monitor its condition, provided an opportunity for an analysis of the sediment record to be incorporated into broader assessments of the site's condition.

The Gunbower Forest is also listed as an icon site under the Living Murray Program (Cooling *et al.* 2012). Here, the wetland sites are targets of restoration through the provision of water from the Murray River and its distributary Gunbower Creek. To achieve this, the government has invested in substantial infrastructure to provide environmental flows to the wetlands in an attempt to ensure that their ecological character and biological diversity are maintained and enhanced. Again, this provided a clear opportunity for sediment-based analyses to contribute to an assessment of the need and benefits of this restoration measure.

#### **4.2.1.2 Selection criterion 2**

The second selection criterion was the capacity of the sites to yield sediment cores suitable for analysis. In some cases, retrieving sediments from the wetlands or lakes in the Murray River floodplain represents a challenge. Under conditions when floodplain

lakes are full, the slack water conditions allow for the accumulation of sediments. Preservation of wetland sediments presents an archive of evidence for biological and sedimentological changes through time within and around the wetlands. Such sediment sequences are available in sites that have remained permanent for extended periods, and are generally lost from those which regularly dry.

In October 2017, several cores were retrieved from the Gunbower forest by Prof. Peter Gell and his team who were working in collaboration with Melbourne University and the North Central Catchment Management Authority on the character of wetlands at Gunbower. They established the presence of suitable sediments, the preservation of fossil diatom indicators, inferred the age of the records and proposed that the sedimentary records reveal substantial environmental changes (Gell *et al.* 2018a). According to this preliminary study on Gunbower Forest, it was concluded that wetlands have responded to climate change and human occupation. This study ascertained the suitability of wetlands for paleoecological analysis and recommended a broader, multi-proxy approach for detailed analysis of the Gunbower Forest wetlands.

#### **4.2.1.3 Selection criterion 3**

In order to maximise the ability to reflect and compare water quality changes of wetlands due to local or regional impacts, the third criterion was that two of the suites of studied wetlands should be located within the zone of irrigated dairy lands. Therefore, of four wetlands to be studied, two were selected from those located near within the dairy farming area, and the remainder were selected from those that were subject to the preliminary study in the centre of the forest. This selection of sites was designed to explore the degree to which the wetlands were responding to local factors, thereby enabling these impacts to be isolated from landscape or regional scale factors which can confound interpretation of the record of individual wetlands (Fritz 2008).

#### **4.2.1.4 Selection criterion 4**

The fourth criterion was related to the significance of these sites. These sites were not previously studied in detail, the only previous work in which limited sedimentary studies had been undertaken on was Black and Green Swamp (Gell *et al.* 2018a). However, in that study, two cores were retrieved from the sites and no dating was undertaken to establish a robust chronology. In that report only two wetlands out of four were briefly discussed on the basis of diatom and pollen analysis. Sediment samples were taken at 5 cm intervals and prepared for diatom analysis undertaken in coarse resolution (Section 4.2). These preliminary diatom analyses confirmed that diatoms were present throughout



the sedimentary record and that there were substantial changes in the fossil diatom record. It was hoped that this would provide further insight into the ecological character of the wetlands.

Thus, after following strict criteria to identify potential sites suitable to a high-resolution study, Taylors and Cockatoo Lagoon (in dairy lands), and Black and Green Swamp (in forest) were selected as the study sites for multi-proxy water quality reconstructions (Figure 4.2). Importantly these proved to have relatively deep sediments maximising the temporal depth that would be provided by detailed analyses. Therefore, four wetlands in contrasting settings were chosen to quantify impacts of anthropogenic pressures, climate change, river regulation, sedimentation and nutrient enrichment on the wetlands of the Island.

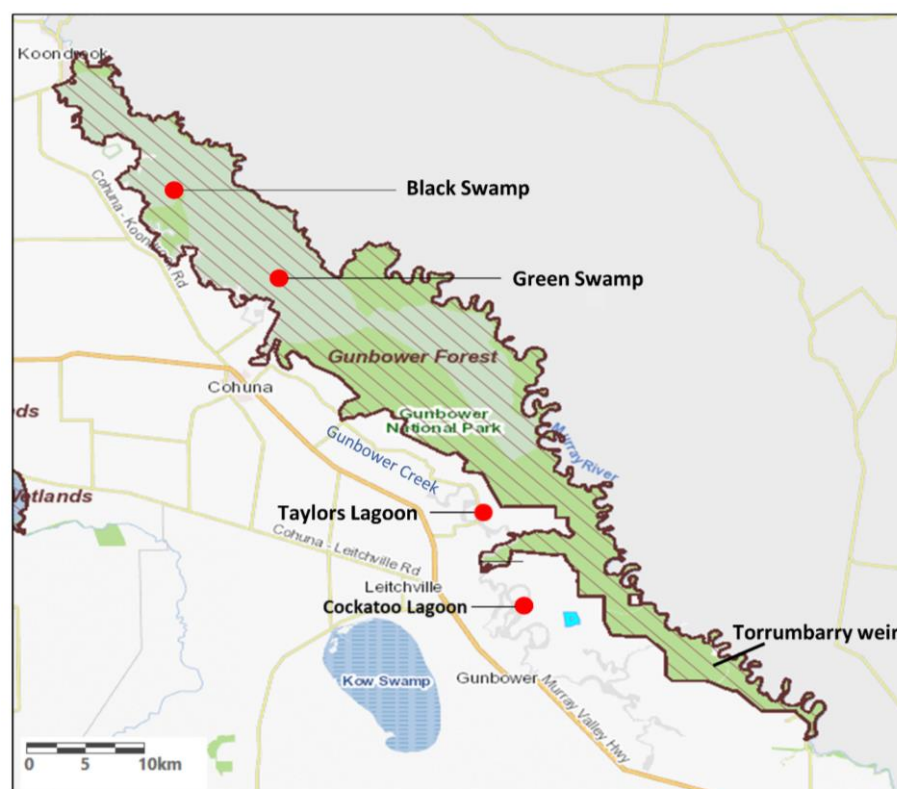


Figure 4.2 Location of the study sites (Black Swamp, Green Swamp, Taylors and Cockatoo Lagoon) for sampling in the Gunbower Forest.

#### 4.2.2 Sediment coring

In the absence of detailed bathymetric measurements, samples were taken from the centre of the wetland under the assumption that the sediment records in this location would be representative of the entire wetland and would have the least likelihood of a bias towards benthic or littoral species, or planktonic species (Vleeschouwer *et al.* 2010). The

multi-core study of Grundell *et al.* (2012) revealed that centrally located cores integrate allochthonous and autochthonous elements and so are generally representative of the entire wetland. There are numerous methods of retrieving sediment cores for analysis depending on the type of sediment, the depth of the lake, the aims of the study, the biological proxy being examined and, often, by the resources available (e.g., Skilbeck *et al.* 2017). In all cases however, the common goal is to retrieve the desired amount of sediment while maintaining the stratigraphic integrity of the sediment column.

Sediment samples were collected using a Russian (or d-section) corer that is used to extract sediment cores in 50 cm contiguous segments (Jowsey 1966). Coring continued until the nose of the corer met with resistance, thus halting the process (Vleeschouwer *et al.* 2010). Special care was taken in calculating the target depth for each sampling core. Several segments were collected at each site with overlaps of 10, 20 or 30 cm commonly taken. It is important to note that, in order to match a composite core, both the upper and lower overlaps are included within each core. Sediment cores from four different wetlands of Gunbower forest were collected in two field trips. Cores were collected from Black and Green swamp in October 2017 whereas Taylors and Cockatoo Lagoons were collected in May 2018. The wetlands and length of sediment extracted are:

Black Swamp (84 cm)

Green Swamp (86 cm)

Taylors Lagoon (95 cm)

Cockatoo Lagoon (74 cm)

After each core was extracted, it was carefully packed and transported to the Federation University sediment core repository (~ 4°C). This involved opening the corer chamber, covering the core with plastic film, and placing it in a length of rigid plastic PVC half-tube (longitudinally sliced plastic tubing of suitable internal diameter) to protect it from damage and compaction during transport (Vleeschouwer *et al.* 2010). Afterwards, multiple layers of kitchen-grade polyethene were used to cover the sediment cores. After individual packing and labelling, the cores were kept in the container and replicates of cores were kept in a portable fridge (4°C) for transportation to the Federation University.

### **4.3 Laboratory techniques**

Sediment cores from each site were sliced and sub-sampled at every 1 cm to store separately in clear labelled bags at 4°C and -20°C at the Federation University laboratory. Further, sub-sampled cores were treated for diatom analysis. At a later stage, frozen

samples were freeze-dried and homogenised for stable isotope analysis. The rest of the cores were repacked and taken to ANSTO for Itrax and  $^{210}\text{Pb}$  analysis.

The techniques employed in the Federation University laboratory consisted of the preparation of diatom slides from fossil samples, and the identification and enumeration of diatom taxa. Sample preparation was also undertaken for radiometric dating and by sieving for stable isotope analysis. As samples were analysed using different techniques so sample preparations were different for each technique. These are described separately in the respective sections below.

#### **4.3.1 Chronology**

Lead dating is a well established approach to establish chronologies for relatively recent sediment sequences (Jia *et al.* 2018), and has been widely used in Murray River wetland studies (Gell *et al.* 2005b; Fluin *et al.* 2010; Reid *et al.* 2007; Grundell *et al.* 2012; Kattel *et al.* 2015) and therefore was utilised in this study. The isotope enters waterbodies through precipitation and dry deposition. This method is used to trace radioactive decay of the naturally occurring isotope  $^{210}\text{Pb}$  which is accumulated in sediments from aerial fall-out. The isotope has a half-life of 22.3 years and the dating method, where local activities are high, can span approximately five half-lives. This generally corresponds to maximum dating range of 110 years. The age of the sediment is determined by comparing activities of unsupported  $^{210}\text{Pb}$  at the sample intervals with the activity at the surface sample (Gell *et al.* 1993).

Accurate dating of sediments is a prerequisite for the paleolimnological study of wetlands (Charles and Smol 1994). It is necessary to establish a set of timelines to determine depth-time relationship in order to reconstruct environmental changes that are based on diatom (and other) assemblages (Dixit *et al.* 1992; Hall and Smol 2010) and elemental composition, as well as to determine sedimentation rates. As the focus of study was to reconstruct environmental changes and the water quality of the wetlands, the specific aim of the study was to support this with a fine-resolution chronology. This required dating methods that focussed on the last approximately 100 years and this was undertaken for all four wetlands so as to encompass much of the period of European settlement in the region. In lakes with small catchments (e.g., crater lakes) almost all of the  $^{210}\text{Pb}$  is derived from the atmosphere and so the decay model provides reliable dates for sediment layers. In fluvial settings such as floodplain lakes however, some  $^{210}\text{Pb}$  is derived from river-borne sediment and this complicates the establishment of a chronology. To understand the chronology of the wetlands, all dating for this project was undertaken at Australian Nuclear

Science and Technology Organisation (ANSTO) laboratories (Fink *et al.* 2004) in Sydney, Australia.

#### **4.3.1.1 Sample preparation for $^{210}\text{Pb}$ dating**

All pre-treatments for dating were performed at the Australian Nuclear Scientific and Technology Organisation (ANSTO) laboratories. The first step was to log the sediment samples in the ANSTO system. Sample IDs were generated and records were maintained for the samples to be processed for dating. Between 1-5 g of each sample required for dating was weighed and placed into pre-labelled beakers. The samples were dried overnight in a sediment oven at 60 °C, and then weighed again to ensure adequate material was available for Lead ( $^{210}\text{Pb}$ ) dating. The dried material was homogenised with an agate mortar and pestle and placed into 70 ml plastic vials labelled with the sample location and depth. Samples were then treated using standard methods and the total activity of  $^{210}\text{Pb}$  was established by measuring the alpha emitting grand-daughter isotope Polonium ( $^{210}\text{Po}$ ) (Appleby and Oldfieldz 1983; Appleby *et al.* 1988; Appleby 2002).

#### **4.3.1.2 Determination of $^{210}\text{Pb}$ chronologies**

Profiles of unsupported  $^{210}\text{Pb}$  activity in sediment sequences may be used to establish chronologies by the use of one or more models. Both the constant rate of supply (CRS) and constant initial concentration (CFCS) models were used to calculate estimated ages and mass accumulation rates for each wetland (Appleby and Oldfieldz 1983). Where it has been possible to check the results of the two models using other methods of dating, the CRS model appears to give the better results for a non-monotonic profile i.e., where Pb activities do not decrease exponentially with depth (Appleby and Oldfieldz 1983). Depending on the characteristics of the study sites, those with small or large catchment inputs, CRS or CIS is selected. To summarise, the CRS model is applied when regular, mostly atmospheric supply is assumed while CFCS accommodates variable rates of sediment input.

In this study,  $^{210}\text{Pb}$  samples were analysed using both models depending upon the site characteristics. In Cockatoo Lagoon, the CRS model was used to calculate the Pb profile whereas both models (CFCS and CRS) were used for the Taylors Lagoon and Black Swamp analysis. However, for Green Swamp, the CFCS model was used. The samples from three sites; Taylors Lagoon, Black and Green Swamp were analysed in two stages for  $^{210}\text{Pb}$  analysis. In the first stage, samples between 0-30 cm were analysed but the unsupported  $^{210}\text{Pb}$  activity was very high between these depths. Hence, at a second stage extra samples below 30 cm were analysed to identify the point at which levels of the  $^{210}\text{Pb}$

activity were at background levels to estimate period of less pollution or pre-industrialisation.

#### **4.3.2 Itrax scanning**

The Itrax-XRF (X-Ray Fluorescence) is a non-destructive core-scanner which is an important instrument to undertake elemental analyses on limited volumes of sedimentary material. Itrax provides a continuous record of relative elemental composition and records fluctuations corresponding to environmental, sedimentological and diagenetic changes (Croudace *et al.* 2006). Under the ANSTO protocol, the Itrax is first used to scan the core's surface, taking a radiographic image and high-resolution CCD-image. The radiograph reflects the organic content and relative density of a sample. The  $\mu$ XRF technique available with the Itrax provides the elemental composition of rocks and sediments (Weltje and Tjallingii 2008). The source of the X-ray is a Molybdenum K-ray tube that excites electrons by incident x-radiation at 500  $\mu$ m increments for a 10 second period. The X-ray excites a 4 mm x 0.2 mm track to a depth of 0.1 mm, causing electrons to eject from the inner to outer shells. Secondary x-radiation is released as the electrons collapse back to their original position in the inner shell. The emitted photon (or florescence) energy and wavelength spectra are characteristic of atoms, producing an estimated relative abundance (Croudace *et al.* 2006).

The relative abundance of elements is produced as thousand counts per second (KCPS), not concentration, and often requires normalization prior to any kind of analysis and interpretation (Kylander *et al.* 2011). The purpose of normalizing the data is to account for the variable distribution of organic matter, water content, pore size and grain size (Croudace *et al.* 2006). Throughout this thesis the Ti counts were used to normalise data, as it is an unambiguous indicator of allochthonous inputs from the catchment (Cohen 2003).

The Itrax core scanner used in this study is unique among the current generation of core scanners in being designed to gather optical and micro radiographic images and  $\mu$ XRF elemental profiles for the same sediment core section (Croudace *et al.* 2006). It can operate on split cores of sediment or rock with a maximum length of 1800 mm and a diameter ranging from a few cm up to 12 cm. At ANSTO, Lucas Heights, Australia, preparation of the cores for scanning was performed. The surface of cores chosen for Itrax analysis were cleaned, levelled and covered with an ultra-thin conductive plastic film. Cores were then scanned through the scanner, allowing for radiographs and elemental composition to be obtained.

The relative abundances of 34 elements were determined for each site however, elements with significant changes were used as a proxy in this research. Table 4.1 provides a summary of the elements and ratios used in this study (Guyard *et al.* 2007; Mueller *et al.* 2009; Brown 2011; Burnett *et al.* 2011; Kylander *et al.* 2011; Lauterbach *et al.* 2011; Marshall *et al.* 2011; Arnaud *et al.* 2012; Aufgebauer *et al.* 2012; Corella *et al.* 2012; Elbert *et al.* 2013). The data was then processed by the in-built software of the machine and detailed information about the elemental composition of the sediment cores was retrieved in order to assess changes in the geochemistry of the cores.

**Table 4.1 Provides summary of the main elements and ratios used to interpret wetland changes**

Element or Elemental ratio	Proxy	References
<b>Magnetic susceptibility</b>	Magnetic evidence for reduced flows	Marshall <i>et al.</i> 2011
<b>Al</b>	Detrital input	Lauterbach <i>et al.</i> 2011
<b>Si</b>	Detrital input	Marshall <i>et al.</i> 2011
<b>P</b>	Nutrient enrichment	Corella <i>et al.</i> 2012
<b>S</b>	Soil derived S from leaching	Burnett <i>et al.</i> 2011
<b>K</b>	Detrital input	Aufgebauer <i>et al.</i> 2012
<b>Ca</b>	Evaporative concentration, dry conditions	Brown 2011
<b>Ti</b>	Detrital input	Elbert <i>et al.</i> 2013
<b>Mn</b>	Oxygenation of bottom waters or lower lake level	Kylander <i>et al.</i> 2011
<b>Fe</b>	Detrital inputs, redox conditions	Kylander <i>et al.</i> 2011
<b>Cu or Cu/Ti</b>	Metal pollution	Guyard <i>et al.</i> 2007
<b>Pb or Pb/Ti</b>	Metal pollution	Guyard <i>et al.</i> 2007
<b>Fe/Ti</b>	Grain size (increase = smaller grain-size)	Marshall <i>et al.</i> 2011
<b>K/Al</b>	Weathering	Burnett <i>et al.</i> 2011
<b>Si/Ti</b>	Increased biogenic silica (principally diatoms)	Brown 2011
<b>Fe/Mn</b>	Reducing conditions	Corella <i>et al.</i> 2012
<b>Ca/Fe</b>	Increased pedogenic input, dry conditions	Mueller <i>et al.</i> 2009

### 4.3.3 Stable Isotopes Analysis

Stable isotopes of carbon and nitrogen were processed using standard methods (Murase and Sugimoto 2001; Papanicolaou *et al.* 2003). Samples were stored in the freezer at -20°C and were dried in the freeze drier at Federation University (Figure 4.3). Thereupon, sediment samples were homogenised using an agate mortar and pestle. To ensure ultrafine consistency of the sediment, samples were sieved before analysis. Only a few samples were analysed for stable isotope analysis: 4 samples for Black Swamp, 6 for Taylors Lagoon, 5 for each of Green Swamp and Cockatoo Lagoon. These samples were pre-treated separately for carbon and nitrogen analysis at the ANSTO laboratory. For

carbon analysis between 1 and 5 g of dried sample was weighed, and acid treatment was performed at the ANSTO laboratory. For nitrogen analysis, each powdered sample was weighed on a microbalance (Figure 4.3) to ensure enough material (10 – 20  $\mu\text{g}$  per sample) was available for analysis. Once weighed, the samples were placed into tin capsules and analysed using a Thermo Fisher Delta V Isotope Ratio Mass Spectrometer (IRMS).

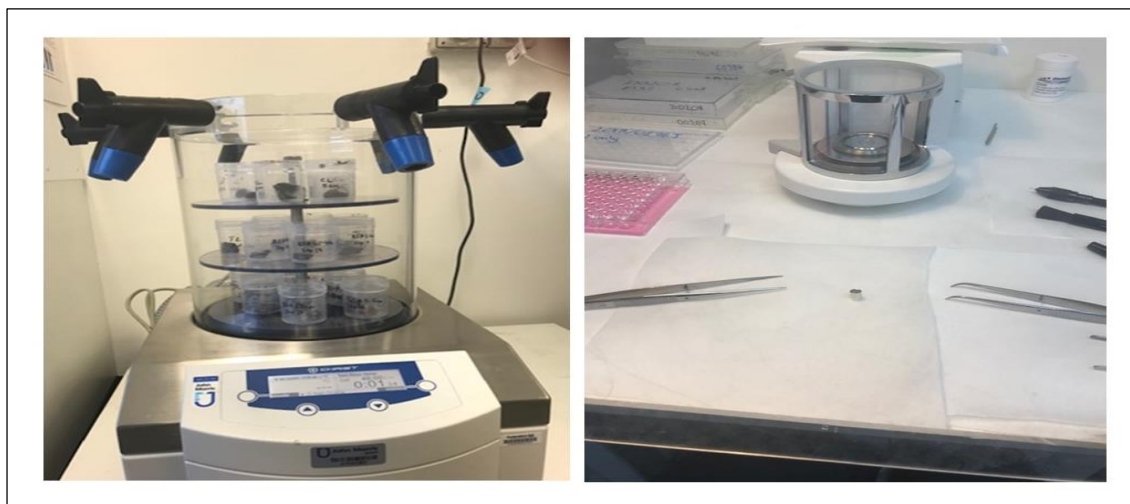


Figure 4.3 Freeze drying of sample (left) at Federation University; Sample weighing (right) for stable isotope analysis at ANSTO

Samples were run using an established on-line combustion, continuous-flow IRMS method. First of all, the dried and homogenised samples were weighed into tin capsules and introduced sequentially into an elemental analyser (Thermo Fisher Flash 2000 HT EA) using an autosampler. Each sample was then combusted into  $\text{CO}_2$  and  $\text{N}_2$  in a combustion furnace (silvered cobaltous oxide, quartz chips and quartz wool) at  $1020^\circ\text{C}$  before being transferred with a helium carrier gas (100 mL/min) into a reduction furnace (copper) at  $600^\circ\text{C}$ , where any excess nitrous oxides were converted into  $\text{N}_2$  and excess  $\text{O}_2$  is removed (Ohlsson and Wallmark 1999). The analyte gases were then passed through a water trap before the  $\text{CO}_2$  and  $\text{N}_2$  were separated by a Gas Chromatography column at  $40^\circ\text{C}$ . The gases were then transferred to a Thermo Fisher ConFlo IV and into a Thermo Fisher Delta V Plus Isotope Ratio Mass Spectrometer for  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  measurements. The  $\delta^{13}\text{C}$  ratios were obtained from the comparison of  $\text{CO}_2$  standard gas reference (99.996 %,  $\delta^{13}\text{CVPDB} = -6.317\text{‰}$ ), and all values stated are relative to this reference (Vienna Pee Dee Belemnite standard). Ratios of the  $\delta^{15}\text{N}$  isotope were derived by comparing a  $\text{N}_2$  standard gas reference (99.99 %  $\delta^{15}\text{NAIR} = -1.706\text{‰}$ ), with all values stated being relative to this reference.

The data is reported relative to the International Atomic Energy Agency (IAEA) secondary standards that have been certified relative to Vienna Pee Dee Belemnite (VPDB) for carbon and air for nitrogen. A two-point calibration was employed to normalise the data, using standards that bracket the samples being analysed. Two quality control references were also included in each run and samples were repeatedly analysed to obtain similar results of two replicates.

The data obtained from the analysis provided data on the Carbon: Nitrogen (C/N) ratio,  $\delta^{13}\text{C}$  ‰ and  $\delta^{15}\text{N}$  ‰ values. This analysis was pursued to determine the source of the organic matter in lake sediments (i.e., terrestrial or lacustrine derived). Lower C/N ratios (4-10) and  $\delta^{13}\text{C}$  ‰ values were used to determine autochthonous inputs while, allochthonous material to the wetland was revealed by higher C/N ratios (above 20) (see Figure 4.4; Lamb *et al.* 2006).

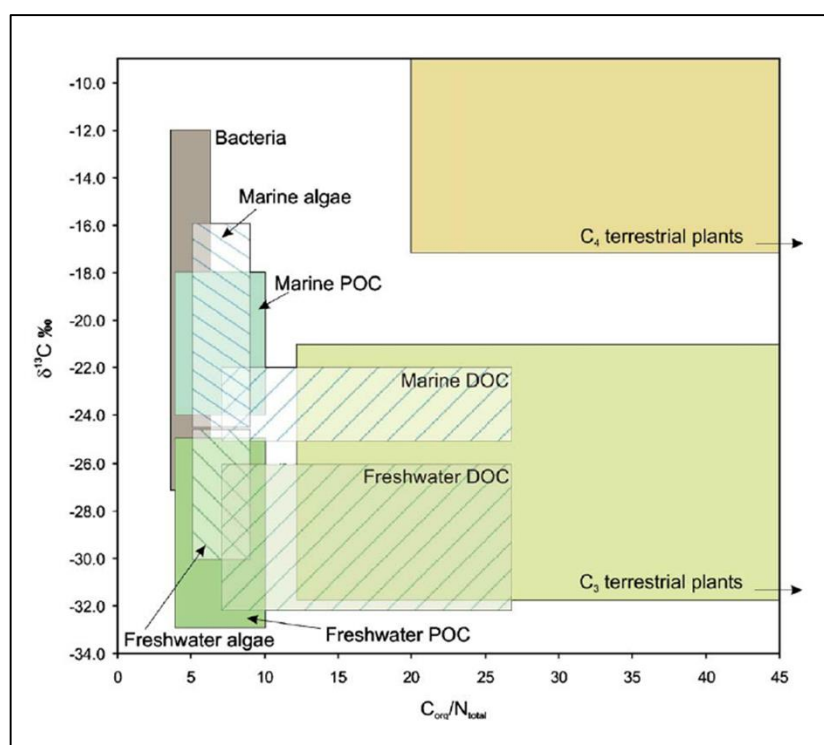


Figure 4.4 Comparison of  $\delta^{13}\text{C}$  ‰ and atomic C/N ratios with marine, freshwater algae and plant material source (Reprinted from: Lamb *et al.* 2006)

#### 4.3.4 Diatom sample preparation

Several methods exist for preparing and fixing diatoms onto microscope slides for enumeration, such as Hasle and Fryxell (1970); Battarbee and Charles (1987); Renberg (1990); Pienitz and Smol (1993). These differ primarily in the amount of original sample required, the materials and chemicals used, and the length of time it takes to prepare the sample. In general, though, all methods aim to isolate the siliceous diatom valves from



extraneous materials within the sediment with minimal damage or chemical dissolution occurring to the valves as a result. While all the aforementioned methods achieve this, those most commonly used, and those employed in this study are the methods of Renberg (1990) and Battarbee *et al.* (2001). Both employ the same general principle of  $\text{H}_2\text{O}_2$  and HCl digestion with the main differences between the two methods being the amount of original sample required and the laboratory equipment used.

Following the method of Battarbee *et al.* (2001), 1 cc of sediment was placed in beakers to which dilute hydrochloric acid (HCl) (10%) was added. These beakers were then placed onto a hotplate and warmed for three hours to remove carbonates (Figure 4.5a). The samples were then left to settle overnight before the supernatant was carefully poured off (Figure 4.5b). A series of three rinses with distilled water followed, with a minimum of six hours between each rinse, followed by the removal of the supernatant. After the final rinse, the samples were treated with 10% hydrogen peroxide ( $\text{H}_2\text{O}_2$ ) and warmed for three hours to remove organic matter before repeating the settling and rinsing process. Distilled water was added to the final sample until the opacity was deemed to be adequate to ensure an appropriate density of valves (Figure 4.5c).

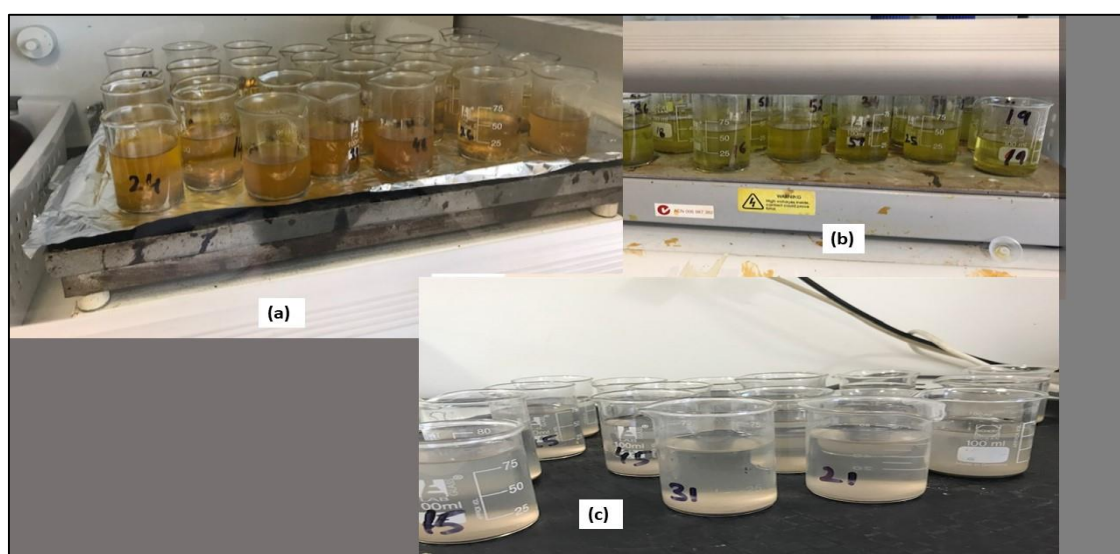


Figure 4.5 Diatom preparation steps (a, b) Removal of carbonates and organic matter on hot plate with dil. HCl and  $\text{H}_2\text{O}_2$  (c) Distilled water treatment

A pipette was used to place an aliquot of 800  $\mu\text{l}$  of the suspension onto a microscope coverslip that was then left to air dry overnight. Once dried, the coverslip was inverted onto a drop of Naphrax mounting medium on a warmed microscope slide. The slide was heated to remove toluene bubbles from the Naphrax before being placed on a cool surface where the coverslips were further pressed to remove all air bubbles and to place the

diatoms into a valve-view orientation. This method of mounting samples onto microscope slides was employed for all diatom samples used in this project.

#### **4.3.4.1 Diatom identification and enumeration**

For consistency, the same microscope was used for all samples from each core. Cells were counted using a Nikon Eclipse 80i microscope at 10x100 magnification and differential interference contrast. Preliminary diatom analysis was also carried out on samples taken every 5 cm to ascertain the presence of valves. Diatoms were then identified and enumerated from samples taken at 2 cm sample intervals. A number of taxonomic texts were used to aid in the identification of species. Diatom identification was based on photomicrographs from the following references; Krammer and Lange-Bertalot (1986); Krammer (1988); Krammer and Lange-Bertalot (1991); Krammer and Lange-Bertalot (2004); Gell *et al.* (1999) and Sonneman *et al.* (2000).

Different systems of nomenclature have evolved through the regular and somewhat independent taxonomic revisions of diatom taxa. Among those that employ taxonomic systems for ecology or paleoecology, of taxonomy itself, different systems are adopted. So, these may vary within and between continents, countries, regions and even laboratories. In part, this is because of the continued use of databases that were built before many recent taxonomic revisions. In general, there are two main systems of nomenclature adopted, one based on the works of Krammer and Lange-Bertalot (1986, 1988, 1991) and the other on the proposed modifications of genera by Round *et al.* (1990) and numerous further modifications. In this project, where possible the most recent names and authorities have been used. Taxonomic authorities and synonyms of the most common fossil species identified are presented in Appendix I.

Diatoms were extracted from 200 sediment samples and at least 200 diatom valves following Bate and Newall (1998) were identified. Where diatoms were sparse at least 3 transects were traversed and all entire valves counted. The relative abundance of each species was calculated as a proportion of the total, and species representing less than 5% of the relative abundance in any sample were removed from the diatom diagram to better reveal the changes of the more common taxa. All taxa were used in ordination analyses. Diatoms were grouped according to known community habitat preferences. The species information was displayed using the paleoecological program Tilia (Grimm 2011) and C2 (v. 1.7.7) (Juggins 2007). This enabled the visualisation of any trends present within the data over time.

## 4.4 Statistical analysis

In the present research, the aim of statistical analysis is to identify the periods in which a notable change in the wetland notable change has occurred (Smol *et al.* 2012). This can be accomplished by sample clustering that demonstrates the greatest similarity in diatom (Legendre and Birks 2012) or elemental composition. The methods used in the statistical analysis are; using ordination method to investigate the gradients of change and the clustering of adjacent samples using CONISS (Grimm 2011). Wherever possible, wetland change was compared with local and regional periods of change. The percentage of diatom abundance relative to the depth of the core and periods of change was visually explored using Tilia software (ver. 2.6.1, Springfield: Illinois State Museum Research and Collections Centre). All statistical analyses for Itrax data were undertaken using the statistical coding interface R (Venables and Smith 2009), with the exception of the stable isotope plots produced using C2 (Juggins 2007). In this study, the statistical analyses were undertaken using CANOCO software for Windows 4.5 (Ter Braak and Šmilauer 2002), RStudio (ver. 1.2.5042, Bristol: Network Theory), C2 software (ver. 1.7.7, Newcastle University, Newcastle upon Tyne; Juggins 2007) and Microsoft Excel (ver. 12.0, Microsoft Corporation).

There are different approaches such as ordination methods available to analyse ecological data that have been discussed extensively over the last two decades (Gerritsen 1995; Norris 1995; Van den Brink and Ter Braak 1998; De'ath 2002; Goslee and Urban 2007; Šmilauer and Lepš 2014). The selection of an appropriate ordination method is based on the type of data which could be linear or unimodal in nature (Šmilauer and Lepš 2014). Linear data has a tendency to be observed along a shorter gradient that exhibit high species diversity within each sample but reveal less comparisons and changes between samples. In contrast, unimodal data is observed along longer gradients, shows a low diversity of each sample (alpha-diversity), but determine significant changes in species over an environmental gradient (beta-diversity). For unimodal data, Detrended Correspondence Analysis (DCA) would be used, while for linear data, PCA would be more suitable (Šmilauer and Lepš 2014).

The data from the Itrax analysis and diatom species abundance were collated for statistical analysis. The dataset comprised values of elemental composition and abundance of diatom species with respect to depth of the core sequences. The screening of data was undertaken to understand the data distribution (i.e., normal vs non-normal) and to ensure that the collected data met the statistical tests. Moreover, the units of all the parameters were different in that diatoms were measured in percentages and Itrax

elements were in counts per second. Hence, there was a requirement for the majority of the data to be transformed in order to 'normalise' the data to ensure they were directly comparable by removing the artefact of unit of measurement (Itrax data). The data were either log transformed or square-root transformed to create a normal distribution.

Correspondence Analysis was applied to find association of diatoms and elements at different depths and an arch in the data was found. There are two main issues associated with CA: 1) the arch effect which is produced by unimodal species response curves and, 2) compression of the ends of the gradient. To overcome this arch issue, DCA was performed on the diatom species and Itrax elements data using CANOCO to determine the length of the respective gradient. Upon DCA, the gradient of axis 1 (the strongest axis) was recorded as <2. The gradient length measures beta diversity in community composition along with the independent gradients (ordination axis). If the value of the gradient is larger than 4.0, then unimodal methods (DCA, CA or CCA) can be used because a linear method would not be appropriate, since the data are too heterogeneous and the species would not show a linear response. However, in the present study, the gradient length of the strongest axis was less than 2 (this number was obtained by a DCA). Hence, PCA analysis (discussed in Chapter 5) was undertaken to remove the arch effect and to ensure that PCA 1 axis is reliable indicator of elements, species composition change and variability with depth due to length of the vector loadings. Further, to explore relationship between elemental composition and diatoms, the PCA was undertaken using CANOCO software. Bi-plots were produced from PCA analysis that locate data points in two-dimensional space relative to first and second principal components (Šmilauer and Lepš 2014) to explore relationships between various parameters based on the clustering of similar samples or species. This allows for the identification of the principal directions (gradients) of the variations in the species data in order to find the patterns in ecological datasets (Jolliffe and Cadima 2016). In the present case, a correlation matrix amongst the selected diatom species and elements was obtained to reveal significant correlations (values > 0.50).

#### **4.4.1 Tilia and C2 software analysis**

Raw counts of all species were entered into Microsoft Excel (v. 12.0) before being converted to proportions and transferred to the program Tilia (Grimm 1987; 2011). The program shows percentage abundance of taxon species proportional to the total number of diatoms. Stratigraphic diagrams were developed followed by the determination of core zonation using the Constrained Incremental Sum of Squares (CONISS) function within Tilia (Grimm 2011). CONISS cluster analysis helps to identify stratigraphic zones across

which the diatom flora changed markedly. The number of significant zones was determined on the basis of dominant diatom assemblages. The sample scores from the first and second principal components from the PCA are plotted to indicate periods of change in species composition (Legendre and Birks 2012). Further, the sample scores of PC1 and PC2 were recorded and plotted in the diatom and Itrax diagrams to demonstrate the timing of change in the diatom assemblage or elemental composition at that site over the study period (Legendre and Birks 2012).

The diagrams in the discussion (Chapter 6) are plotted against age to enable comparison with respect to time to enhance environmental inferences that further facilitate comparison with existing paleoenvironmental reconstructions for that particular region. Stable isotope data was analysed using C2 software to determine nitrogen and carbon concentrations in bulk organic sediments throughout the core. The multi-proxy analysis of the diatom and Itrax PCA curves were undertaken by plotting them together with the C:N ratio,  $\delta^{13}\text{C}$  ‰ and  $\delta^{15}\text{N}$  ‰ against time. Thereby, a visual rationale of the proxies with the wetland changes was explored and the drivers responsible for these changes to be further confirmed or contested.

## **4.5 Summary**

This study employed established techniques to analyse changes in multiple indicators down-core. Using standard methods in sediment preparation and analysis facilitates reproducibility of results and improved comparisons between sites. The diatom and  $^{210}\text{Pb}$  dating approaches have been employed to estimate age of the core and sediment accumulation rates. The use of Itrax and stable isotope analyses supplements these analyses allowing for a clearer interpretation of the causes of the documented changes through time. The next chapter presents result of the analysis of the change in ecological condition and water quality of Gunbower Wetlands.

## CHAPTER 5- RESULTS

### 5.1 Introduction

This study is based on the use of paleoecological proxies to reconstruct changes in Gunbower wetlands over the period of the last ~ 100 years. The main proxies used in this study were: diatoms, elemental composition,  $^{210}\text{Pb}$  chronology and stable isotopes of carbon and nitrogen. This chapter presents the results from analyses undertaken on these proxies for the four study sites that are located in Gunbower Forest. A brief introduction is provided in section 5.1 and then this chapter is divided into three sections (5.2) Forested wetlands: Black and Green Swamp; and (5.3) Dairyland wetlands: Cockatoo and Taylors Lagoon; and (5.4) Summary of results. Section 5.2 and 5.3 are further divided into sub-sections and provide results of Lead-210 ( $^{210}\text{Pb}$ ) dating, Itrax X-ray Fluorescence (XRF) scanning, stable isotope analysis and diatom analysis. Sediment accumulation rates ( $\text{g cm}^{-2}\text{yr}^{-1}$ ), Pb activity ( $\text{Bq/kg}$ ) and the chronology (age in years) of the cores are determined by  $^{210}\text{Pb}$  dating. The elemental data produced by Itrax are qualitative, and the results are presented as counts per second (cps) (Croudace *et al.* 2006; Davies *et al.* 2015) while stable isotopes (C and N) as per mil (‰) and diatom species are presented as percentage abundance (%). In addition, planktonic to benthic ratios were calculated for all the sites and compared with the stable isotope results. Throughout this chapter PCA diagrams are presented for diatoms and Itrax elements (using CANOCO) and stratigraphic diagrams of Itrax elements (using RStudio) and their variations are discussed based on their clustering.

### 5.2 Forested wetlands

The Black and Green Swamp Wetlands Complex is located in the Gunbower forest (see Chapter-3). Sediment sequences were retrieved from both sites in October 2017. The length of sediment cores extracted from Black and Green Swamp were 84 cm and 86 cm respectively. As these were taken with a d-section coring device, it is assumed that there was no compaction of the sediment. The depths represent the total length of accumulated sediment (less 10 cm of the corer nose). The results of the different analyses for these sediment sequences are described in the subsequent sections.

## 5.2.1 Black Swamp

### 5.2.1.1 Lead-210 dating

As discussed in Chapter 4, determining the chronology of the Black Swamp Core was not straightforward, but was determined in two stages (see Chapter 4). Eleven  $^{210}\text{Pb}$  dates were obtained from the Black Swamp by selecting samples on the basis of Itrax scanning results and visual inspection of the sediment core. The CFCS (Constant Flux Constant Sedimentation) and CRS (Constant Rate of Supply) core chronology models were applied to the  $^{210}\text{Pb}$  data to determine the mass accumulation rates and ages of the sediment core. However, the ages in between the selected samples were calculated using CFCS model in MS-Excel by an extrapolation method using the equation  $y=1.014x-2.2061$ . The results are presented in the Appendix III, and graphical presentation of results are presented in Figure (5.1).

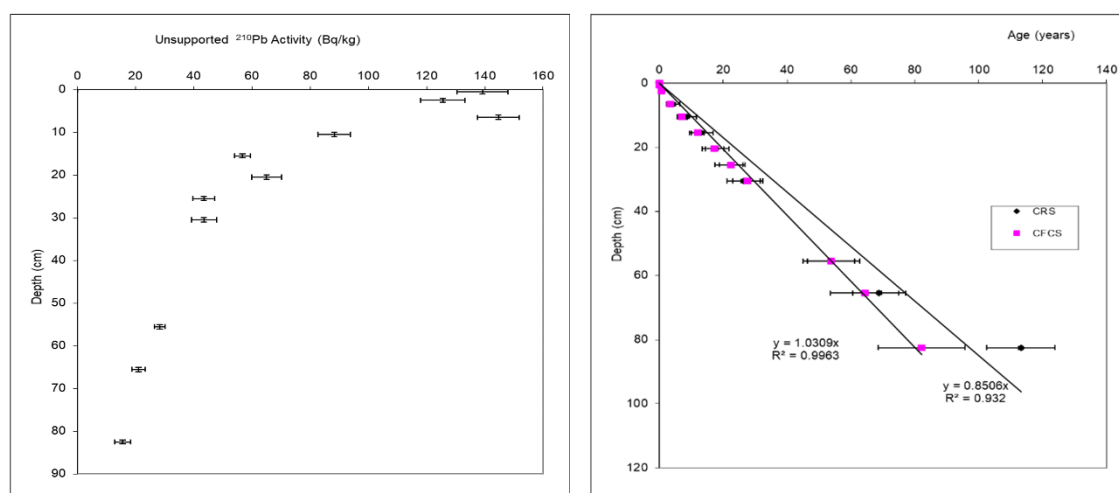


Figure 5.1 Unsupported  $^{210}\text{Pb}$  activity versus depth (left-hand panels) and CRS and CFCS calculated ages by depth (right-hand panels) for Black Swamp. Data show the mean  $\pm$  s.d.

Lead-210 activities were measured from 11 core samples each of 1 cm thickness. The results showed a declining trend of the  $^{210}\text{Pb}$  activity over time as is anticipated for a decay series. The background levels of  $^{210}\text{Pb}$  were 15 Bq/kg at 82 cm depth. However, the  $^{210}\text{Pb}$  activity between 0 and 6 cm did not show a decreasing trend which suggested the top 3 cm of sediments were accumulated very rapidly or were poorly consolidated. In contrast, below 6 cm, fine sediments were found to be well preserved, where the sediment records were chronologically ordered, with no evidence of sediment mixing.

The CFCS  $^{210}\text{Pb}$  dating model assumes a constant rate of sediment mass accumulation whereas the CRS model assumes a constant rate of  $^{210}\text{Pb}$  supply and allows for the determination of varying mass accumulation rates (Appleby and Oldfieldz 1983). The

CFCS model estimated a sediment mass accumulation rate of  $0.287 \text{ g cm}^2 \text{ yr}^{-1}$  ( $r^2 = 0.8393$ ). The CRS-based sediment mass accumulation rate ranged from  $0.2 - 0.38 \text{ g cm}^2 \text{ yr}^{-1}$  between 0-30 cm to  $0.21 - 0.06 \text{ g cm}^2 \text{ yr}^{-1}$  between 55-83 cm depth layers. The CFCS and CRS age depth models were found largely consistent in age estimation. The CFCS model however, estimated the age of the deepest sample (82 cm) to be  $82 \pm 13$  years old, dated to circa  $1936 \pm 13$  CE, while the CRS-based age for this deepest layer was  $113 \pm 11$  years and dated to circa  $1904 \pm 11$  CE (Figure 5.1).

### **5.2.1.2 Itrax scanning**

The Itrax X-ray Fluorescence (XRF) core scanning results for Black Swamp are presented in Figures 5.2 and 5.3. There were noticeable differences in magnetic susceptibility (MSE), and elemental composition of Pb (Lead), Ca (Calcium), Aluminium (Al), Potassium (K), Fe (Iron), Manganese (Mn), Copper (Cu), S (Sulphur) and P (Phosphorus), and Si (biogenic Silica).

Magnetic susceptibility (MSE) data did not vary greatly but values were comparatively high between 70-55 cm (Zone 4) and 25-15 cm (Zone 2) while they decreased gradually towards the top of core (Figure 5.2) suggesting the influence of erosion in the lower sections of the core. Indicators of erosional (detrital inputs), which include the lithogenic elements, Al, Si, K, Ti, and Fe, were commonly recorded. These elements are geochemically stable, usually hosted by resistant minerals, and are conservative to most geochemical environments (Boës *et al.* 2011) For instance, a geochemically stable lithogenic element, Titanium (Ti) is regarded an erosion indicator suggesting that catchment erosion could be increasingly a source of sediments. They are often associated with particular grain characteristics. For example, Ti, Fe, and K are linked with fine grained materials and clay particles (Kylander *et al.* 2011) while Si is with coarse silt and sand. Detrital enrichment was evident in Zones 2,3 and 4 (70-12 cm) compared to Zone 1 (11-0 cm; Figure 5.2). The lithogenic elements increased gradually upwards followed by a rapid decline after 10 cm. Such enrichments could be related to increased catchment dynamics (Figure 5.2). The ratio of K/Al can be used to investigate the weathering regime (Brown 2011; Arnaud *et al.* 2012) and Ca/Fe for biogenic carbonate and pedogenic input. However, in zone 1, these ratios showed little variation except for minor peaks at 0-2 cm and 14-15 cm respectively (Figure 5.3). The ratio of Fe/Ti were generally consistent in this core but comparatively high in Zone 4; the lower part of the core suggesting reductions in lake level or enhanced influx.

Sulphur (S) is a proxy for organic matter (Croudace *et al.* 2006). The values of S remained stable in all the zones indicating organic activity throughout the core that could be due to



allochthonous organic input at the surface of water. The values of Fe were high between Zone 4 and Zone 3 (70-35) cm and inform on reducing conditions. They declined from Zone 2 to Zone 1 suggesting oxygenated conditions. The values of Mn were observed only at a few depths. High P values are mainly seen between 60-15 cm indicating erosion as a source of P. The metals, Cu, Pb and Ni showed a similar upward declining trend but only until 52 cm and the values then gradually increased towards the top of the core indicating recent increases in anthropogenic pollution.

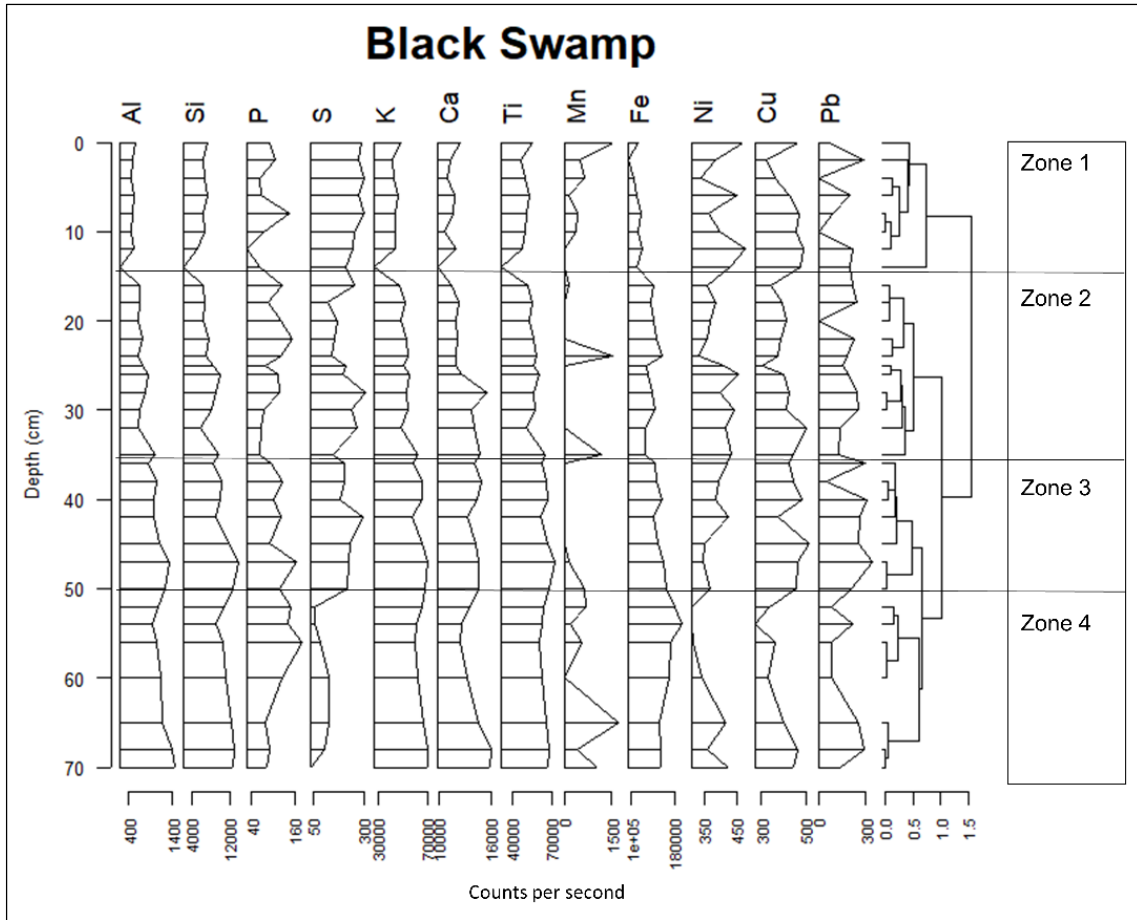


Figure 5.2 Stratigraphic plot of Itrax elements for Black Swamp core. CONISS output separating samples into zones on the basis of variations in sediment sources across the samples

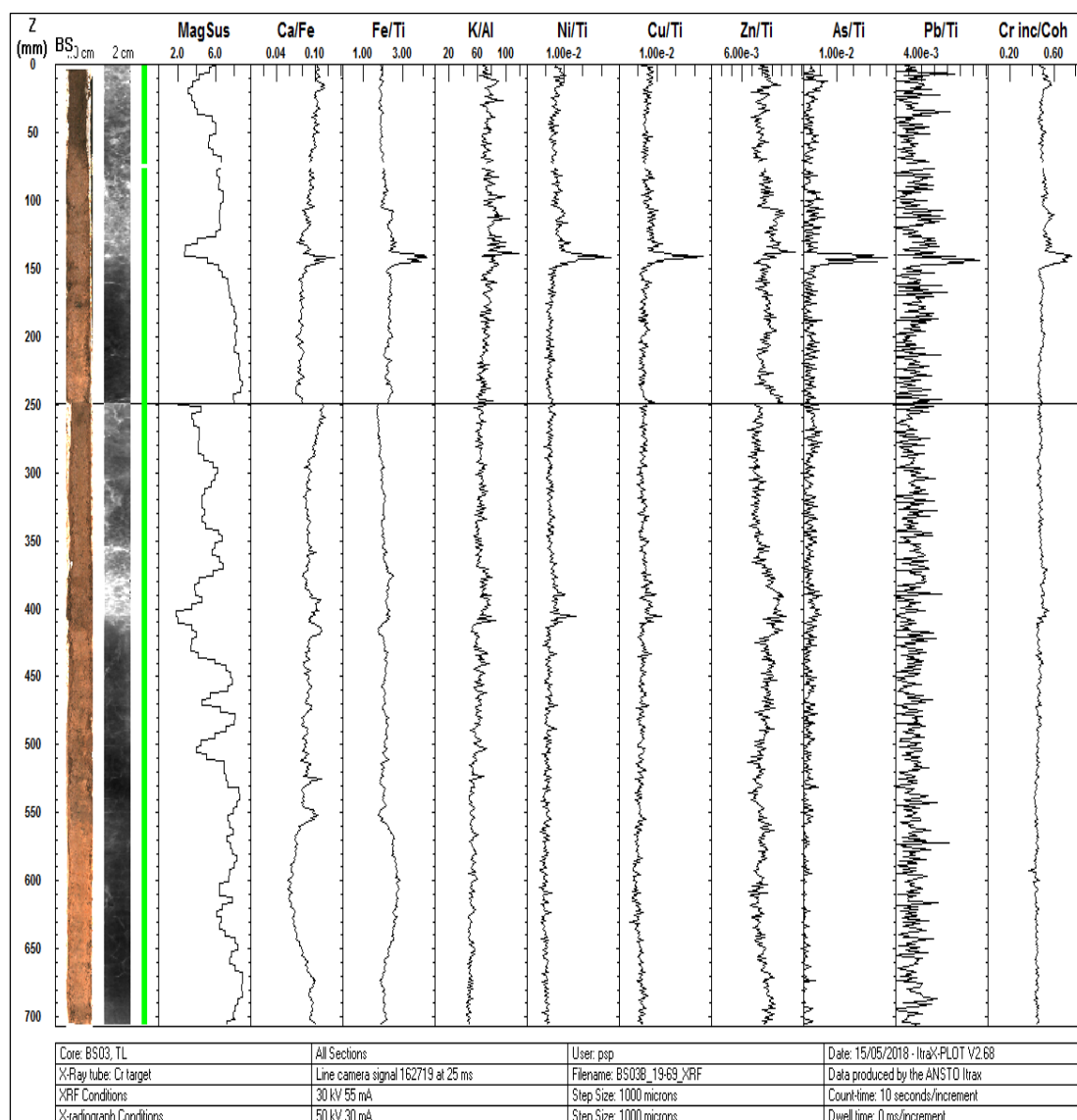


Figure 5.3 Elemental ratio plot for Black Swamp. The elements: Fe, K, Ni, Cu, Zn, As, Pb are normalised with Ti.

The correlation matrix of elements is presented in Table 5.1. The values highlighted in bold show high positive, or negative, correlation values between the elements ( $r \geq 0.5$  or  $r \leq -0.5$ ). According to the correlation matrix, the detrital elements are strongly correlated with values of correlation coefficient  $r > 0.5$  whereas  $r$  values of Ni, Cu or Pb are less than 0.5 showing insignificant relationships with each other. However, a significant relationship is noticed between Fe and P indicating their detrital origin.

**Table 5.1 Correlation coefficients among the elements obtained for Black Swamp core by the Itrax XRF Core Scanner**

	Mg	Al	Si	P	S	K	Ca	Ti	Mn	Fe	Ni	Cu	Pb
<b>Mg</b>	1.00												
<b>Al</b>	0.19	1.00											
<b>Si</b>	0.14	<b>0.96</b>	1.00										
<b>P</b>	0.29	0.34	0.36	1.00									
<b>S</b>	-0.03	-0.56	-0.44	-0.26	1.00								
<b>K</b>	0.12	<b>0.96</b>	<b>0.94</b>	0.36	-0.54	1.00							
<b>Ca</b>	0.09	<b>0.83</b>	<b>0.78</b>	0.08	-0.30	<b>0.84</b>	1.00						
<b>Ti</b>	0.10	<b>0.91</b>	<b>0.93</b>	0.38	-0.44	<b>0.98</b>	<b>0.80</b>	1.00					
<b>Mn</b>	-0.11	0.16	0.21	-0.07	-0.25	0.19	0.12	0.19	1.00				
<b>Fe</b>	0.18	<b>0.71</b>	<b>0.64</b>	<b>0.59</b>	-0.71	<b>0.76</b>	<b>0.50</b>	<b>0.70</b>	0.01	1.00			
<b>Ni</b>	-0.06	-0.28	-0.27	-0.68	0.38	-0.29	0.02	-0.27	-0.01	-0.59	1.00		
<b>Cu</b>	-0.17	0.02	0.00	-0.37	0.32	0.02	0.30	0.01	-0.03	-0.28	0.42	1.00	
<b>Pb</b>	0.06	0.26	0.23	0.00	0.02	0.25	0.33	0.25	-0.20	0.16	0.13	0.10	1.00

In order to show the relationships among the elements as measured on the XRF scanner, multivariate ordination, in the form of a Principal Components Analysis (PCA), was performed (Figure 5.4). The PCA results showed the variation in the elemental components is mostly described by the two independent axes; Axis 1 and 2. The first axis (PC1) is closely aligned with the elements Fe, Si, Al, Ti, Ni, Mg, K, Mn, and S and, combined, which together accounted for 46 % of the total variance (Figure 5.4). The second axis (PC2) closely aligned with the elements Ni, Cu, P, Mg, and Pb and together accounted for a further 17% of the total variation (Figure 5.4, Table 5.2). PCA axis 1 revealed a clustering of the detrital and pedogenic elements such as, Ti, Fe, Si, Al, K and Ca, while the organic matter indicator S, showed a negative correlation with these detrital elements suggesting the input of S from dead remains (Ivanov 1981).

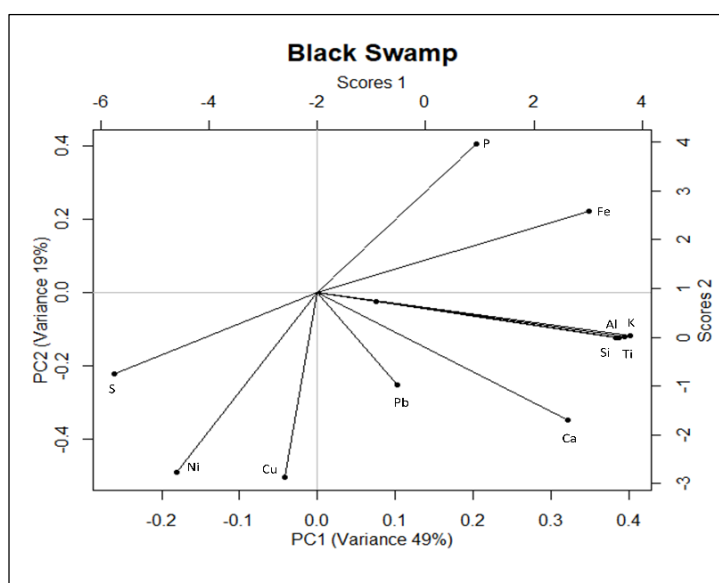


Figure 5.4 Principal component analysis of Black Swamp elements. PC1 and PC2 scores show 49% and 19% variance respectively.

The PC2 aligned with P, Cu, Pb and Ni but the relationship was not significant, possibly due to the natural location of Black Swamp. Under natural conditions, a redox transition can occur and the biogeochemical characters may vary at the mm scale in sediments (Ratering and Schnell 2001). However, the elemental profile varies significantly when the external environment changes (such as change in pH, pollutants, climate etc.; Kylander *et al.* 2013). These results suggest allochthonous material supply to Black Swamp is important to understand the source of sediment inputs to the wetland. It confirms that the detrital input is a mixture of the carbonate and siliciclastic components as a positive correlation was seen between Ca and Si elements which is evident from the simultaneous increase of these elements from the bottom to the top of the core.

**Table 5.2 Principal component analyses for Black Swamp core. The factor loads for each variable for the two main axes are given in the table**

	PC1	PC2
<b>Mg</b>	0.07917	0.134081
<b>Al</b>	0.393479	-0.12095
<b>Si</b>	0.38193	-0.12692
<b>P</b>	0.207319	0.414201
<b>S</b>	-0.25927	-0.20317
<b>K</b>	0.400187	-0.12328
<b>Ca</b>	0.319223	-0.34576
<b>Ti</b>	0.385922	-0.12848
<b>Mn</b>	0.073332	-0.0437
<b>Fe</b>	0.34849	0.216033
<b>Ni</b>	-0.18028	-0.47772
<b>Cu</b>	-0.04482	-0.50301
<b>Pb</b>	0.102524	-0.23951

### 5.2.1.3 Stable isotopes

Four samples from the Black Swamp core were analysed for stable isotopes using Isotope Ratio Mass Spectrometry at ANSTO. Pre-treatment of these samples is discussed in Chapter 4. The samples were analysed to determine  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  variations in the bulk sediments. Moreover, C%, N%, C/N and P/B ratios were also calculated. The results of the analysis are shown in Figure 5.5. The dynamics of carbon and nitrogen in Black Swamp is revealed by the response of  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  values that the  $\delta^{13}\text{C}$  values consistently declined, the  $\delta^{15}\text{N}$  values fluctuated over time (Figure 5.5).

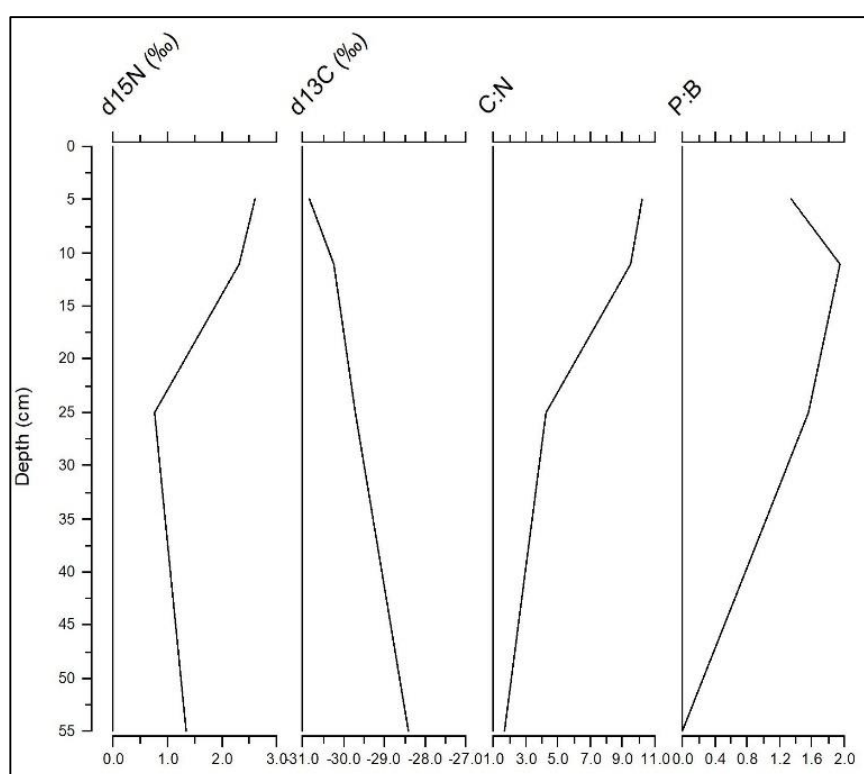


Figure 5.5 Variations in  $\delta^{15}\text{N}$ ,  $\delta^{13}\text{C}$ , C/N and P/B with depth of Black Swamp Core. Isotopes are measured in per mille units (‰)

The  $\delta^{13}\text{C}$  values for discrete samples were between -28.4‰ and -30.8‰ from 55 cm to 5 cm of the core. A decrease of 2.4‰ in  $\delta^{13}\text{C}$  was noted from the bottom to the top of the core between 55 and 5 cm. After the 11 cm depth, the  $\delta^{13}\text{C}$  value rapidly declined to the lowest values of the entire profile (-30.8‰). However, the C/N ratios were lower at the bottom depth of core (below 55 cm) and greater at the core top (5 cm) reflecting a recent increase in nitrogen input to the wetland. These inferences of  $\delta^{13}\text{C}$  values are further supported by P/B ratios which are high for surface sediments indicating more plankton diatoms.

#### 5.2.1.4 Diatom analysis

The results of the diatom analysis from the Black Swamp core are illustrated in Figure 5.7. The diagram was separated into three zones (BS1, BS2 and BS3) based on the most significant observed changes in diatom assemblages using the CONISS function in the Tilia software (Grimm, 2011). Zone BS1 is further subdivided into BS1a and BS2b. Diatom species were classified in accordance with their known habitat preferences, i.e., being either, planktonic, aerophilous, benthic, epiphytic or facultative planktonic. These zones are discussed below in chronological order.

##### Zone BS3 (85-56cm)

This zone was dominated by the benthic forms *Pinnularia* spp., *Stauroneis phoenicenteron*, *Eunotia serpentina* and *Craticula cuspidata* with limited numbers of plankton (mostly *Aulacoseira italica*). In this zone, the benthic species represented 74% of species suggesting the prevalence of clear water conditions allowing for these photosynthetic algae to proliferate at the sediment-water interface. At 80 cm the planktonic *Aulacoseira italica* showed a peak of 32%, however it declined to 7% at 56 cm. Epiphytes and facultative planktonic taxa were uncommon contributing an average of 6.3% and 3.3% respectively to the total valve count. Epiphytes were increasingly observed through the zone before increasing in BS2. Aerophilous species were absent. The high numbers of *Pinnularia* spp., and absence of *Cocconeis placentula* and *Epithemia* spp., suggest this wetland tended towards acid, rather than alkaline, conditions at this time.

##### Zone BS2 (56-35cm)

Throughout this zone benthic taxa (*Stauroneis phoenicenteron* and *Pinnularia* spp.) and the epiphyte *Gomphonema* spp. were predominant. *Stauroneis phoenicenteron* showed a peak of 43% at 42 cm while *Stauroneis anceps* had low values. *Pinnularia* spp. gradually declined upwards from 56 to 35 cm of the zone. *Eunotia serpentina*, *Rhopalodia gibba*, *Lemnicola hungarica*, *Neidium ampliatum* and *Craticula cuspidata* were recorded in low numbers. The nutrient indicator *Nitzschia palea* first appeared at 40 cm. The planktonic form *Aulacoseira italica* declined through this zone whereas the abundance of *Aulacoseira granulata* sspp. increased to a peak of 28% at 36 cm. The epiphytes *Gomphonema* spp. accounted for an average of 26% of all taxa (between 56-35 cm) reaching a peak of 40% at 38 cm and 60% at 55 cm. The aerophilous form *Sellaphora pupula* appeared in this zone but was in low abundance.

### Zone BS1b (35-21cm)

In this zone epiphytes were common. A noticeable peak of epiphytes at 27 cm in Zone BS1b, and high abundance of *Cocconeis placentula* were observed, but they declined upwards to ~ 10% at 20 cm and decreased further in zone BS1a. The river plankton *Aulacoseira granulata* spp. showed a gradual increase from 36 cm (3%) to 20 cm where it reached 52% of the count while *Aulacoseira italica* and *Aulacoseira alpigena* showed lower percentage values. Generally, above 30 cm, benthic diatoms were less abundant being replaced mostly by epiphytic and planktonic forms. *Lemnicola hungarica*, an epiphyte on floating plants (e.g., *Lemna*, *Azolla*) was persistent and increased in abundance to the top of zone BS1a. The nutrient indicators *Cyclotella meneghiniana* and *Nitzschia palea* were evident in several samples while the acidophilous *Pinnularia* spp. and *Eunotia* spp. were rare.

### Zone BS1a (21-0cm)

This zone was characterized by the dominance of planktonic species (47%), mainly of *Aulacoseira granulata* spp. *Cyclotella meneghiniana* showed peaks between 15-20 cm and *Nitzschia palea* increased upwards from 21 cm to the top of the core. *Lemnicola hungarica* gradually increased, reaching a peak of 14% at 2 cm. Benthic diatoms represented, on average, ~ 26% of all taxa. Aerophilous forms were common in this zone with *Diadesmis confervaceae* and *Sellaphora pupula* occurring with average abundance values of 8% and 9% respectively. Epiphytic forms were uncommon.

PC1 accounted for 42% of the variation in diatom species composition throughout the core. Diatom results were plotted with PC1 scores, and demonstrated a transition from benthic and epiphytic diatoms with negative scores to planktonic, facultative planktonic and aerophilous diatoms with positive PC1 scores (Figure 5.6). The main shifts in PC1 scores corresponded to the changes in distinct zones identified by the cluster analysis (Figure 5.7). Zone BS3 showed the lowest PCA 1 sample scores which were less than 0 (between -1.3 and -0.84) while the scores of the zone BS2 samples were still negative but higher as compared to the zone BS3 (between -1.1 and -0.1). This showed an ongoing shift in diatom assemblages between zones. In contrast, BS1a and BS1b zones showed high and positive score values of the PCA 1 ranging between 0.3 and 1.51.

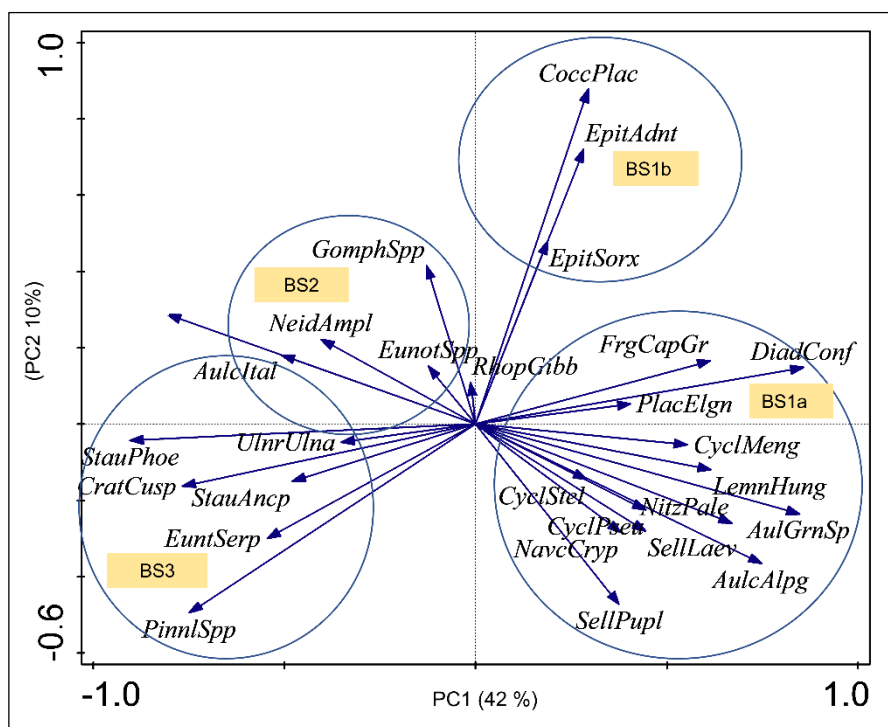


Figure 5.6 PCA for the diatom results and clustering of diatoms with zones for Black Swamp. Diatom species are shown by arrows and are labelled using species abbreviations.



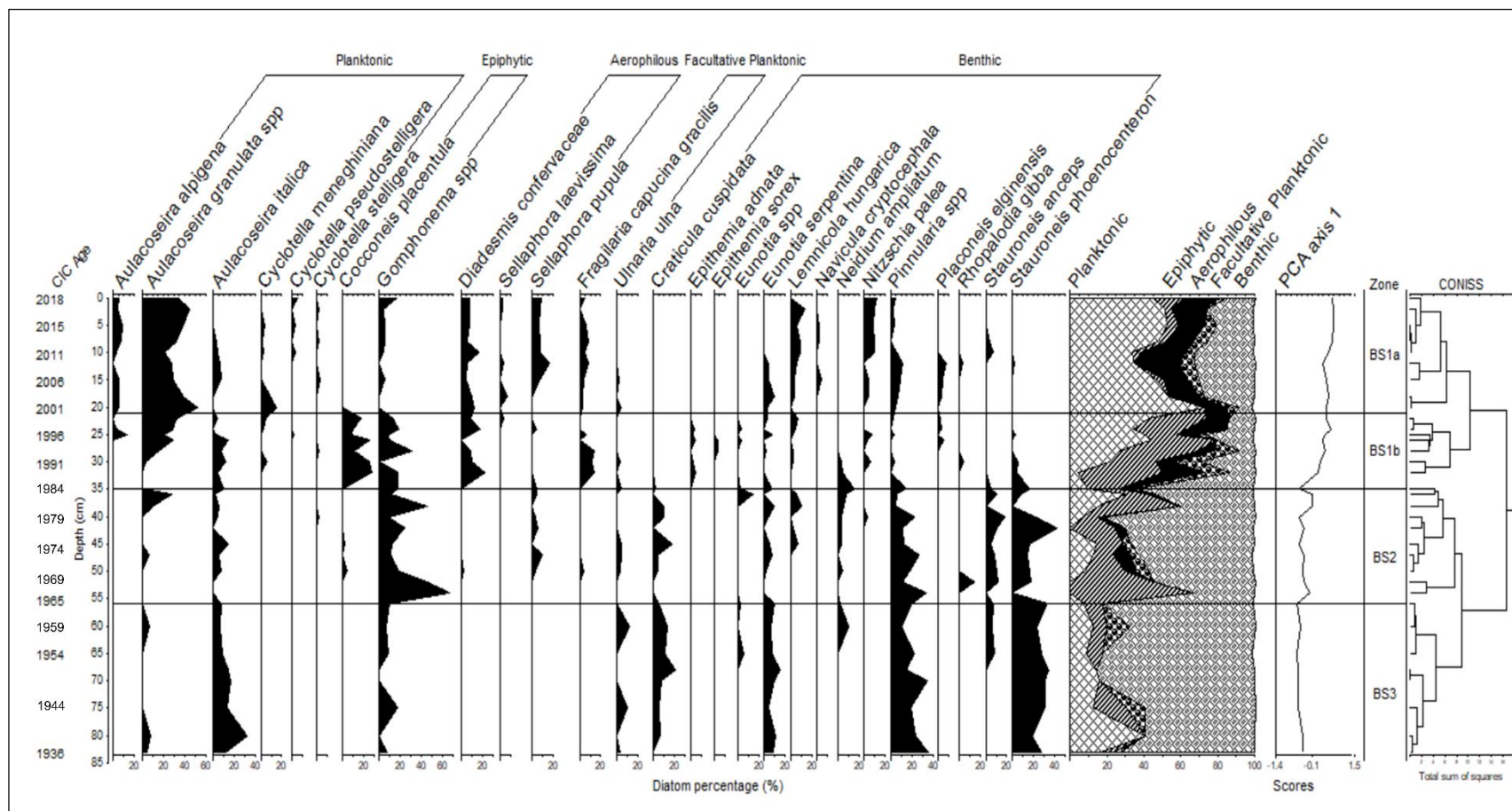


Figure 5.7 Diatom stratigraphic diagram of the relative abundance of diatoms (percentage %) of Black Swamp based on CONISS. Species >5% are shown and are grouped alphabetically according to their habitat preference.

## 5.2.2 Green Swamp

### 5.2.2.1 Lead-210 dating

The Green Swamp core samples were analysed for  $^{210}\text{Pb}$  dating in two stages as discussed in Chapter 4). The CFCS  $^{210}\text{Pb}$  dating model used to calculate the sediment core chronology is shown in the Appendix III. The results of this analysis are illustrated in Figure 5.8.

Eight samples, each of 1 cm depth, were analysed to determine the  $^{210}\text{Pb}$  activity in the core. Unsupported  $^{210}\text{Pb}$  activities showed declining values towards the bottom of the core but only between 0 and 31 cm depth. The CRS  $^{210}\text{Pb}$  dating model was not used to calculate the sediment core chronology because the activity had not reached the background level at the bottom of the core (at 75 cm). Hence, only the CFCS  $^{210}\text{Pb}$  dating model was used being the most suitable model to calculate the age of the core. The estimated sediment accumulation rate between 0-31 cm is  $0.141 \text{ g cm}^{-2} \text{ yr}^{-1}$  ( $r^2 = 0.6333$ ). An age of  $65 \pm 25$  years was estimated for the depth of 30-31 cm which means the core, at this depth, was dated back to  $1953 \pm 25$  CE (Figure 5.8). Below 31 cm the age of the core samples was calculated using the CFCS model in MS-excel by an extrapolation method using the equation  $y = 2.0169x + 0.1918$ . Hence, as per the extrapolated age of the deepest sample at 75 cm was dated back to 1867 CE but this date needs to be regarded with some uncertainty.

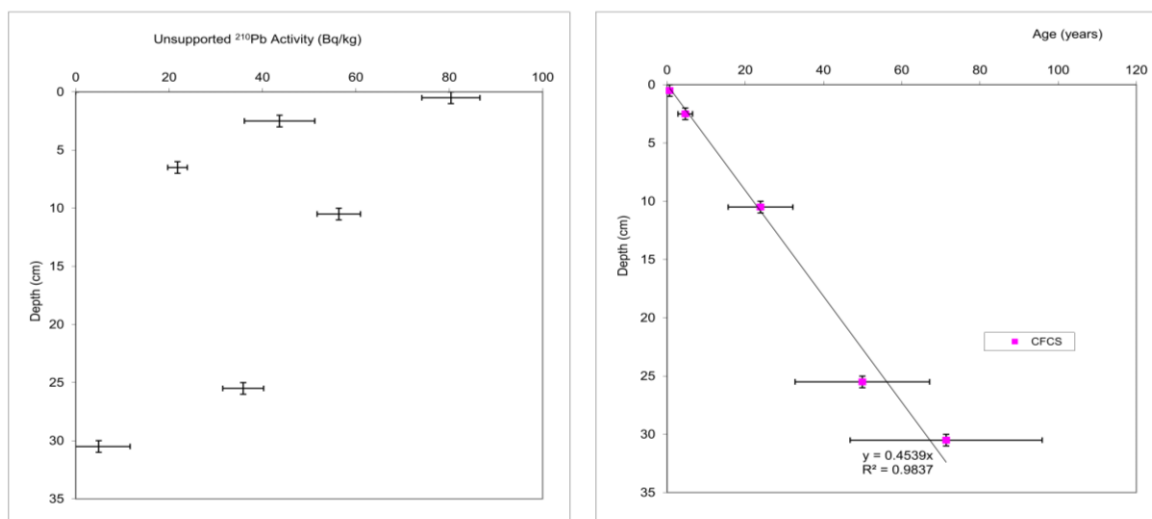


Figure 5.8 Unsupported  $^{210}\text{Pb}$  activity versus depth (left-hand panels) and CRS and CFCS calculated ages by depth (right-hand panels) for Green Swamp. Data show the mean  $\pm$  s.d.

### 5.2.2.2 Itrax Scanning

The Itrax results for the Green Swamp core were very similar to those from the Black Swamp (5.9 and 5.10). Magnetic susceptibility values did not vary greatly but increased gradually to the top (5.10). The detrital elements Ti, Al, K, Fe and Si showed higher values in Zone 3 (70-50 cm) and declined slightly towards the top. Lower values of detrital elements revealed reduced input of fine clay sediments from the catchment in recent times. Phosphorous, Pb, Ni and Cu were seen throughout the core while Fe and Mn declined in Zone 2a through to Zone 1.

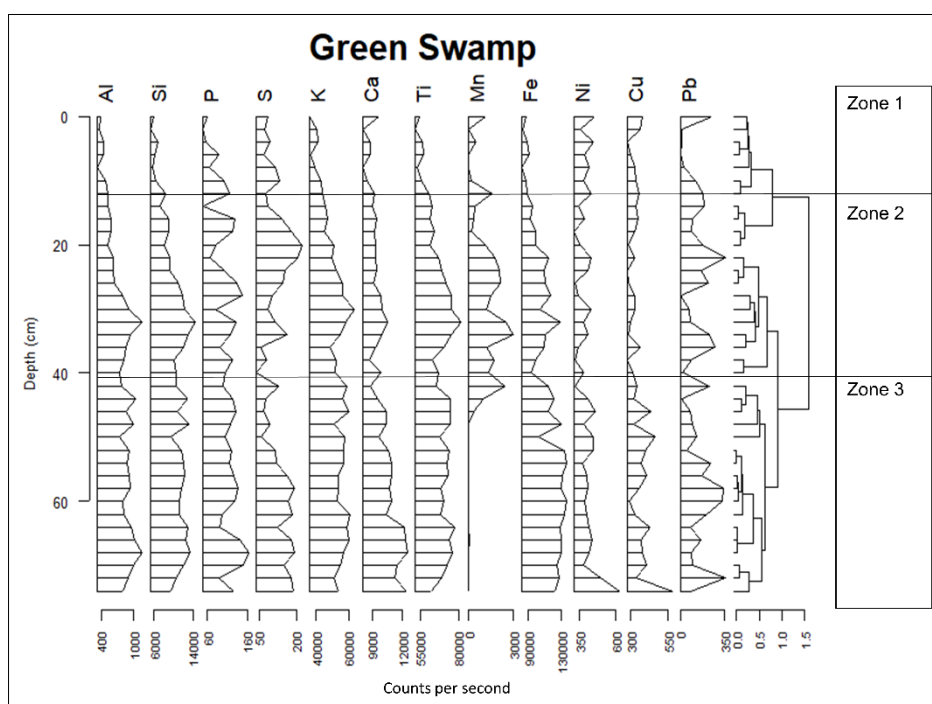


Figure 5.9 Stratigraphic plot of Itrax elements for Green Swamp core. CONISS output separating samples into zones on the basis of variations in sediment sources across the samples

The Itrax ratio results (Figure 5.10) showed some variability in trends throughout all profiles. The ratio of Si/Ti, which indicates biological productivity (Brown 2011), was relatively high from 87.5-35 cm and then slightly declined into 35-0 cm (Figure 5.9). Furthermore, pedogenic input was consistent throughout the core length as indicated by Ca/Fe (Elbert *et al.* 2013) and K/Al ratios (Burnett *et al.* 2011). The Fe/Ti ratios, used as indicators of grain-size fluctuations from allochthonous material (Marshall *et al.* 2011), were high, revealing smaller grain sizes and enhanced erosion influx, between 87.5-70 cm (0-37cm) relative to the top of the core (0-37 cm). Ratios of Pb/Ti, Cu/Ti and As/Ti showed higher values at the top and so reflect anthropogenic impacts and contamination.

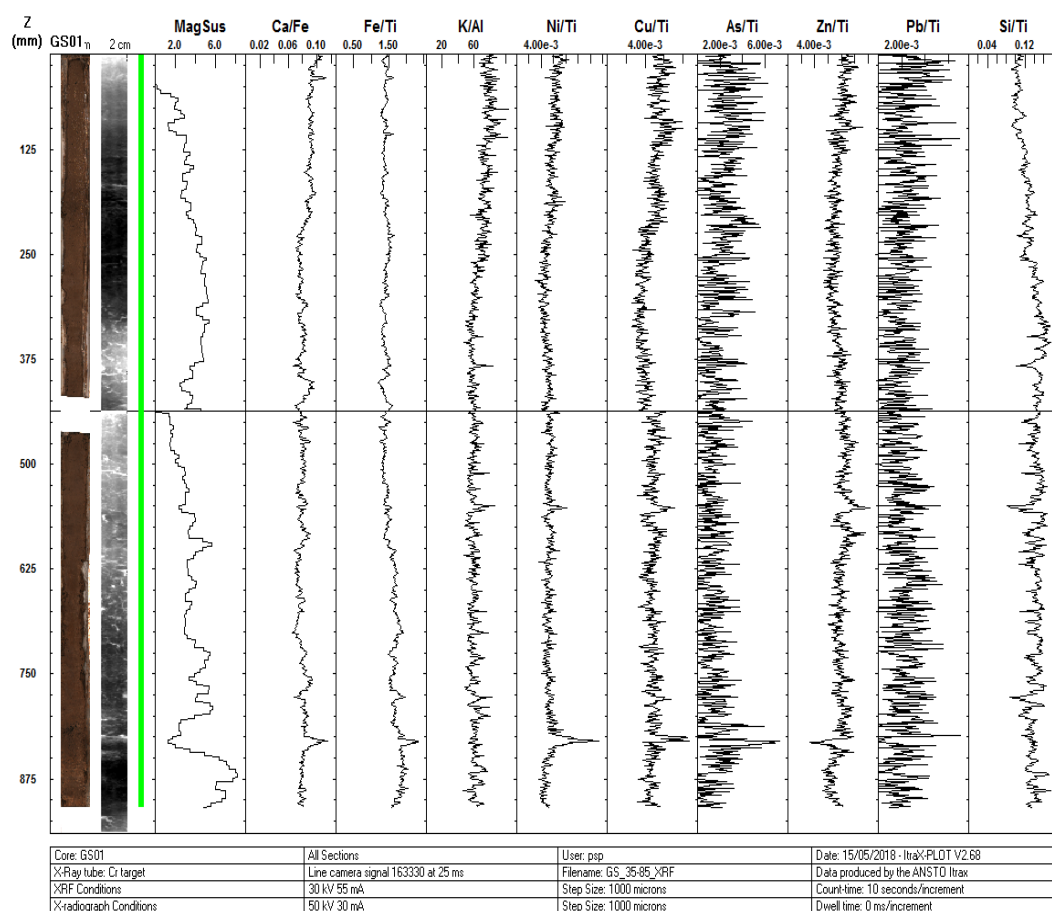


Figure 5.10 Elemental ratio plot for Green Swamp. The elements: Fe, K, Ni, Cu, Zn, As, Pb and Si are normalised with Ti.

The correlation matrix of elements (Table 5.3) revealed the relationship among elements with significant values highlighted in bold ( $r \geq 0.5$  or  $r \leq -0.5$ ). Accordingly, S, Mn and Pb, did not reveal any strong correlation with other elements, as they most likely originated from redox processes (Eusterhues *et al.* 2005). A strong correlation was noticed between detrital elements and P which is evident from its pattern from the stratigraphic diagram, suggesting input of P on clays derived from catchment erosion. However, Cu and Ni showed significant correlation with Ca maybe an indication of redox soil processes under reducing conditions.

A PCA was carried out using 12 variables to objectively describe the main variance in the Itrax data using a restricted number of factors. This was done to confirm the patterns observed previously using correlation analyses (Table 5.3). The PCA results showed the variation in the elemental components were mostly described by the two independent axes; axis 1 and 2. The first two axes alone accounted for 66% of the total variance. The first axis represented 48% of the total variance and was correlated with detrital and pedogenic elements including Fe, K, Ti, Si, Ca and some redox elements such as Cu, Ni

and Fe. The second axis accounted for about 18.51% of the total variance. Ni and Cu can be seen at the positive end and Mn at the negative end (Figure 5.11, Table 5.4).

**Table 5.3 Correlation coefficients among the elements obtained for Green Swamp core by the Itrax XRF Core Scanner**

	Al	Si	P	S	K	Ca	Ti	Mn	Fe	Ni	Cu	Pb
Al	1.00											
Si	0.96	1.00										
P	0.67	0.66	1.00									
S	0.19	0.18	0.27	1.00								
K	0.81	0.83	0.55	0.23	1.00							
Ca	0.73	0.65	0.55	0.44	0.58	1.00						
Ti	0.90	0.95	0.61	0.15	0.84	0.63	1.00					
Mn	0.03	0.17	0.00	-0.09	0.14	-0.34	0.18	1.00				
Fe	0.83	0.82	0.57	0.50	0.74	0.78	0.80	-0.18	1.00			
Ni	0.11	-0.03	-0.02	0.18	0.08	0.50	-0.01	-0.29	0.17	1.00		
Cu	0.20	0.07	0.19	0.03	0.21	0.53	0.07	-0.47	0.25	0.63	1.00	
Pb	0.08	0.13	-0.06	0.47	0.10	0.15	0.10	0.15	0.32	0.01	-0.09	1.00

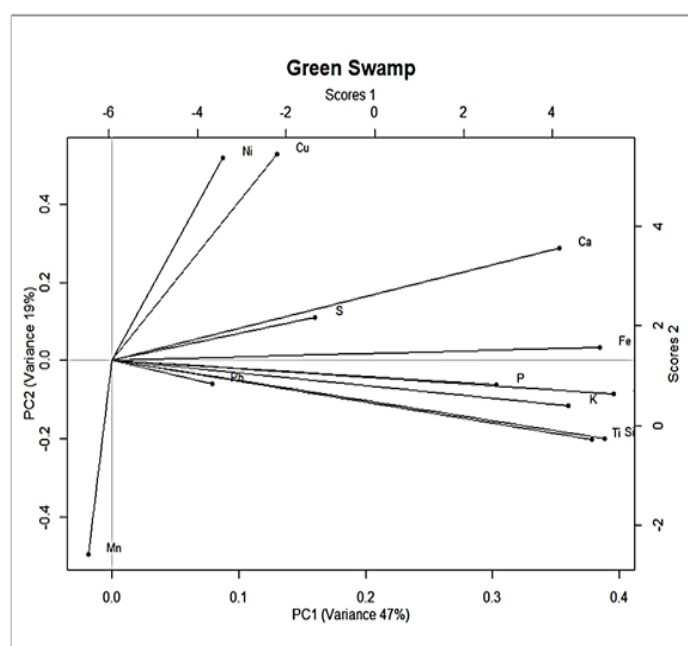


Figure 5.11 Principal component analysis of Green Swamp elements. PC1 and PC2 scores show 47% and 19% variance respectively.

**Table 5.4 Principal component analyses for Green Swamp core. The factor loads for each variable for the two main axes are given in the table**

	PC1	PC2
<b>Al</b>	0.39522	-0.0868
<b>Si</b>	0.388091	-0.20109
<b>P</b>	0.302746	-0.06265
<b>S</b>	0.160027	0.110948
<b>K</b>	0.359905	-0.11732
<b>Ca</b>	0.352866	0.28669
<b>Ti</b>	0.377894	-0.20235
<b>Mn</b>	-0.01832	-0.49602
<b>Fe</b>	0.384691	0.033435
<b>Ni</b>	0.087003	0.518347
<b>Cu</b>	0.130175	0.528586
<b>Pb</b>	0.078963	-0.06047

### **5.2.2.3 Stable isotopes**

Five samples from Green Swamp core were analysed for stable isotopes of carbon and nitrogen with the IRMS at ANSTO. Pre-treatment of these samples is discussed in Chapter 4. The  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  values in the bulk sediments and C%, N% and C/N ratios were determined. The results of the analysis are shown in Figure 5.12 providing the graphical representation of the results.

The  $\delta^{13}\text{C}$  values in the Green Swamp core varied little and fluctuated slightly between -29‰ and -30.1‰ with slightly lower values recorded at the base of the core. The  $\delta^{13}\text{C}$  values fluctuated from lowest to highest in the middle and then declined towards the top of the core (Figure 5.12). The C/N ratios were between 6.9 and 15 between 4 cm and 76 cm depth. Like the  $\delta^{13}\text{C}$  values, the C/N ratios were lower towards the top. These results did not interpret any specific trend in  $\delta^{13}\text{C}$  values. However, high P/B ratios and  $\delta^{15}\text{N}$  values in the middle sections of the cores suggested increased nutrients that may have favoured plankton rich conditions.

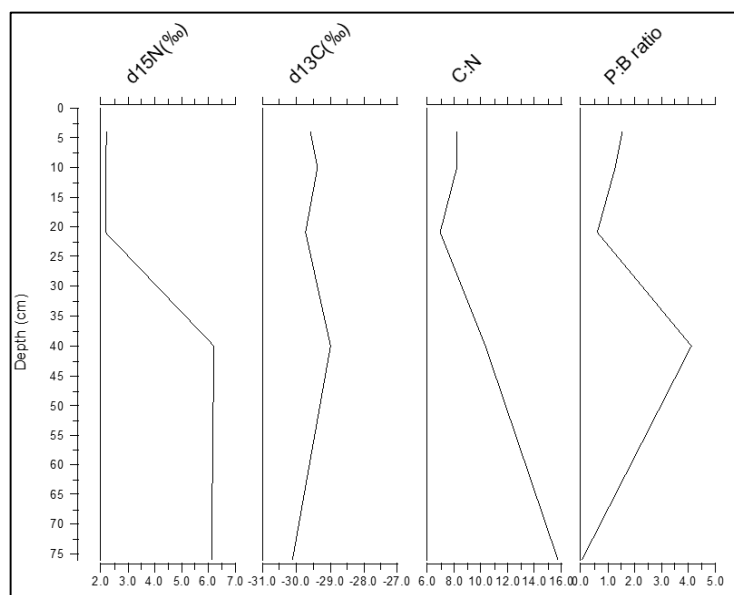


Figure 5.12 Variations in  $\delta^{15}\text{N}$ ,  $\delta^{13}\text{C}$ , C/N and P/B with depth of Green Swamp Core. Values of Isotopes are presented in per mille units (‰)

#### 5.2.2.4 Diatom analysis

The summary diatom diagram of Green Swamp (Figure 5.14) shows four major assemblage zones (GS1, GS2, GS3, and GS4) as defined using CONISS. Zone GS2 is further subdivided into GS2a and GS2b. The zones are described in the following sections.

##### Zone GS4 (75-65cm)

This zone was characterized by the dominance of benthic (*Pinnularia* spp., *Gyrosigma* spp., *Eunotia serpentina*), planktonic (*Aulacoseira italica*, *Aulacoseira granulata*) and aerophilous species (*Diadlesmis confervaceae*). The presence of *Eunotia* spp. and *Pinnularia* spp. suggested occasional low pH values, although *Cocconeis placentula* and *Epithemia* spp. suggested alkaline conditions. Taken together, they represent variable conditions with clear water. The aerophilous species in Zone GS4 showed higher percentage values (24%) than planktonic species. High values of aerophilous species i.e., *Diadlesmis confervaceae* and *Sellaphora pupula*, likely indicate sediment input and/or prevailing mudflat conditions and hence are another indication of high variability. Epiphytic species were rare suggesting low plant cover.

##### Zone GS3 (65-50cm)

This part of the core was dominated by the planktonic species *Aulacoseira alpigena* and *Aulacoseira ambigua* constituting as much as 80% of the diatom sum. *Aulacoseira alpigena* reached its maximum abundance of 68% at 60 cm whereas *Aulacoseira ambigua*

peaked at 48% at 55 cm. This shift to plankton dominance suggested that the river or creek was more connected to the wetland and was likely a source of sediments. *Aulacoseira alpigena* was observed to enter sediment records elsewhere after regulation (e.g., Fluin *et al.* 2007; Reid *et al.* 2007) and so its rise above 70 cm supports the dating which suggests the record covers the post-regulation phase. The arrival of *Cyclotella meneghiniana* at 55 cm provides the first evidence for nutrient enrichment.

A rapid decrease in the abundance of most aerophilous and benthic data was seen in Zone GS4 and continued here. This decline in benthic forms suggested more turbid water conditions that favour plankton diatoms. The gradual rise in *Gomphonema* spp. suggested an increase in aquatic plant cover which continued to zone G2b where it reaches maximum. *Eunotia* spp. were not recorded above 60 cm. This zone is considered as a transition zone where the shift from benthic to planktonic diatoms occurred.

#### *Zone GS2b (50-25cm)*

In this zone the species composition changed rapidly. This zone was characterised by the shift in dominance of the planktonic diatoms to *Aulacoseira granulata* which reached a maximum abundance of 78% at 35cm depth. Lower percentages were recorded in the remaining planktonic diatoms such as *Aulacoseira alpigena*, *Aulacoseira ambigua* and *Aulacoseira italica*. However, aerophilous species declined to 4% and benthic species were at a low of 19%. This large rise in *Aulacoseira granulata*, and loss of aerophilous (mudflat) species, suggested greater influx of river or creek water showing an increased connection with the Murray River system. The regular occurrence of *Cyclotella meneghiniana* and *Cyclotella pseudostelligera* signalled ongoing periods of elevated nutrient concentrations.

#### *Zone GS2a (25-11cm)*

This part of the zone continued to show a dominance of planktonic diatoms and these accounted for up to 66% of valves, while some benthic and epiphytic forms persisted with average values of 20% and 8.4% respectively. The eutraphentic taxon *Nitzschia palea* became common in this zone rising to 10 % marking increased nutrient concentrations. The presence of *Lemnicola hungarica* suggested an increase in the cover of floating plants, whereas there is no indication of facultative planktonic diatoms.

#### *Zone GS1 (11-0cm)*

While benthic values increased, planktonic forms remained dominant with over 70% of valves and typical planktonic taxa here included *Aulacoseira granulata angustissima*, *Aulacoseira italica*, *Aulacoseira ambigua* and *Cyclotella meneghiniana*. *Aulacoseira*



*granulata angustissima* reached its maximum abundance (43%) at 12 cm. Of the benthic forms *Bacillaria paradoxa* was at 22% of benthic species, which may be reflective of salt input to the wetland. Other benthic diatoms were persistent in low abundances. Epiphytic taxa were largely absent.

PC1 represented 27% of variance and PC2 showed variance of 13%. PC1 scores were used to interpret shifts in diatom assemblages in distinct zones identified by cluster analysis (Figure 5.13 and 5.14). In GS4 and GS3, PC1 score values ranged between -0.4 and -1.4 while a peak of 1.3 score value was obtained in zone GS3 at 58 cm corresponding to a shift to planktonic diatoms. The score values remained consistent between zone GS2a and GS2b while higher values were observed in zone GS1 that ranged between 1.7 and 2.2. PCA diagram showed overlapping between the clustering of zones suggesting mixing conditions in the wetland.

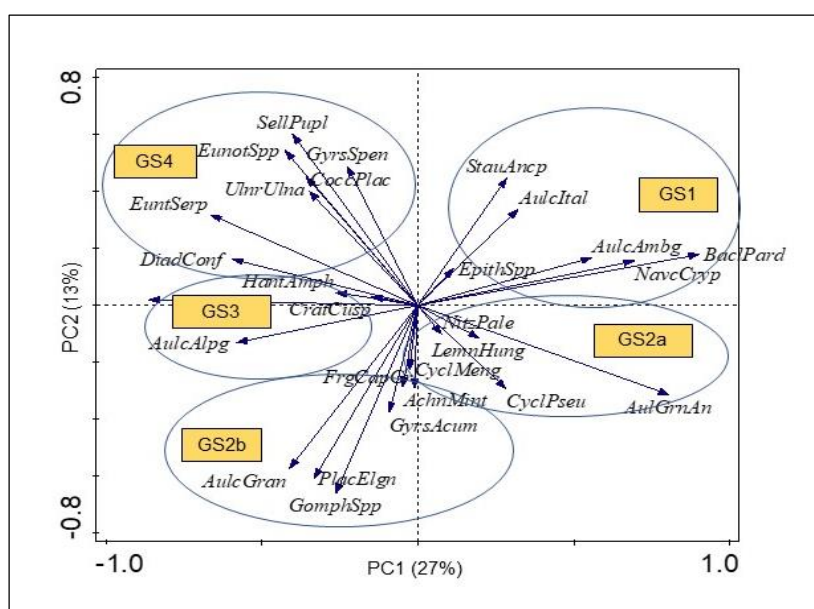


Figure 5.13 PCA for the diatom results and clustering of Green Swamp. Diatom species are shown by arrows and are labelled using species abbreviations. Overlapping between the clustering of the Green swamp zones are reflecting mixing conditions in the wetland.

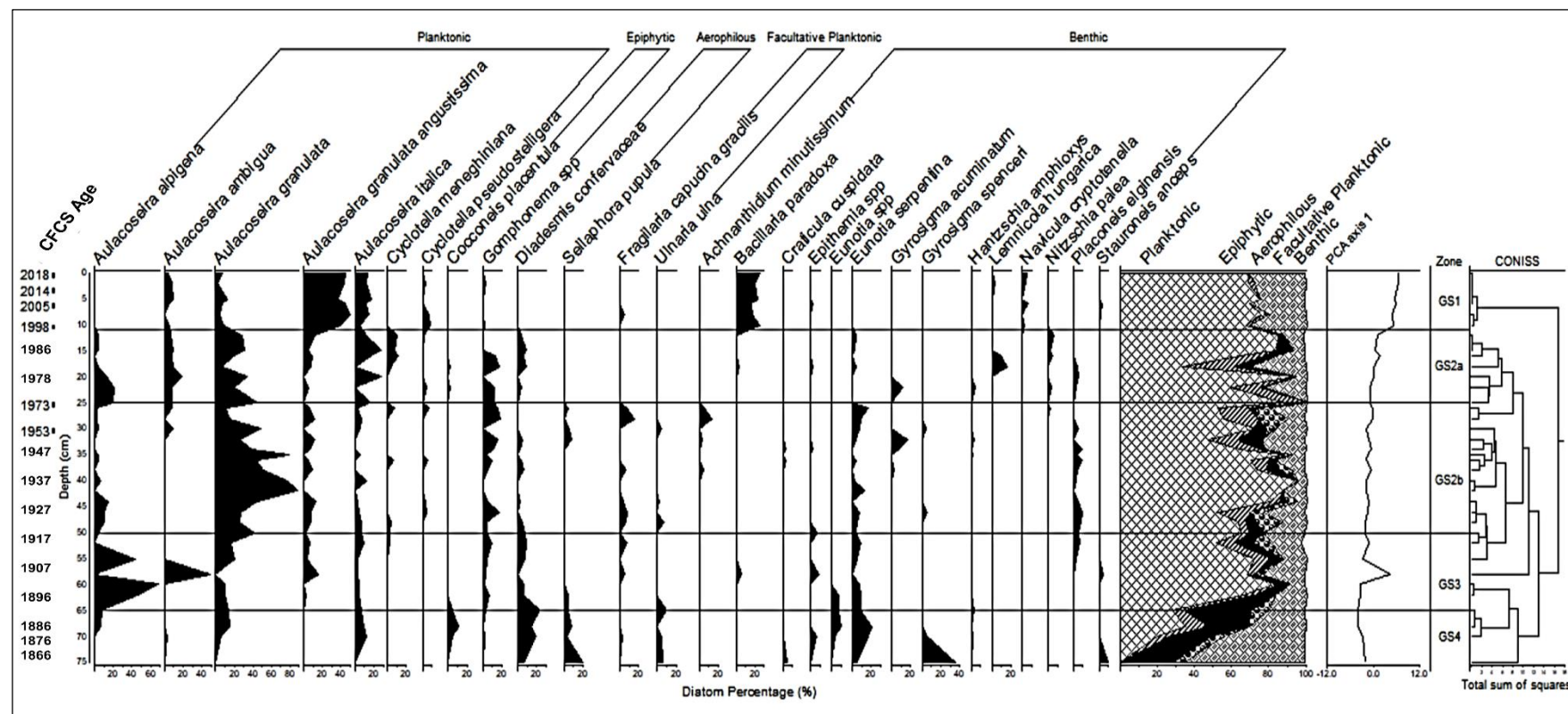


Figure 5.14 Diatom stratigraphic diagram of the relative abundance of diatoms (percentage %) of Green Swamp based on CONISS. Species >5% are shown and are grouped alphabetically according to their habitat preference.

## 5.3 Dairyland wetlands

In May 2018, two sediment cores, 75 cm and 95 cm long, were retrieved from two sampling sites, Cockatoo and Taylors Lagoon, located in the dairy land of Gunbower Island (see Chapter-3). The results of different analysis on these sediment sequences are described in the subsequent sections.

### 5.3.1 Cockatoo Lagoon

#### 5.3.1.1 Lead-210 dating

For Cockatoo Lagoon, nine samples were selected for the  $^{210}\text{Pb}$  analysis. Both CFCS and CRS  $^{210}\text{Pb}$  dating models were used to determine the chronology of the sediment core (Figure 5.15). The results are shown in the Appendix III and illustrated in Figure 5.15. The unsupported  $^{210}\text{Pb}$  activities in the core exhibited a decreasing profile with depth (Figure 5.15) but did not decrease exponentially, but as a non-monotonic profile. The CRS  $^{210}\text{Pb}$  dating model was found to be more reliable for dating sediment cores as the CRS model exhibits non-monotonic  $^{210}\text{Pb}$  profiles (Appleby and Oldfield 1983). Between 7 and 25 cm the  $^{210}\text{Pb}$  activities were close to being constant and indicated a rapid mass accumulation. Using the CFCS  $^{210}\text{Pb}$  dating model, the calculated sediment mass accumulation rate was  $0.080 \text{ cm}^2 \text{ yr}^{-1}$  ( $r^2 = 0.9229$ ), whereas the CRS  $^{210}\text{Pb}$  dating model estimated variable sediment accumulation rates of  $0.069 - 0.10 \text{ g cm}^2 \text{ yr}^{-1}$  between 0-35 cm and  $0.05-0.07 \text{ g cm}^2 \text{ yr}^{-1}$  between 50 and 75 cm depth. Both CFCS and CRS models were found to be in good agreement. According to the CRS model, the age of the deepest sample was  $70.4 \pm 12.4$  years, which means the core was dated back to 1944 CE (Figure 5.15). However, according to the extrapolation method, the age of the deepest sample at 75 cm was 1926 CE.

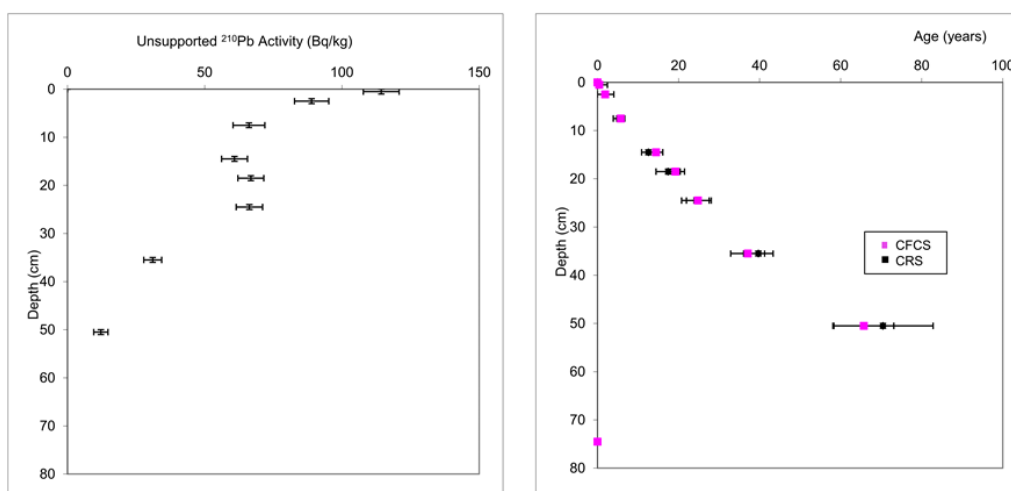


Figure 5.15 Unsupported  $^{210}\text{Pb}$  activity versus depth (left-hand panels) and CRS and CFCS calculated ages by depth (right-hand panels) for Cockatoo Lagoon Data show the mean  $\pm$  s.d.

### 5.3.1.2 Itrax scanning

The Itrax data from the Cockatoo Lagoon core are presented in Figure 5.16 and clustering of four Zones were defined by CONISS. Detrital elements Ti, Al, K, Fe and Si show higher values between 70-53 cm and 40-3 cm in Zone 4 and 5 respectively while these values briefly declined in between these depths. Moreover, S and Mn variations were similar to the detrital elements. Phosphorous was seen throughout the core with high values in Zone 2 (35-10 cm). The values of Ca slowly increased towards the top of the core in Zone 2 and sharply declined afterwards in Zone 1. High values of Ca in Zone 2 (40-8) cm suggested increased pedogenic inputs. Furthermore, high values of Pb were noticed in Zone 4 and Zone 3 whereas Cu, Ni and P gradually increased towards the top of the core from Zone 3 through to Zone 1. Moreover, in Zone 2, Phosphorous showed a correlation with detrital elements and Ca suggesting input from the surrounding catchment and from soil profiles.

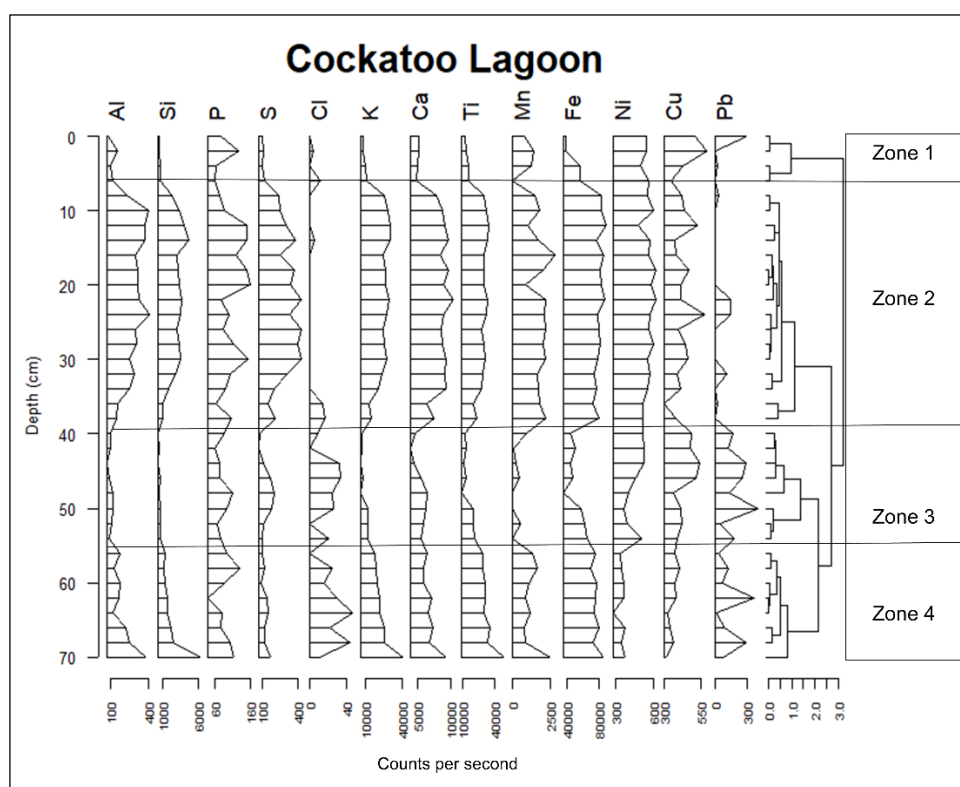


Figure 5.16 Stratigraphic plot of Itrax elements for Cockatoo Lagoon. CONISS output separating samples into zones on the basis of variations in sediment sources across the samples

The correlation matrix was obtained for the Itrax data to identify relationships among those elements (Table 5.5). High correlation values are highlighted as bold ( $r \geq 0.5$  or  $r \leq -0.5$ ). Detrital elements (Al, Si, Fe, Ti, and K) showed strong correlation amongst each other and

with Ca, S, P and Mn. PCA was conducted using an elemental dataset of 13 variables to objectively describe the main variance in the data using only a few factors (Figure 5.17).

The PCA results showed the variation in the elemental components mostly described by the two independent axes; axis 1 and 2. The first two axes represented 74% of the total variance. The first axis accounted for 56% of the total variance, and this axis was controlled mainly by detrital, pedogenic and some nutrient indicating elements. The second axis accounted for 17.56% of the total variance, and was controlled by the presence of Pb and Cl at the positive end while Ni and Cu gradients increased towards its negative end (Figure 5.17, Table 5.6).

**Table 5.5 Correlation coefficients among the elements obtained for Cockatoo Lagoon core by the Itrax XRF Core Scanner**

	Al	Si	P	S	Cl	K	Ca	Ti	Mn	Fe	Ni	Cu	Pb
Al	1.00												
Si	0.93	1.00											
P	0.52	0.54	1.00										
S	0.74	0.70	0.54	1.00									
Cl	-0.46	-0.34	-0.25	-0.45	1.00								
K	0.89	0.95	0.50	0.63	-0.25	1.00							
Ca	0.89	0.87	0.53	0.85	-0.39	0.87	1.00						
Ti	0.78	0.86	0.37	0.45	-0.12	0.96	0.77	1.00					
Mn	0.67	0.67	0.39	0.59	-0.45	0.67	0.71	0.61	1.00				
Fe	0.75	0.76	0.32	0.56	-0.25	0.89	0.80	0.90	0.66	1.00			
Ni	0.39	0.24	0.21	0.59	-0.67	0.10	0.36	-0.10	0.36	0.10	1.00		
Cu	-0.02	-0.15	0.15	0.08	-0.27	-0.27	-0.15	-0.38	-0.09	-0.39	0.41	1.00	
Pb	-0.41	-0.36	-0.39	-0.37	0.54	-0.35	-0.38	-0.28	-0.51	-0.44	-0.35	0.17	1.00

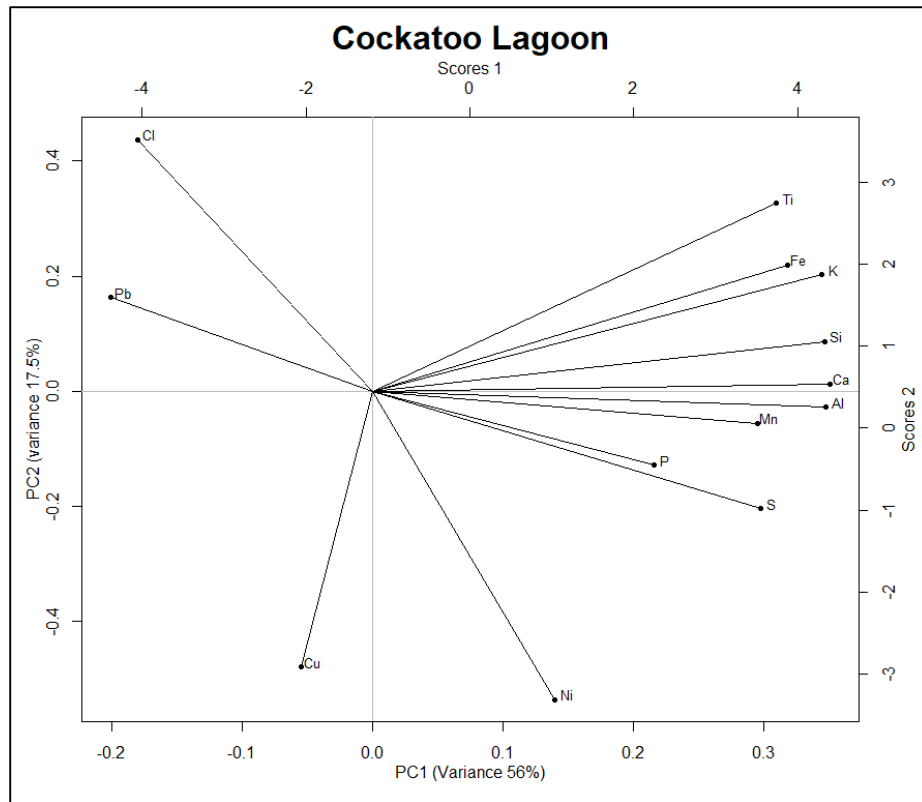


Figure 5.17 Principal component analysis of Cockatoo Lagoon elements. PC1 and PC2 scores show 56% and 17.5% variance respectively.

**Table 5.6 Principal component analyses for Cockatoo Lagoon core. The factor loads for each variable for the two main axes are given in the table**

	PC1	PC2
Al	0.348222	-0.02834
Si	0.346904	0.086227
P	0.216232	-0.1285
S	0.2979	-0.20432
Cl	-0.18034	0.43689
K	0.345151	0.202263
Ca	0.351139	0.012642
Ti	0.310161	0.326468
Mn	0.295399	-0.05621
Fe	0.318666	0.219762
Ni	0.140265	-0.53592
Cu	-0.05496	-0.4793
Pb	-0.20061	0.163041

### 5.3.1.3 Stable isotopes

Stable isotope values were obtained for five samples from Cockatoo Lagoon by analysing them on the IRMS at the ANSTO laboratories. The samples were analysed to determine variations in  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  values in the bulk sediments; C%, N% and C/N values were also determined. Graphical representation of the results is shown in Figure 5.18. The  $\delta^{13}\text{C}$  values for five samples were between -25.9‰ and -27.7‰. At the bottom and surface depth of the core, the  $\delta^{13}\text{C}$  values were lowest -27.4‰ and -27.7‰ and remained quite constant between 45 cm and 10 cm (around -26‰), and also lower  $\delta^{15}\text{N}$  values were observed in the surface sediments (Figure 5.18). Moreover, C/N values fluctuated; the lowest C/N ratios were recorded for 66 cm and 5 cm depths whereas higher values were recorded with 47 at 45 cm, 28 at 25 cm and 25 at 10 cm. These results were plotted with the P:B ratios and no specific correlation between  $\delta^{13}\text{C}$  values and P:B ratios were observed except for at the 5 cm such that if  $\delta^{13}\text{C}$  value was low the P:B ratio was higher.

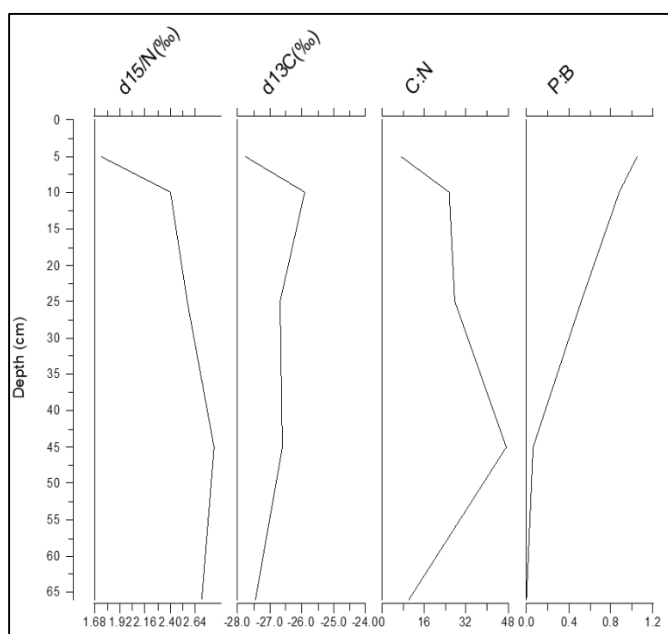


Figure 5.18 Variations in  $\delta^{15}\text{N}$ ,  $\delta^{13}\text{C}$ , C/N and P/B with depth of Cockatoo Lagoon Core. Isotopes are measured in per mille units (‰)

### 5.3.1.4 Diatom analysis

The summary diatom diagram of Cockatoo Lagoon (Figure 5.20) is divided into four major diatom assemblage zones (CL1, CL2, CL3, and CL4) as defined using CONISS. These zones are further described in the following sections.

#### Zone CL4 (75-60cm)

This zone was dominated by a diverse assemblage of benthic diatoms such as *Eunotia* spp. (23%) and *Stauroneis phoenicenteron* (22%) and *Stauroneis anceps* (14%). Benthic taxa dominated between 75-60 cm (75-100%), and, although values decreased slightly towards 60 cm, they remained persistent and dominated the assemblage ranging from 70-100% of valves. Large benthic diatoms suggested clear water conditions in this zone. Planktonic diatoms were in very low abundance (~ 3% at 60 cm) whereas facultative planktonic (*Staurosira* (syn *Fragilaria*) *construens*) forms accounted for 4% at 70cm. The percentage of *Epithemia* spp. in this zone showed an average percentage of about 18% that increased through to zone CL3. However, the percentage abundance of *Eunotia binularis*, an acid tolerant species, decreased from 42% to 24%. The shifts between *Eunotia* spp. and *Epithemia* spp. suggested variable water pH. A dramatic increase in epiphytes was observed with the arrival of *Cocconeis placentula* at 63 cm.

#### Zone CL3 (60-32cm)

This zone was mostly composed of benthic forms (mainly by *Epithemia* and *Eunotia* spp.) and the epiphytes *Cocconeis placentula* and *Gomphonema* spp. The average percentage of epiphytic (36%) and benthic (53%) taxa was less than that of zone CL4. Planktonic (*Aulacoseira granulata*) and facultative planktonic (mainly *Staurosira* spp.) forms were more abundant but remained in low percentages of 6% and 3.7% respectively. The eutraphentic form *Nitzschia palea* was first recorded at the top of this zone. A gradual increase in planktonic species is likely an indication of increased turbidity whereas the rise in *Nitzschia palea* suggests the onset of eutrophication (Van Dam *et al.* 1994). *Amphora veneta* was common in the mid-zone symbolizing increased sources of salt or nutrients (Van Dam *et al.* 1994).

#### Zone CL2 (32-6cm)

This zone is marked by a rise in facultative planktonic taxa which were found to be dominant, with epiphytic diatoms, throughout Zone CL2 averaging 35% and 22% of all valves respectively. The planktonic *Aulacoseira* spp. averaged 19% of all taxa. In contrast, benthic diatoms maintained their presence with a gradual decline until 5 cm. *Eunotia* spp. persisted in low numbers. The appearance of large numbers of facultative plankton, e.g., *Staurosira construens*, throughout the core suggested turbid water conditions.

#### Zone CL1 (6-0cm)

This zone saw a marked increase in planktonic diatoms which dominated with as much as 48% of the sum. *Aulacoseira granulata* sspp. numbers increased to a maximum value of



48% at 0 cm and generally showed an increasing trend towards the top of the core. On the other hand, benthic types, mainly made up of *Achnantheidium minutissimum* and *Encyonema silesiaca*, were 10% at 4 cm and decreased to 4% towards the surface. *Eunotia* spp. persisted suggesting continued high levels of organic acids to the top of the core. Epiphytic species persisted through the core but *Cocconeis placentula* declined to the surface. High numbers of *Staurosira construens* and *Nitzschia palea* in this zone indicated degraded conditions.

PC1 and PC2 scores accounted for 24% and 17% of the variation in diatom species throughout the core (Figure 5.19). PC1 species scores were plotted with diatom results demonstrating high positive scores in zone CL4 transitioning to the rest of the zones with consistent positive scores (5.20). The PCA axis 1 scores showed positive values in zone CL4 (0.1 and 3.1) while in zone CL3 the values were positive for two samples and negative for the remaining 11 (between -0.2 and -0.07). Maximum transition was observed between zones CL4 and CL3. The score values for zone CL1 and CL2 were consistent and ranged between -0.8 to -0.2. Epiphytes maintained a consistent relative abundance in all zones. PC1 Scores showed a peak in CL4, largely driven by epiphytes and benthic diatoms. However, a consistent decrease in PC1 scores in CL3 (mostly negative), CL2 through CL1, from zero to strongly negative, demonstrated a transition caused by the abundance of the facultative planktonic diatom, *Staurosira construens*.

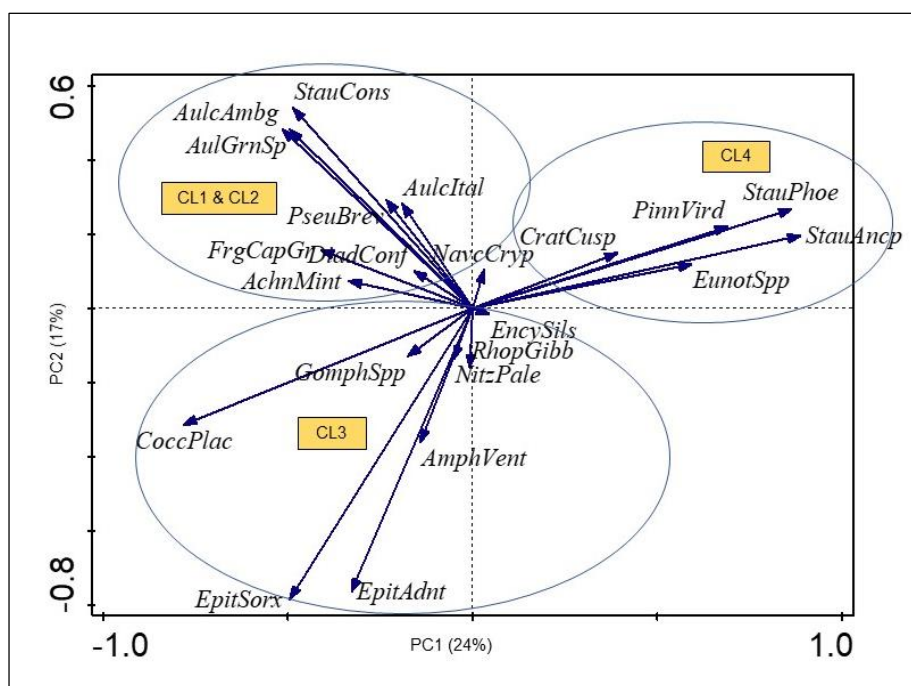


Figure 5.19 PCA for the diatom results and clustering of diatoms with zones of Cockatoo Lagoon. Diatom species are shown by arrows and are labelled using species abbreviations.

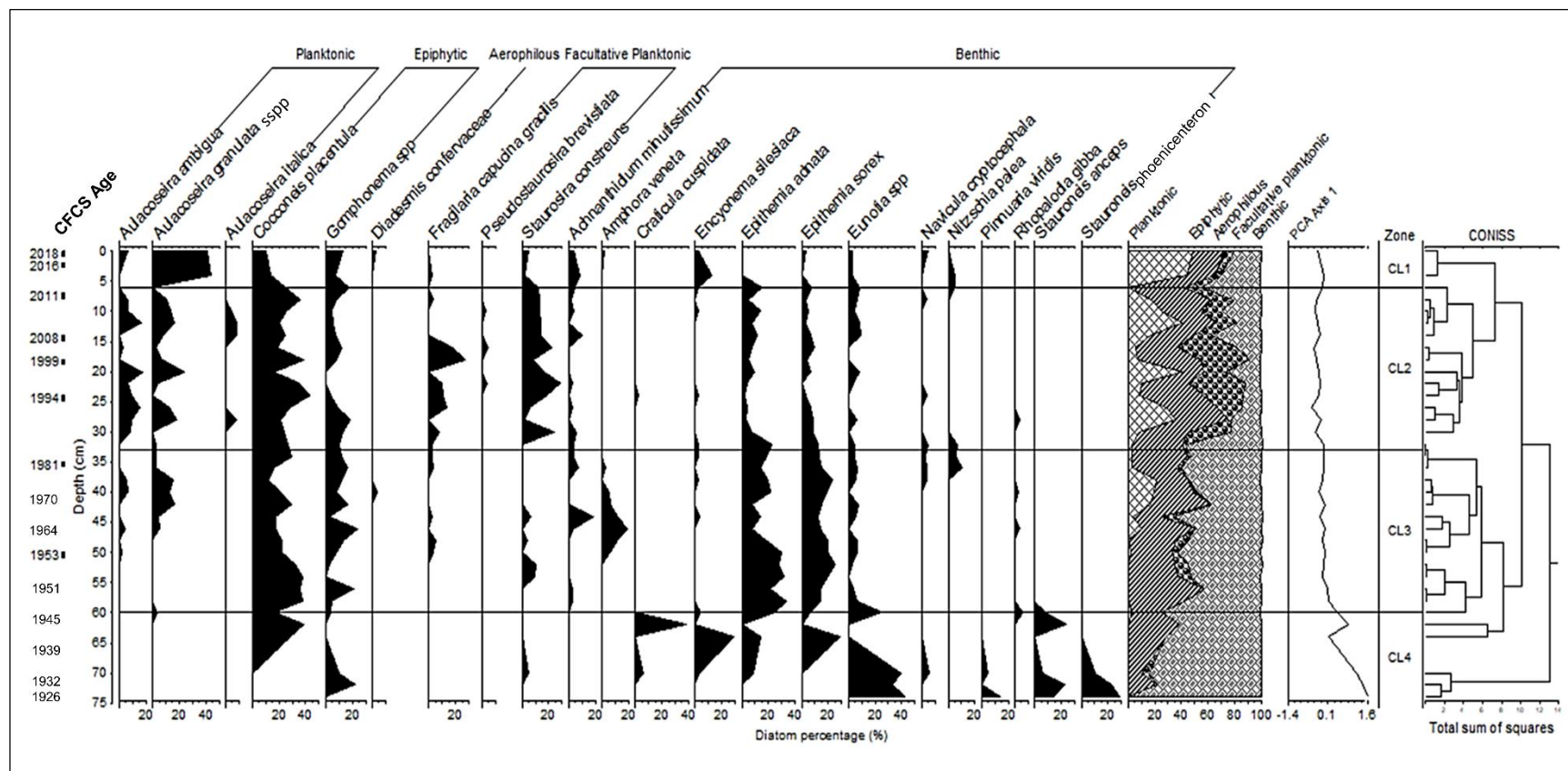


Figure 5.20 Diatom stratigraphic diagram of the relative abundance of diatoms (percentage %) of Cockatoo Lagoon based on CONISS. Species >5% are shown and are grouped alphabetically according to their habitat preference

## 5.3.2 Taylors Lagoon

### 5.3.2.1 Lead-210 dating

Taylors Lagoon core samples were analysed for  $^{210}\text{Pb}$  dating in two stages (discussed in Chapter 4). In the first attempt samples between 0 and 30 cm depth were analysed. The unsupported  $^{210}\text{Pb}$  activity at 30 cm was still relatively high, hence extra samples below 30 cm depth were analysed. The CFCS and CRS dating models were used to analyse the mass accumulation rates and ages of the sediment core. The results are presented in the Appendix III, and in Figures 5.21.

Like Cockatoo Lagoon, the unsupported  $^{210}\text{Pb}$  activities in Taylors Lagoon exhibited decreasing profile with depth (Figure 5.21) but did not decrease exponentially (i.e., non-monotonic). Between 25 and 60 cm the activities are close to being constant, usually an indication of rapid sediment accumulation (Figure 5.21). The CFCS model estimated a sediment mass accumulation rate of  $0.111 \text{ g cm}^{-2} \text{ yr}^{-1}$ . However, the CRS model determined different sediment mass accumulation rates such as  $0.23\text{-}0.39 \text{ g cm}^{-2} \text{ yr}^{-1}$  between 0 and 25 cm and  $0.23\text{-}0.13 \text{ g cm}^{-2} \text{ yr}^{-1}$  between 30 and 101 cm of the core. While the age profiles were estimated using both CFCS and CRS models, the CRS model (Appleby and Oldfield 1983) was considered more reliable for dating sediment cores as this model shows non-monotonic  $^{210}\text{Pb}$  profiles. According to the CRS  $^{210}\text{Pb}$  dating model the basal was aged  $105 \pm 4.9$  years and so the core was dated back to  $1913 \pm 4.9$  CE (Figure 5.21).

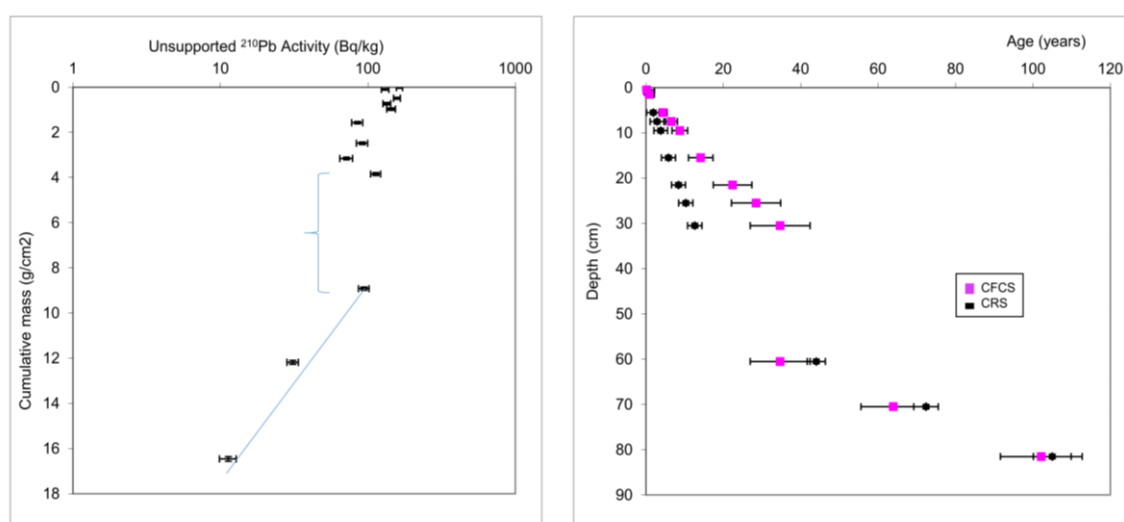


Figure 5.21 Unsupported  $^{210}\text{Pb}$  activity versus cumulative mass ( $\text{g/cm}^2$ ). Bracket shows rapid accumulation rates in Taylors Lagoon (left-hand panels) and CRS and CFCS calculated ages by depth (right-hand panels) for Taylors Lagoon. Data show the mean  $\pm$  s.d.

### 5.3.2.2 Itrax scanning

The Itrax results for the Taylors Lagoon core are shown in Figures 5.22 and 5.23. Magnetic susceptibility values were high in Zone 3, between 87.5-72 cm and gradually decreased towards the surface (Figure 5.22). Detrital enrichment by Al, Ti, Fe, K was evident in Zone 3 and Zone 2, but this gradually declined upwards from Zone 3 to Zone 2 and then rapidly declined towards the top of the core in Zone 1. Phosphorous and S were present throughout the core with higher values in Zone 3 and 2 followed by a gradual decline upwards through the core (Figure 5.22). Fe was found to be more closely associated with K, Ti, Si and Al signalling its detrital origin.

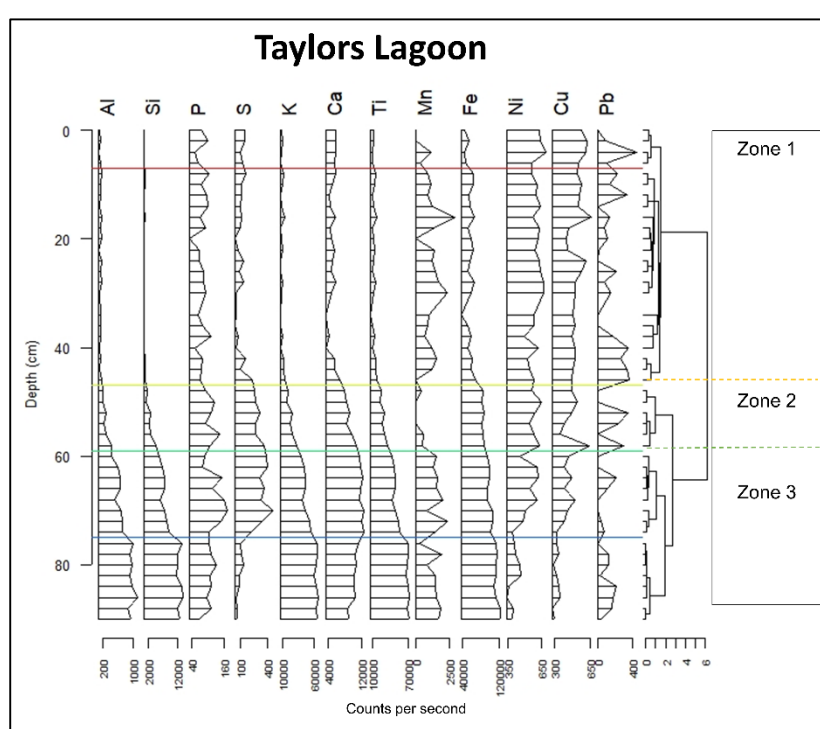


Figure 5.22 Stratigraphic plot of Itrax elements for Taylors Lagoon core. CONISS output separating samples into zones on the basis of variations in sediment sources across the samples

The elemental ratios of Ni/Ti, Cu/Ti, As/Ti and Pb/Ti showed an increasing trend towards the top of the core suggesting recent anthropogenic impacts on the wetland (5.23). The ratio of Ca/Fe are associated with authigenic carbonate inputs which increased towards the top, but showed comparatively high values between 75-52 cm and 5-0 cm indicating dry conditions. However, the variations in catchment erosion or weathering regime were shown by K/Al ratios which were low between 87.5-60 cm.

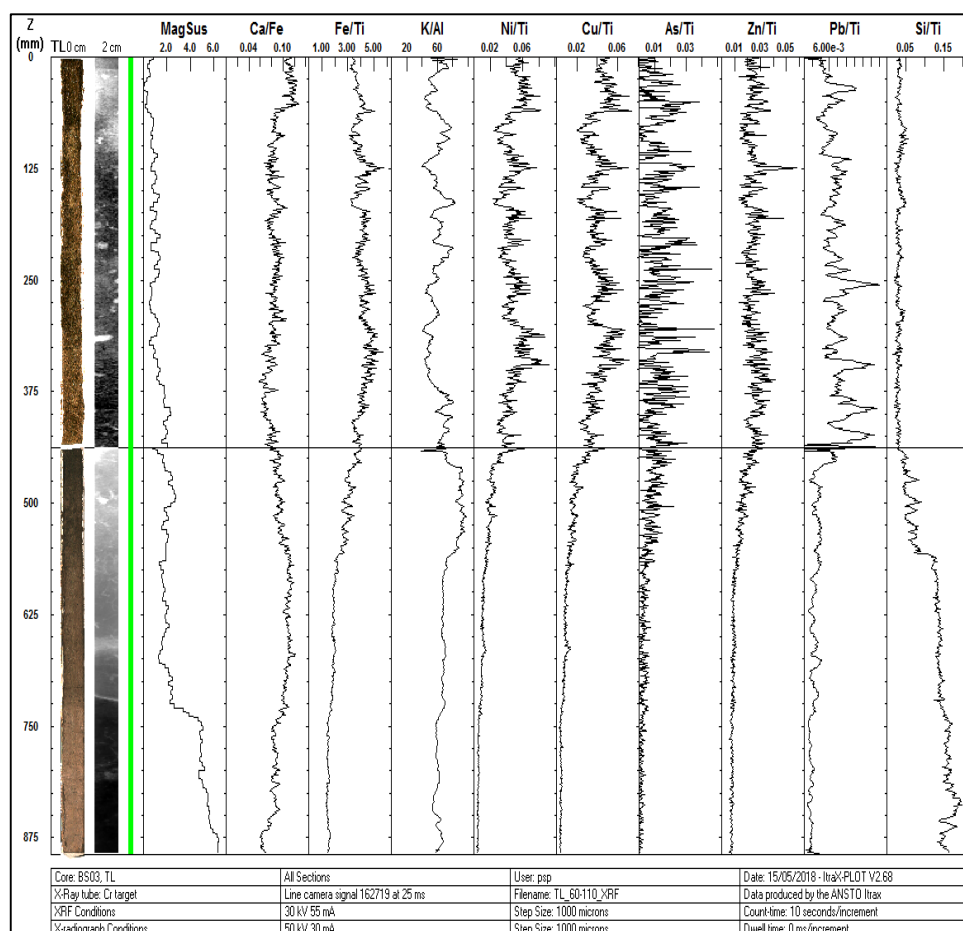


Figure 5.23 Elemental ratio plot for Taylors Lagoon. The elements: Fe, K, Ni, Cu, Zn, As, Pb and Si are normalised with Ti.

As with other cores, a correlation matrix was constructed in order to quantify the strength of association between pairs of elements in the dataset. In Table 5.7, strong correlations ( $r \geq 0.5$  or  $r \leq -0.5$ ) are highlighted. The only group of elements that correlate strongly were Ti, K, Fe, Si, Ca and Al that suggests an association with clay minerals and detrital input. The results of the PCA of the elemental data is shown in Table 5.7. Although 13 axes were defined by the PCA the major variations were shown by two independent axes; PC1 and PC2, that reflect 62% and 14% of total variance respectively (Figure 5.24). The main elements K, Ti, Al, Si, Ca and Fe clustered along PC1 suggesting detrital input is a primary influence whereas Mn, Pb, Cu, Ni and S were aligned with PC2 indicating anthropogenic input or redox processes (Table 5.8).

Calcium has both allogenic (erosion) and authigenic (within lake precipitation) sources (Cohen 2003). Based on good correlations with Ti, Si, K, Fe and Ca, it is suggested that the behaviour of Ca is mainly controlled by silicate and carbonate sources and higher

values were recorded between 80-46 cm (Figure 5.22). The strong correlation between Ti, Si, K, Fe and Ca suggested that carbonate precipitation is particularly important between these depths. Little or no correlation existed between detrital elements and Ca across 40-0 cm. It is possible that, in these upper sediments, a real change in concentrations in the sediment was observed rather than seeing dilution of the elemental signal by carbonates. Manganese did not reflect this dilution, perhaps due to the influence of redox remobilisation. Furthermore, the upper, recently deposited sediments reflect an increase in pollution loads.

**Table 5.7 Correlation coefficients among the elements obtained for Taylors Lagoon core by the XRF Core Scanner**

	Al	Si	P	S	K	Ca	Ti	Mn	Fe	Ni	Cu	Pb
Al	1.00											
Si	1.00	1.00										
P	0.50	0.50	1.00									
S	0.15	0.13	0.57	1.00								
K	0.99	0.99	0.52	0.25	1.00							
Ca	0.80	0.79	0.67	0.68	0.86	1.00						
Ti	0.99	0.99	0.50	0.22	1.00	0.84	1.00					
Mn	0.28	0.28	0.13	-0.10	0.28	0.15	0.27	1.00				
Fe	0.92	0.93	0.58	0.37	0.96	0.88	0.96	0.24	1.00			
Ni	-0.82	-0.83	-0.24	0.07	-0.81	-0.54	-0.82	-0.20	-0.75	1.00		
Cu	-0.76	-0.77	-0.32	-0.02	-0.77	-0.57	-0.78	-0.10	-0.76	0.77	1.00	
Pb	-0.19	-0.18	-0.22	-0.12	-0.20	-0.22	-0.19	-0.08	-0.21	0.13	0.25	1.00

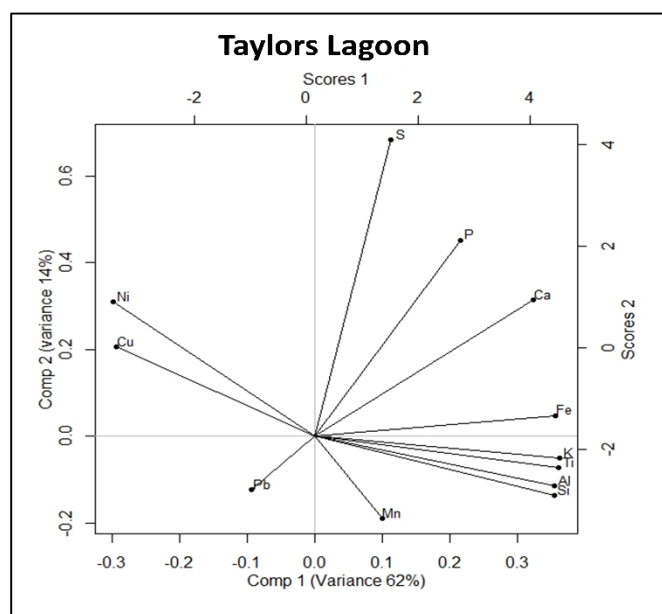


Figure 5.24 Principal component analysis of Taylors Lagoon elements. PC1 and PC2 scores show 62% and 14% variance respectively.

**Table 5.8 Principal component analyses for Taylors Lagoon core. The factor loads for each variable for the two main axes are given in the table**

	PC1	PC2
Al	0.354541	0.104416
Si	0.354392	0.122988
P	0.214644	-0.43707
S	0.11244	-0.66991
K	0.362239	0.042724
Ca	0.322692	-0.30534
Ti	0.361206	0.062009
Mn	0.102651	0.248282
Fe	0.35532	-0.0568
Ni	-0.29761	-0.29308
Cu	-0.29333	-0.18624
Pb	-0.09331	0.110672

### **5.3.2.3 Stable isotopes**

Stable isotope values were obtained for six samples from Taylors Lagoon by analysing them on the IRMS at the ANSTO laboratories. The samples were analysed to determine variations in  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  values in the bulk sediments and C%, N% and C/N values were also estimated. The results of the analysis are shown in Figure 5.25 which provides the graphical representation of the results. The  $\delta^{13}\text{C}$  values of the Taylors Lagoon bulk sediments were between -29‰ to -25‰ and showed a profile of increasing depletion towards the top of the core. The value of  $\delta^{13}\text{C}$  for the basal sample was relatively high at -27.3‰. The most enriched  $\delta^{13}\text{C}$  value (-25.2‰), recorded at 70 cm, can be related to low planktonic input that is evident from the P:B ratio. In contrast,  $\delta^{15}\text{N}$  values varied throughout and did not reveal any distinct trends to suggest any major sustained changes. Moreover, the C/N ratio remained stable throughout the core.

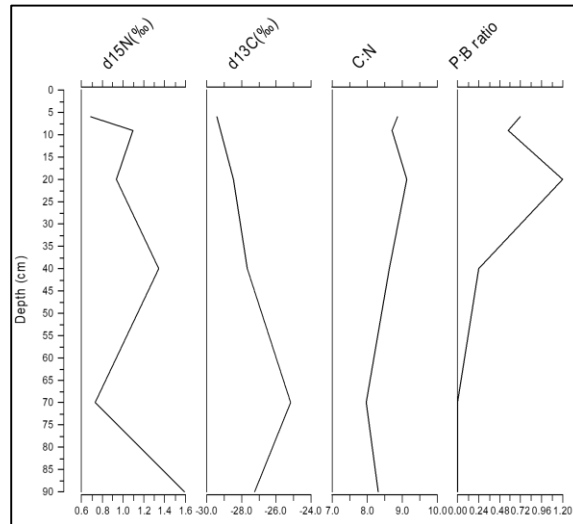


Figure 5.25 Variations in  $\delta^{15}\text{N}$ ,  $\delta^{13}\text{C}$ , C/N and P/B with depth of Taylors Lagoon Core. Isotopes are presented in per mille units (‰)

#### 5.3.2.4 Diatom analysis

The summary diatom diagram of Taylors Lagoon (Figure 5.27) shows three major diatom assemblage zones (TL1, TL2, TL3, and TL4) that were defined using CONISS. These zones are further described in the following sub-sections.

##### Zone TL3 (95-54cm)

This zone was characterised by a diverse flora of epiphytic and benthic diatoms and types that suggest clear, freshwater conditions prevailed at the time. According to the  $^{210}\text{Pb}$  dating, the lower part of the zone (95-82 cm) suggests these sediments may have been deposited prior to regulation whereas the remainder of the core spans the post regulation phase. Throughout the zone benthic species were dominant averaging 58% of all diatom valves. The epiphytic species, *Cocconeis placentula* and *Gomphonema* spp. accounted for up to 29% of species, while the aerophilous *Sellaphora laevissima* and facultative planktonic *Fragilaria capucina* attributed to 1.8% and 1.2%. *Eunotia bilunaris* and *Eunotia formica*, with *Pinnularia* spp. suggest the presence of acidic conditions yet the *Epithemia* spp. and *Rhopalodia* spp. suggest alkaline waters at times. The abundance of different indicator species such as these suggest variability in the water conditions of the lake. The lack of *Aulacoseira* sp. in this zone also suggested the lack of, or distant, connection with the river. Towards the top of the zone (60 cm) there are signs of eutrophication as indicated by the first appearance of *Nitzschia palea* and *Cyclotella meneghiniana*.



### Zone TL2 (54-38cm)

The assemblage of diatoms in this zone undergoes a transition towards planktonic forms although, benthic and epiphytic species remain common throughout. This zone was composed of, on average, 30% planktonic, 25% epiphytic and 39% benthic species. *Aulacoseira granulata* spp. increased from 55 cm to 45 m to an average of 18% in this zone. *Gomphonema* spp. were the main epiphyte while *Craticula* spp. and *Epithemia sorex* were the main benthic types. The rise in *Nitzschia* spp., including *Nitzschia palea*, and *Eolimna minima*, suggested ongoing eutrophication. The rise in *Aulacoseira granulata* spp. indicates increased connectivity of the wetland to the river or increasingly turbid conditions. *Actinocyclus* sp. appeared for the first time at 50 cm depth with an average abundance 8% and, at 40 cm, the abundance declined to 3%.

### Zone TL1 (38-0cm)

The zone is characterised by the dominance of a suite of planktonic taxa, that collectively represented an average of 52.4% of all taxa. *Aulacoseira granulata* spp. increased markedly at 28 cm with an abundance of 30% and varied between 18% and 7% for the remainder of the core. A marked rise in *Nitzschia palea* suggested the onset of high nutrient inputs to the wetland. This interpretation is supported by the loss of *Epithemia* spp. The epiphyte *Gomphonema* spp. declined and was replaced by *Cocconeis placentula* that appeared again at 38 cm which suggests a switch to neutral/alkaline conditions. Of the benthic species *Epithemia* and *Rhopalodia* were recorded infrequently while *Achnantheidium minutissimum* and *Navicula cryptocephala* were relatively common. The rise in *Diadlesmis confervaceae* may be explained by the influx of fine sediments; or the prevalence of mudflats when water levels were low.

PC1 demonstrated 40% and PC2 18% of variation in diatom species in the sediment core (Figure 5.26). The zones clustered according to the PC1 and PC2 scores while only PC1 species scores were plotted with diatoms to understand the variation throughout the core. The scores for PC1 in zone TL3 showed negative values that varied little throughout the zone, ranging between -1.3 and -0.4. In zone TL2, for two samples the score values were positive while negative values were seen for other samples (5.27). In contrast, axis 1 score values of zone TL1 were positive for all 18 samples ranging between 0.5 and 1.5. This transition was mainly driven by planktonic diatoms.

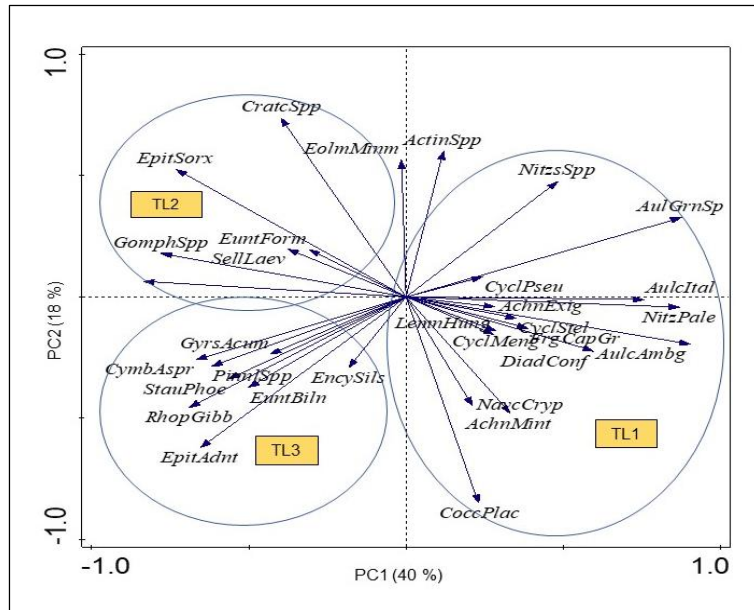


Figure 5.26 PCA for the diatom results and clustering of diatoms with zones for Taylors Lagoon. Diatom species are shown by arrows and are labelled using species abbreviations.

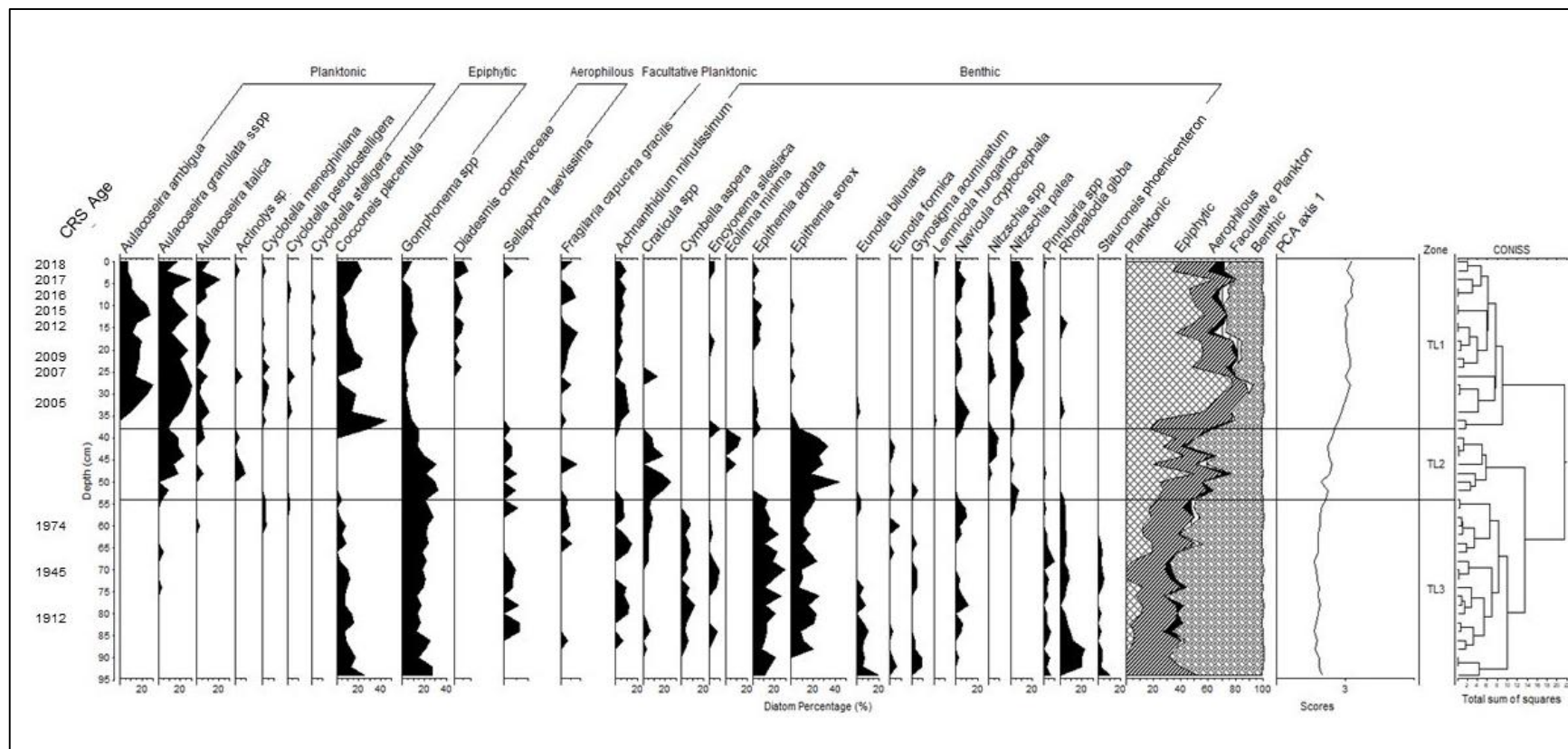


Figure 5.27 Diatom stratigraphic diagram of the relative abundance of diatoms (percentage %) of Taylors Lagoon based on CONISS. Species >5% are shown and are grouped alphabetically according to their habitat preference.

## 5.4 Summary of results

The dating of the cores suggest that net sedimentation commenced at, or shortly before, river regulation. The diatom records of the forest sites revealed a shift from clear water benthic species to an assemblage dominated by river plankton. A similar change was evident in the dairy sites, however the recent increase in eutraphentic diatom species i.e., *Nitzschia palea* suggested some additional, local sources of nutrients. Also, the dairy wetlands showed even more abrupt switches to plankton and higher inferred nutrient levels which indicates impacts of the surrounding dairy catchments. The geochemical proxies indicate changes in soil erosion and anthropogenic inputs from land clearing, climate induced changes and some evidence for metal pollution. The stable isotopes of C and N reveal a shift towards phytoplankton but do not suggest substantial increase in  $^{15}\text{N}$  over the time span. The detailed discussion of these results is presented in Chapter-6.

## CHAPTER 6- DISCUSSION

### 6.1 Introduction

The primary aim of this study is to develop a multi-proxy paleoecological reconstruction to determine the main drivers of change in the Gunbower Wetlands. The need for such research is highlighted through knowledge gaps identified in *Chapter 2 (Literature Review)*. The results of diatom, geochemical, stable isotope and  $^{210}\text{Pb}$  analyses of sediment profiles extracted from Black Swamp, Green Swamp, Cockatoo and Taylors Lagoon are presented in *Chapter 5 (Results)*. This chapter develops paleoecological reconstructions for each site, and explores water quality, ecology and condition of wetlands, based on inferences of the proxy changes reflected in these results. Ultimately, the analysis addresses the question “how has the water quality, ecology and conditions of wetlands varied in the past in response to anthropogenic pressures and climate changes?”

As this chapter presents interpretations of the results, all proxy changes are plotted and discussed against age rather than depth to facilitate the analysis of temporal changes in the wetlands and regional environment. The Gunbower wetlands have yielded sediment records that provide clear evidence for their changing ecological history. Dating has shown continuous sedimentation rates in the cores and maximum ages at the base of sediment sequences of around 100 years. Diatoms have shown consistent changes with a shift from benthic and epiphytic forms to a plankton-dominated state in the forest wetlands and more epiphytic and eutrophic conditions in the dairy wetlands. Itrax core scanning has shown variations mostly associated with detrital input in forested catchments while detrital and nutrient inputs were observed in the dairy sites. While the isotope values varied little, there is some support for a shift to phytoplankton and elevated nutrient levels. A comparison of the records from four sites with the previous studies across the southern Murray-Darling Basin, supports previous evidence that inform on change within the whole catchment.

The response of wetlands to anthropogenic pressures is addressed by determining the chronology of the cores and sedimentation accumulation rates which are established in section 6.2. The inferred wetland changes in Gunbower Forest are described in section 6.3 and causes of the observed changes are discussed in section 6.4. In section 6.5, catchment scale change in the Gunbower wetlands is explored and implications for the Gunbower Forest Ramsar site are outlined in section 6.6.

## 6.2 Chronology and sediment accumulation rates

Radiometric dating approaches, particularly Lead (Pb) isotope signatures, have proven to be a powerful tool that can identify the  $^{210}\text{Pb}$  activities within riverine environments (Bird 2011). The approaches using radiometric Lead to date sediment sequences present particular challenges in changing riverine conditions owing to the substantial input of lead entering the system from fluvial, rather than atmospheric sources (Gell *et al.* 2005a; Bird 2011). For example, flooding events may have contributed to mixing and episodic accumulation of sediments. Grundell *et al.* (2012) revealed greater  $^{210}\text{Pb}$  activities in cores closer to the water inlet from the main river suggesting such a source. Despite this, there are several published records that have provided age-depth curves for floodplain lake sediment sequences (He and Walling 1996; Gell *et al.* 2005a; Gell *et al.* 2005b; Gell *et al.* 2006; Fluin *et al.* 2007; Gell and Little 2007; Gell *et al.* 2009). So, the determination of the chronology of the cores, in such floodplain settings, provided a challenge.

While establishing chronologies in these settings using  $^{210}\text{Pb}$  poses challenges, understanding timelines can be assisted by referring to consistent patterns of change in chemical and biological remains across multiple sites. This allows for a regional pattern to be established, particularly if the river channel is a likely source of these indicators (Gell *et al.* 2018a). In addition to dating inferred by  $^{210}\text{Pb}$ , the dating inferences can be made by interpretation of diatom taxa changes with known environmental changes and analysis of *Pinus* pollen. For example, *Aulacoseira alpigena* (syn *A. subborealis*; *A. pusilla*) appears to arrive in Murray River wetland records soon after river regulation towards the mid-20<sup>th</sup> century (Fluin *et al.* 2010). The planktonic species reflect relatively permanent or deeper water, as well as enriched or turbid conditions. The record from Junction Park Billabong (Tibby *et al.* 2020) downstream suggests that *A. alpigena* peaked much earlier, however this is based on low levels of exotic pollen and a record that lacks radiometric dating across the relevant period. The record from near the Perricoota Forest, across the River from Gunbower (Gell *et al.* 2018a), records *A. alpigena* from the mid-19<sup>th</sup> century confirming it as a reliable indicator of post-regulation times in this vicinity. So early increases in this taxon provide a temporal tie-point around 1940 CE (e.g., Gell *et al.* 2018b). Its presence likely confirms post-regulation time.

The arrival of *Pinus* pollen in western Victoria sediments has generally been inferred to represent a date of 1850 CE (Dodson and Mooney 2002; Tibby *et al.* 2006). Along the Murray River Tibby *et al.* (2020) used it to ascribe an age of 1880 CE. However, due to the spatial variability in the development of landscapes around south-east Australia it is likely that, in some cores, there are many decades of post-European sediment

accumulation that are without *Pinus* (Bickford *et al.* 2008). Therefore, the presence of pine pollen definitively recognises post-European sediment, but the first arrival does not necessarily equate to 1850 CE (Gell *et al.* 1993; Mooney *et al.* 2001).

In the present study, the chronology of the Gunbower wetlands was determined by  $^{210}\text{Pb}$  dating and supported by other proxies such as diatoms while exotic pollen inferences were discussed in the chronological interpretations of Black and Green Swamp.

### 6.2.1 Black Swamp

The core extracted from Black swamp was 84 cm long and the CFCS and CRS model suggested an age for the deepest sample as  $82 \pm 13$  years and  $113 \pm 11$  years respectively. While it cannot be ruled out that sediments accumulated before regulation increased the permanence of inundation of Gunbower wetlands, it is likely that the basal age relates to sometime after regulation (allowing for the ~10 cm sediment penetrated by the corer nose but not sampled), and so the age derived from the CFCS model is favoured. Irrespective of which model is accepted, a sediment accumulation rate of 1 cm/yr, or slightly less, is inferred.

Reid *et al.* (2007) observed increases in fine sediment and aerophilous diatoms soon after European settlement followed by a shift to plankton dominance. Elsewhere the shift to plankton follows the onset of river regulation. In this record, an aerophilous taxon increased at 30 cm and the plankton rise began at 35 cm. This may suggest these sediments were deposited at circa 1930; however, for these depths,  $^{210}\text{Pb}$  dating suggests the age of the core to be between 1980 and 1990. These changes are attributable to more recent changes (e.g., 1970s) consistent with Kings Billabong downstream (Kattel *et al.* 2015). The appearance of the nutrient indicator *Cyclotella meneghiniana* at 35 cm supports the thesis that the upper sediment layers represent the recent high impact period. Further, the exotic *Pinus* first appeared in the Black Swamp record at 20 cm. The absence of exotic pollen in Black Swamp and prevalence of clear water, benthic diatoms to 40 cm in the Black Swamp core, suggests the record extends as far back as the onset of river regulation and possibly into times some time before (Gell *et al.* 2018a). While it has been suggested that the first arrival of *Pinus* marks 1850 in western Victoria (e.g., Mooney *et al.* 2000) the town nearest to Black Swamp, Cohuna, was not settled until ~1880. Black Swamp is surrounded by native forest, and has no extensive pine plantations in its vicinity. This suggests that the sediments above this point are deposited post 1880; but are likely to be much younger (Bickford *et al.* 2008). So, the preferred chronology here is that sediments began to accumulate after river regulation, likely following the release of water for irrigation from Torrumbarry weir via Gunbower Creek from 1923.

### 6.2.2 Green Swamp

The core extracted from Green Swamp was 86 cm long and the lowest point of  $^{210}\text{Pb}$  activity was measured at 31 cm by the CFCS model. This model was used to yield estimated ages between 0-31 cm only because below this depth the  $^{210}\text{Pb}$  activity was very high. So, the CFCS model suggested an age of  $71 \pm 25$  years for 31 cm. The age below this depth was calculated by extrapolation of the CFCS model and this yielded an age of 151 years at the deepest sample at 75 cm. This equates to a date of 1866 CE which may suggest the time before Cohuna was settled (~1880 CE). At the base, the diatom assemblage had low numbers of plankton and was mostly dominated by large benthic species (*Pinnularia* spp., *Gyrosigma* spp., *Eunotia serpentina*) that were soon replaced by aerophilous and planktonic diatoms as observed in Hogan's Billabong (Reid *et al.* 2007). However, nutrient indicators are present towards the bottom of the core suggesting an age much after European settlement. Lastly, *Aulacoseira alpigena* has been shown to often arrive in floodplain records after regulation, and so its appearance at 70 cm suggests a younger basal age.

Exotic *Pinus* was first observed at 80 cm in the Green Swamp core (Gell *et al.* 2018a) which confirms its post-European age, but is likely younger than the generally accepted age for the first arrival of *Pinus* in western Victoria at 1850 CE. The extrapolated basal age may suggest net sediment deposition may have occurred before regulation in this case. This is further supported by the high  $^{210}\text{Pb}$  activity at the bottom of the core. Also, given the young age for the arrival of *Pinus* at Black Swamp, its presence at 80 cm in Green Swamp would suggest late 20<sup>th</sup> century age. These features combine to suggest that this record covers a period of high disturbance and so, like Black Swamp, may extend only as far as the early 20<sup>th</sup> century regardless of the extrapolated age.

So, it is likely that these wetlands were not permanently wet at the contact period in the case of Black Swamp, and perhaps not until regulation in the case of Green Swamp. This suggests that the pre-regulation condition included frequent phases where the wetlands were dry and the accumulated sediment was lost (Kattel *et al.* 2015). The diatom valves preserved in the lower part of each core were mostly broken indicating, at most, very shallow water. So, these preferred chronologies described that the average sedimentation rates at these sites were 0.8 cm/yr for Black Swamp and 0.5 cm/yr for Green Swamp (between 0-30 cm). These rates are likely to have been lower early and increased through time. Nonetheless, they are considerably lower than those recorded from wetlands situated closer to the main river channel including in the Perricoota Forest nearby (Gell *et al.* 2018a).



### 6.2.3 Cockatoo Lagoon

The CRS age of the deepest sample of the Cockatoo Lagoon was  $70.4 \pm 12.4$  years which equates to a date of 1944 CE. The age below this depth was calculated by extrapolation resulting in an age for the deepest sample at 74 cm of 98 years which equates to 1919 CE. This suggests that sediment accumulation began before river regulation. Based on the  $^{210}\text{Pb}$  dating, the very basal sediments may mark the period spanning the phase from European settlement to river regulation, but the bulk of the record covers the post-regulation period.

The planktonic diatom *Aulacoseira granulata* was first observed at 60 cm depth in this core. As per the extrapolated age 60 cm equates to 1940 CE which suggest that planktonic diatoms appear in the wetland post-regulation, consistent with other studies. The diatom record between 75-60 cm showed that the wetland was less connected to the River or Gunbower Creek while the sign of eutrophication may signal dairy activities in the region which began in mid-1900. Another taxon, *Gomphonema*, is known to be tolerant of high levels of pollution (Kelly and Whitton 1998) and was present throughout the core. The rise in *Aulacoseira granulata*, dated to the mid-late 20<sup>th</sup> century in the other cores, suggests a younger basal age than the extrapolation suggested. This is supported by the arrival of *Nitzschia palea* at 35 cm suggesting the top half of core relates to the high impact phase over recent decades.

### 6.2.4 Taylors Lagoon

In Taylors Lagoon, the CRS age of the deepest sample was  $105 \pm 4.9$  years which means the core was dated to 1916 CE at 81 cm. The extrapolated age below this depth at 95 cm is 130 years, so the core was dated to 1888 CE. This suggests that sediment accumulation began before river regulation. Based on this dating, the basal sediments may mark the period after European settlement but before regulation. *Cyclotella meneghiniana* and *Nitzschia palea* however, enter the record at or near 60 cm suggesting that these sediments may mark the onset of eutrophication which would suggest a more recent age than the Pb dating would suggest. The rise in planktonic species occurs above 55 cm however suggesting a mid-20<sup>th</sup> century time for this depth.

The sedimentation accumulation rates were high in the upper sediment layers of Cockatoo and Taylors Lagoon which are an indication of recent pollution and inputs from the catchment. In Cockatoo Lagoon, the sediment accumulation rates between 0-50 cm are  $1.01 \pm 0.2$  cm/yr and 0.35 cm/yr at the base of the core (55-75 cm). This is supported by the low abundance of planktonic diatoms indicating less connection with the river but high

numbers of epiphytes in the lower part of the core suggests catchment inputs or may indicate pre-regulation time. In Taylors Lagoon, very high sediment accumulation rates were observed between 0-30 cm of 2.5 cm/yr which indicated significant inputs from catchment disturbance which is supported by the presence of the aerophilous diatom *Diadesmis confervaceae* which can be transported in during erosion events. However, the lower part of the core (60-82 cm) showed a lower sedimentation rate similar to Cockatoo Lagoon.

### **6.2.5 Comparison of sediment accumulation rates**

Many MDB wetlands experienced net sediment accumulation only after river regulation (post 1922) and provide records of change from the late 19<sup>th</sup> century (Gell 2020; Figure 6.1). Kings Billabong was used as a floodplain impoundment servicing one of the first river pumps at Psyche Bend from 1891, and this age coincides with that of its basal sediments (Kattel *et al.* 2015). Before this time, it seems to have dried out frequently enough for any flood imported sediments to be lost (Kattel *et al.* 2015). Further downstream, sediments accumulated in Sinclair Flat after the commissioning of lock 1 in 1922, as is also evident in cores from Loveday Wetland (Gell *et al.* 2007), Lake Cullulleraine (Fluin *et al.* 2010), Green Swamp (Gell *et al.* 2018a), and Dead River Lagoon (Gell *et al.* 2018a). It is not reported that the upper river sites became permanent on account of river regulation (Gell and Reid 2014). In a synthesis of 51 fossil diatom records from wetlands of the Southern Murray Basin floodplain Gell and Reid (2014) revealed comprehensive change in wetland condition. Based on previous studies on mid-reaches of the Murray, increased sedimentation was observed in more than 50% of wetlands raising the threat of complete sedimentation in the shallow systems (Gell and Reid 2016) particularly shallow sites downstream of Wentworth. The analysis of four sites of Gunbower Forest takes the total number of wetlands analysed across the Southern Basin to over 60 (Gell and Reid 2016). The majority of the sediment sequences of these wetlands were short with one no longer than 40 cm retrieved from Little Reedy Lagoon while sequences as long as 1450 cm were extracted from a wetland near Wellington (South Australia) (Gell *et al.* 2005a). A core of 700 cm was extracted from Swanport Lagoon (Gell and Reid 2014) and 460 cm from a site downstream of Wentworth (Gell *et al.* 2005b). The sediment records of only eight wetlands extend into Indigenous time and the typical pre-European sedimentation rates ranged between 0.5-3 mm/yr (Gell *et al.* 2018a). The sedimentation rates in the shorter records, and that in the upper sections of the longer records, were typically 10-50 mm/yr. Other sites have records that extend over 200 cm and were deposited continuously from ~3,000 years ago. These include Mundic Creek (280 cm), Hopcrofts Lagoon (240 cm), and Hogans Billabong (260 cm) (Gell and Reid 2014).

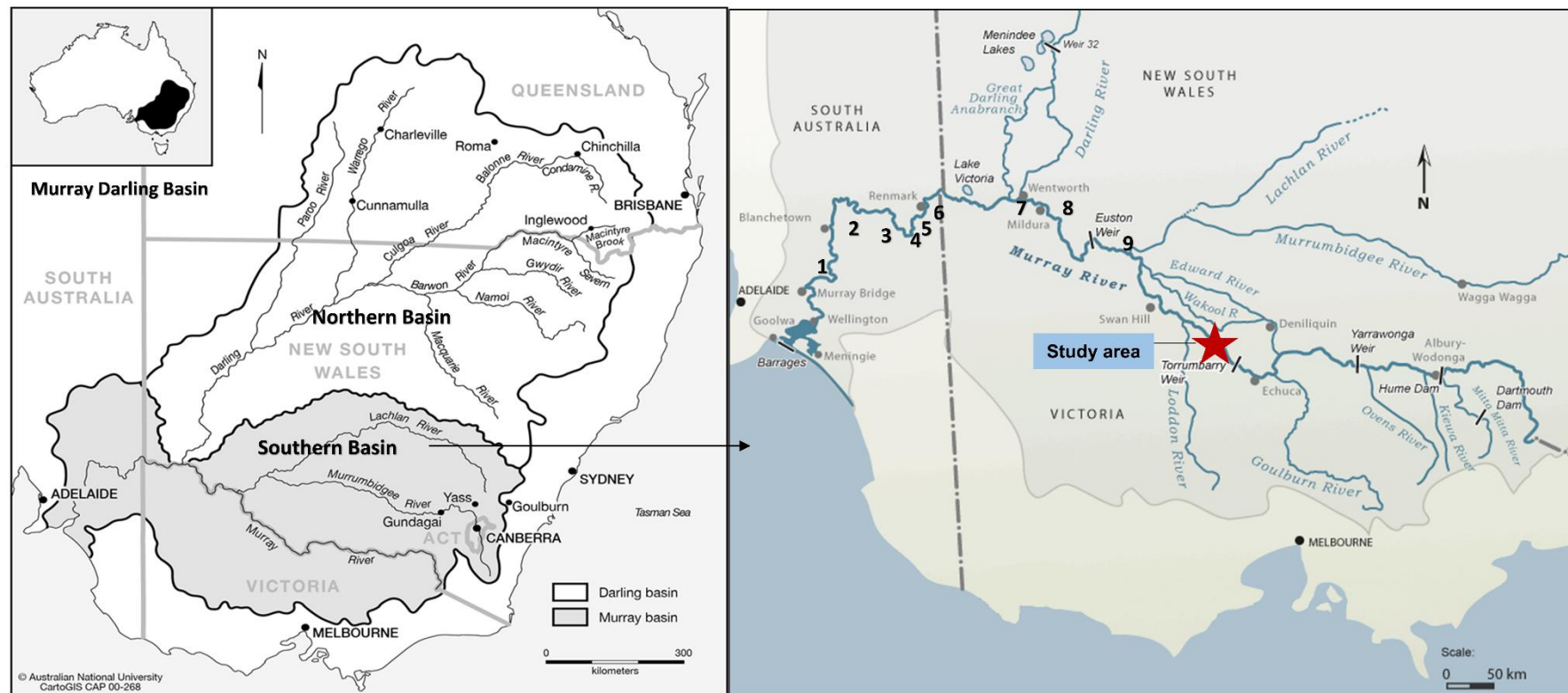


Figure 6.1 Location of the Murray Darling Basin, Australia (map source: CartoGIS Services, College of Asia and the Pacific, The Australian National University). Red star indicates study area: Gunbower Region near Torrumbarry weir. Black circles numbered from 1-9 refer to other sites of Murray Darling Basin. 1= Swanport; 2 = Sinclair Flat; 3 = Loveday Wetland; 4 = Ral Ral Creek; 5 = Lake Cullulleraine; 6 = Mundic Creek; 7 = Kings Billabong; 8 = Psyche Bend Lagoon; 9 = Hopcrofts Lagoon.

In general, while  $^{210}\text{Pb}$  dating in Gunbower wetlands presents a great challenge, there is considerable agreement between sites with sediment accumulation commencing at, or possibly just before, regulation. This is supported by the consistent shifts to plankton dominance, and the arrival of eutrophic indicators, at a range of levels towards the surface. While it is possible that these species arrived in the wetlands at different times, the common pattern suggests that the records have quite similar chronologies. Of the four Gunbower wetlands studied here, only one sequence, from Taylors Lagoon, exceeded 90 cm. Using a mid-range sedimentation rate this would suggest that sediments began depositing in Taylors Lagoon after 1950 CE, although being distant from the main river the rate may be lower than the basin average. The slightly shorter sediment records (< 90 cm) from Cockatoo Lagoon, Black and Green Swamp suggested slightly younger ages, or lower sedimentation rates. This evidence from basin-wide sedimentation rates supports the suggestion that the wetlands with sequence length <90 cm did not experience net sediment accumulation until after regulation.

Being located in the dairy farming area, Cockatoo and Taylors Lagoon showed high sedimentation rates comparable to those that are located in the forested area. So, there is a likely regional source; but this background rate appears to be being supplemented by a supply associated with dairying. Most of these sediments are inorganic and so the river/creek is the likely source. There is more organic content towards the core top; which may reflect increase plant growth as the wetland shallows with infilling. As sedimentation is filling these wetlands, they become shallow increasing the likelihood of macrophyte encroachment, as evident in epiphytic diatoms through to the core top, that also act to trap sediments, influencing the water quality of the Gunbower Wetlands.

### **6.3 Change in the wetlands of Gunbower Island**

Presented in this section are the interpretations of wetland change through synthesis of Itrax, diatom assemblage and stable isotope data for the four sites: Black Swamp, Green Swamp, Cockatoo Lagoon and Taylors Lagoon. This combined analysis can provide information about how wetlands function in response to detrital and anthropogenic inputs.

#### **6.3.1 Pre-regulation**

Gunbower wetlands were likely intermittent prior to the hydrological changes associated with river regulation and irrigation. The basal ages suggest that continuous sedimentation may not have commenced until after 1920. If it occurred prior, then it has been mostly lost during regular drying events. As there was no sediment preserved before regulation, there

are no indicators preserved that may be used to infer the conditions of the wetlands when inundated.

### 6.3.2 Post regulation - permanent clear water

The wetlands of Gunbower Island became mostly permanent at around the time the Gunbower Creek and Torrumbarry irrigation system brought water from the River to the irrigation lands. This development has acted to either increase the frequency of inundation, or for them not to have drained as readily as previously. During the initial phase of inundation, sometime between 1930s and the late 1940s (below 60 cm in all cores), the wetlands were predominately shallow and supported by submerged macrophyte communities. It is likely the lower sections of the Gunbower wetland cores reveal that in early regulation phase, the Gunbower wetlands were oligotrophic and with high water clarity as documented in other diatom studies (e.g., Gell *et al.* 2002; Grundell *et al.* 2012).

The dominance of diatoms in all studied cores is displayed in the summary diatom diagram (Figure 6.2). The basal sediments of the four studied wetlands were dominated by mostly large benthic diatoms as well as epiphytic species because they prefer shallow water owing to their requirement for light. Further, the temporary peaks of planktonic diatoms in Green and Black Swamp, which are known to appear in the Murray system after river regulation, indicate the hydrological variability of the wetlands in response to changes in the river. The appearance of planktonic diatoms in basal sediments suggests their abundance may be reflecting increased river connectivity. This evidence of *Aulacoseira granulata* in sediment records from Green and Black Swamp reflect periods of River flooding (1950s). This inference is further supported by peaks in PC1 scores of diatoms and a transition to relatively negative PC1 scores of Itrax elements, mainly after the occurrence of 1950 flood events (refer Figure 6.3). During this time however, such transitions were not shown by PC1 scores of Cockatoo and Taylors Lagoon (refer Figure 6.4). Further, the absence of *Aulacoseira granulata* in lower sections of Taylors and Cockatoo Lagoon contradict the inference of increased river connectivity or may be reflecting turbid or nutrient enriched conditions that favoured the planktonic diatoms elsewhere. Krull *et al.* (2009) reported that phytoplankton generally has a  $^{13}\text{C}$  depleted signature relative to macrophytes and so this change is likely to be consistent, with the changes in the diatom assemblages.

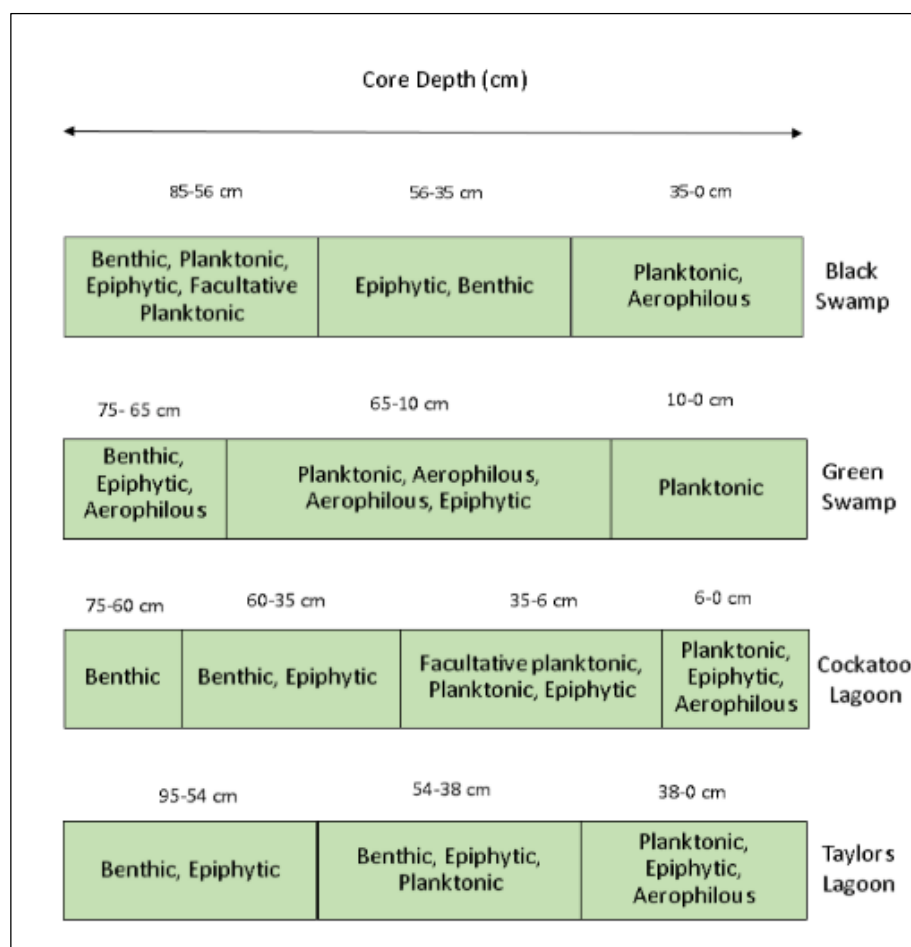


Figure 6.2 Summary diatom diagram of all cores (Black Swamp, Green Swamp, Cockatoo and Taylors Lagoon) showing abundance of diatoms relative to depth. Diatoms infer habitats such as benthic, planktonic, epiphytic, aerophilous, facultative planktonic.

Despite, the occurrence of planktonic diatoms in the lower sections of the Gunbower wetland cores, submerged macrophyte beds are most likely to have played a key role in maintaining a clear-water littoral–benthic dominated system, stabilising sediments, and limiting the release of nutrients for phytoplankton. This general pattern for Gunbower is consistent with Kings Billabong (Kattel *et al.* 2017) and Sinclair’s Flat (Grundell *et al.* 2012). So, while regulation was bringing in more water; or limiting drainage, the quality of water was relatively high.

### 6.3.3 Transition to plankton

The diatoms extracted and identified from the sediment profiles of the Gunbower Wetlands represent a range of environmental attributes and habitats, with notable transitions between sites (Table 6.1). The proportion of river plankton, mainly *Aulacoseira granulata*, is generally high in the upper sections of the Gunbower wetland cores. *Aulacoseira granulata* is a heavy chain forming planktonic diatom that blooms in turbulent and meso-

eutrophic conditions and serves as an indicator of connectivity with turbid rivers (Van Dam *et al.* 1994, Gell *et al.* 2002, Reid and Ogden, 2009, Grundell *et al.* 2012). This can be used to infer connectivity with the main channels at this time (Gell *et al.* 2002, Reid and Ogden, 2009, Grundell *et al.* 2012).

All the Gunbower sites transitioned to plankton dominance, but at different times. Early transitions occurred in Taylors Lagoon (at 55 cm) and Green Swamp (at 65 cm) while such transitions appeared late in the cores of Black Swamp (at 35 cm) and Cockatoo Lagoon (at 5 cm). This shift was mostly of similar diatom species i.e., *Aulacoseira granulata* except the high abundance of the facultative planktonic diatom, *Staurosira construens* that was observed in the upper sections of Cockatoo Lagoon, may be reflecting turbid conditions of the wetland. The Gunbower wetlands exhibited a shift to plankton or to a facultative planktonic species (e.g, *Staurosira construens* in Cockatoo Lagoon) at the expense of benthic (*Epithemia* spp., *Eunotia* spp.) or epiphytic (*Cocconeis placentula*, *Gomphonema* spp.) diatoms (Sonneman *et al.* 2000). This shift is consistent with other diatom records from wetlands of the southern MDB (Gell and Reid 2014). In essence, diatom analysis suggested that gradually, benthic taxa transitioned to epiphytic taxa and, ultimately, the wetlands were dominated by planktonic flora that reveals a transition to more enriched, turbid conditions (Figure 6.2). These incremental changes in water clarity and increasing turbidity and eutrophication may have played a role in the transition in ecological status.

**Table 6.1 Diatom-inferred water quality conditions (Source: Sonneman *et al.* 2000)**

Species	Habitat	pH	Nutrients	Turbidity
<i>Aulacoseira</i> spp.	Planktonic	Neutral – alkaline	Mesotrophic	Moderate
<i>Cyclotella meneghiniana</i>	Planktonic	Alkaline	Eutrophic	High
<i>Diadesmis</i> <i>Conferveaceae</i>	Aerophilous	Neutral-alkaline	Oligotrophic to mesotrophic	Moderate
<i>Eunotia</i> spp.	Benthic	Acid-neutral	Oligotrophic	Low
<i>Gomphonema</i> spp.	Epiphytic	Neutral-alkaline	Mesotrophic to eutrophic	Low
<i>Nitzschia palea</i>	Benthic	Alkaline	Eutrophic	High
<i>Stauroneis</i> spp.	Benthic	Circumneutral	Oligotrophic	Low
<i>Staurosira Construens</i>	Facultative Planktonic	Neutral-alkaline	Eutrophic	High

The plankton assemblage that dominates the upper sediment layers of Gunbower wetlands is linked with transitions shown by PC1 scores (Figures 6.3 and 6.4). Based on geochemical data retrieved from Itrax analysis, this transition may have coincided with changed flow conditions and sediment/nutrient inputs, inferred from detrital inputs/enhanced erosion into the wetlands. In upper sections of Gunbower wetland cores, the detrital inputs were low suggesting comparatively high-water level as inferred by planktonic diatoms with an exception of variable detrital enrichment in Cockatoo Lagoon. The phytoplankton rise may have come from increased connectivity but there was a lag post-regulation inferred from low planktonic diatoms in Gunbower wetlands for quite some time, except in Green Swamp. So, nutrients or turbidity could be likely cause of their dominance. Furthermore, Phosphorous (P) was observed in all Gunbower sites, suggesting considerable inputs from catchment erosion causing turbid conditions of the wetlands (Arai and Livi 2013). However, the stable isotope data revealed little about nutrient changes suggesting most nutrient sources may have been stable consistent with other studies (e.g., Kings Billabong; Kattel *et al.* 2017).

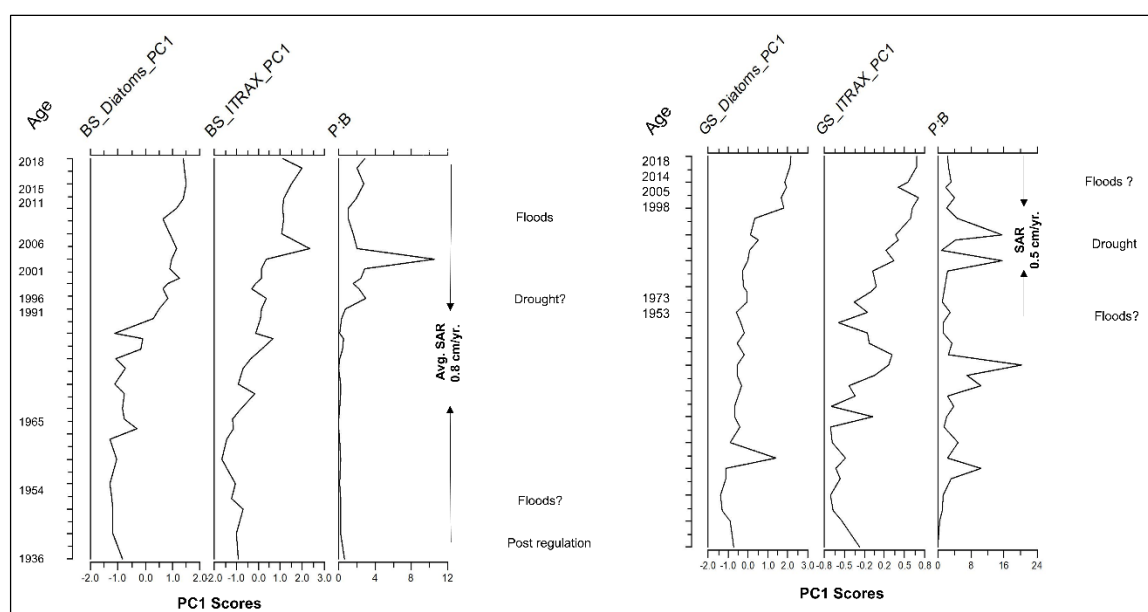


Figure 6.3 Multi-proxy stratigraphic plot for Black Swamp (BS) and Green Swamp (GS). PC1 scores of diatoms and detrital elements are compared with P: B ratio. Average sedimentation accumulation rate (SAR) for Black Swamp and Green Swamp are 0.8 cm/yr and 0.5 cm/yr, respectively.



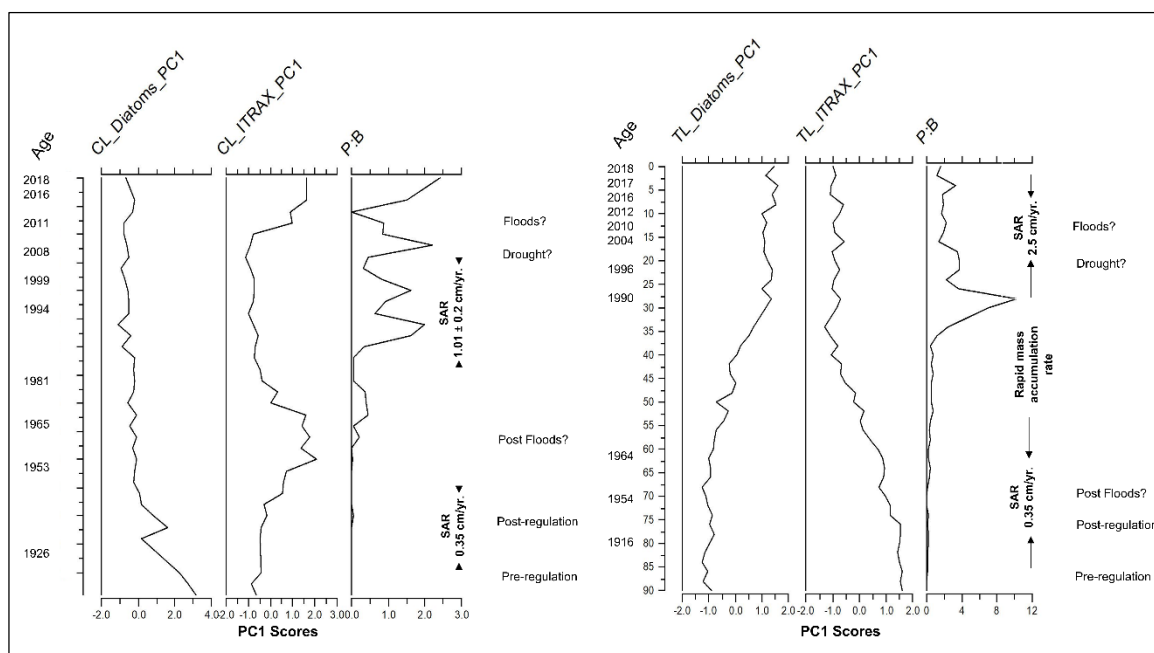


Figure 6.4 Multi-proxy stratigraphic plot for Cockatoo Lagoon (CL) and Taylors Lagoon (TL). PC1 scores of diatoms and detrital elements are compared with P: B ratio.

### 6.3.3.1 Transition to plankton enrichment shown by forested wetlands

In the forested wetlands (Black and Green Swamp), the transition to planktonic diatoms was gradual. The timing of the shift coincided with the introduction of 1955 CE sediment inputs, mainly of detrital origin. The detrital inputs at the base of the core, slowly declined upwards, may be referring to reduced catchment disturbance (small grain size) and increased water levels and regulated flows. These detrital elements are known to carry P, suggesting significant organic inputs into the system. The PC1 scores of Itrax and diatoms reflected a shift to species more tolerant of alkaline water, nutrient enrichment and higher levels of organic pollution. Further, a shift to aerophilous and planktonic diatoms was observed, reflecting turbid conditions. The occurrence of planktonic diatoms after river regulation in the case of Black Swamp and throughout the core in Green Swamp revealed that wetlands are hydrologically variable and may be connected to the river at times. Meanwhile, the negative relationship of planktonic diatoms with erosional indicators (high in basal sediments) revealed the recent abundance of planktonic diatoms (mainly *Aulacoseira granulata* spp.) may be associated with limited light conditions and reduced drainage. In addition, the high abundance of *Diadesmis confervacea* suggested prevalence of mudflats, indicating high variability in the wetlands. The  $\delta^{15}\text{N}$  values showed high nutrient levels in the middle sections of the forested wetland cores that could be associated with animal wastes or irrigation activities. For example;  $^{15}\text{N}$  enrichment was

found in bulk sediments of Lake Alexandrina, South Australia (Herczeg *et al.* 2001) and Kings Billabong (Kattel *et al.* 2017) owing to land use activities.

The high C/N (more than 10) ratio for the surface sediments of Black Swamp core and for the basal sediments (75-65 cm) of Green Swamp core could be related to input from terrestrial sources which is supported by the presence of the aerophilous species *Diadsmis confervacea*, an indicator of catchment disturbance (Figure 6.5). A relatively high contribution of organic matter from aquatic sources was consistent with macrophyte loss and correlated with the diatom *Lemnicola hungarica* around 1975 CE (see Figure 5.7) owing to reduced light transmission. This reflects a shift to turbid waters and plankton-dominated system with variable pH conditions. Further, the  $\delta^{13}\text{C}$  enriched macrophyte in the lower sections of Black Swamp core may be replaced by depleted  $\delta^{13}\text{C}$  phytoplankton in the upper sections of cores, under hydrological variation (e.g., (Krull *et al.* 2009) consistent with other studies (e.g., Kattel *et al.* 2020), indicating algae as the likely main source of organic matter (Lamb *et al.* 2006). This is also shown by the high P/B ratio in the upper sections of Black Swamp core. A significant correlation between nutrient (*Cyclotella* spp. and *Nitzschia* spp.), salinity (e.g., *Bacillaria paradoxa* in Green Swamp) indicators and planktonic diatoms (*Aulacoseira* spp.) is inferred from PC1 scores, suggesting a recent, multi-faceted degradation in the forested wetlands (refer Chapter 5). As discussed earlier (in section 6.2.3) sedimentation rates were lower in the forested wetlands than elsewhere. However, the influx of sediment has been sufficient to impact the algal assemblages to cause the ecological transitions described above. Further, the transition from benthic to planktonic forms is consistent with a state transition that may arise from sudden or consistent increases in nutrient or sediment loads. As such, it appears that, while it may be expected that the condition of these wetlands may be afforded protection from impacts on account of the neighbouring forests, these records show that nutrients, presumably from outside the forest reserve, are entering the waterways. The Gunbower system is flooded by Murray River water which would carry nutrients from highly developed agricultural areas upstream. So, it is expected that benthic diatoms at the base of the cores suggest pre-regulation clear water conditions. However, a shift to planktonic rich diatoms in the upper sections of the cores are suggesting nutrient increases likely well after the more frequent connection following the Torrumbarry weir regulation.

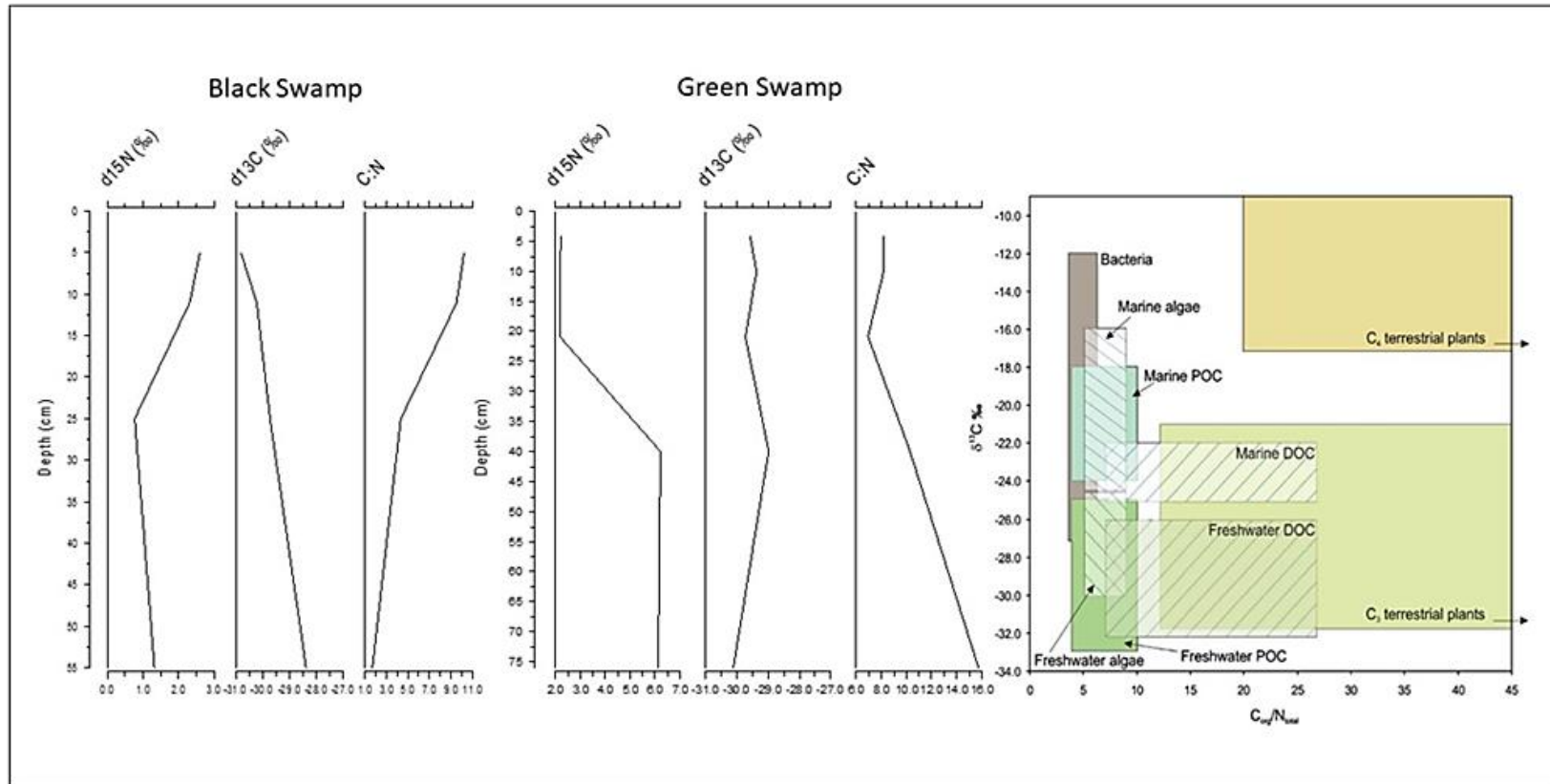


Figure 6.5 Variations in  $\delta^{15}\text{N}$ ,  $\delta^{13}\text{C}$  and C: N, with depth of Black and Green Swamp and their identification of sources of organic matter with C/N vs  $\delta^{13}\text{C}$  (Lamb *et al.* 2006).

### 6.3.3.2 Transition to plankton shown by dairy land wetlands

The transition to plankton dominance was gradual in the dairy land wetlands (Cockatoo and Taylors Lagoon) and occurred at the expense of benthic diatoms because epiphytes persisted throughout the cores (see Chapter 5). The PCA scores of diatoms and ITRAX elements indicated eutrophic, and turbid water conditions with occurrences of nutrient indicators suggesting nutrient driven variation in the wetlands and evidence of further, recent declining water quality.

In the dairy land wetlands, nutrient indicators were observed in the upper sections the cores. A relatively high abundance of nutrient indicating diatoms was found in Taylors Lagoon suggesting nutrients as a cause behind phytoplankton dominance. In Cockatoo Lagoon, the increase in facultative planktonic diatoms, such as *Staurosira construens*, provides evidence for a change towards phytoplankton dominance, and a turbid water regime (Grundell *et al.* 2012) as observed in Kings Billabong. In Taylors Lagoon, low erosional inputs preceded a marked change to nutrient proxies after the 1960s, driving greater phytoplankton and reduced macrophyte dominance likely to have resulted from reduced light transmission. However, in Cockatoo Lagoon, a phase of elevated soil erosion is indicated by increases in detrital elements (Al, Si, K, Ti, Fe) between 1961-2010 CE which corresponds to the recent inputs from dairy activities, further explaining the current degraded state of the wetland. Erosion resulting from dairy and agricultural activities in a lake catchment commonly increases not only the amounts of minerogenic material, but also the input of terrestrial organic matter to the lake, which can be traced by increasing C/N ratios and decreases in TOC content (Li *et al.* 2014). In the middle section of Cockatoo Lagoon, high C/N revealed terrestrial plant derived organic matter and  $\delta^{13}\text{C}$  values suggested a dominance of  $\text{C}_3$  plants (Figure 5.19). This rise in C/N values is supported by the evidence for eutrophic and turbid conditions that correlated with increases in the abundance of *Nitzschia palea* and *Stauroneis construens*. Moreover, Phosphorous and Sulphur correlated with the turbidity indicator, *Stauroneis construens* while *Cocconeis placentula*, *Nitzschia palea* and *Gomphonema* spp. were closely linked to Pb values indicating the degraded and turbid state of the wetland and supporting the growth of planktonic diatoms. This type of erosion-driven degradation of submerged macrophytes have been reported by paleolimnological reconstructions elsewhere, including from the United States (Brush and Hilgartner 2000), Australia (Reid *et al.* 2007) and United Kingdom (Salgado *et al.* 2010).

Similar to the forested wetlands, the lower C/N ratios (less than 10) for surface sediments of the dairy wetlands suggested little contribution of the terrestrial organic matter supply to the sediment, leaving algae as a main source of organic matter (Muller and Voss 1999) (Figure 6.6). This evidence of low values of stable isotopes of carbon and nitrogen are consistent with other studies, with most organic carbon derived from terrestrial or macrophyte sources suggesting little evidence for  $^{15}\text{N}$  enrichment (Kattel *et al.* 2017). This could be referring to the internal feedback mechanisms that advantage growth of planktonic diatoms. In the dairy land wetlands, a switch to neutral/alkaline conditions with planktonic dominance was likely to be driven by both internal nutrient loading and external nutrient influx. Although the high sediment accumulation rate at the top of the core indicates an increase in run-off into the wetlands, the values for C/N ratio are still low (Figure 6.6), indicating that the proportion of allochthonous inputs into the wetlands has remained small.

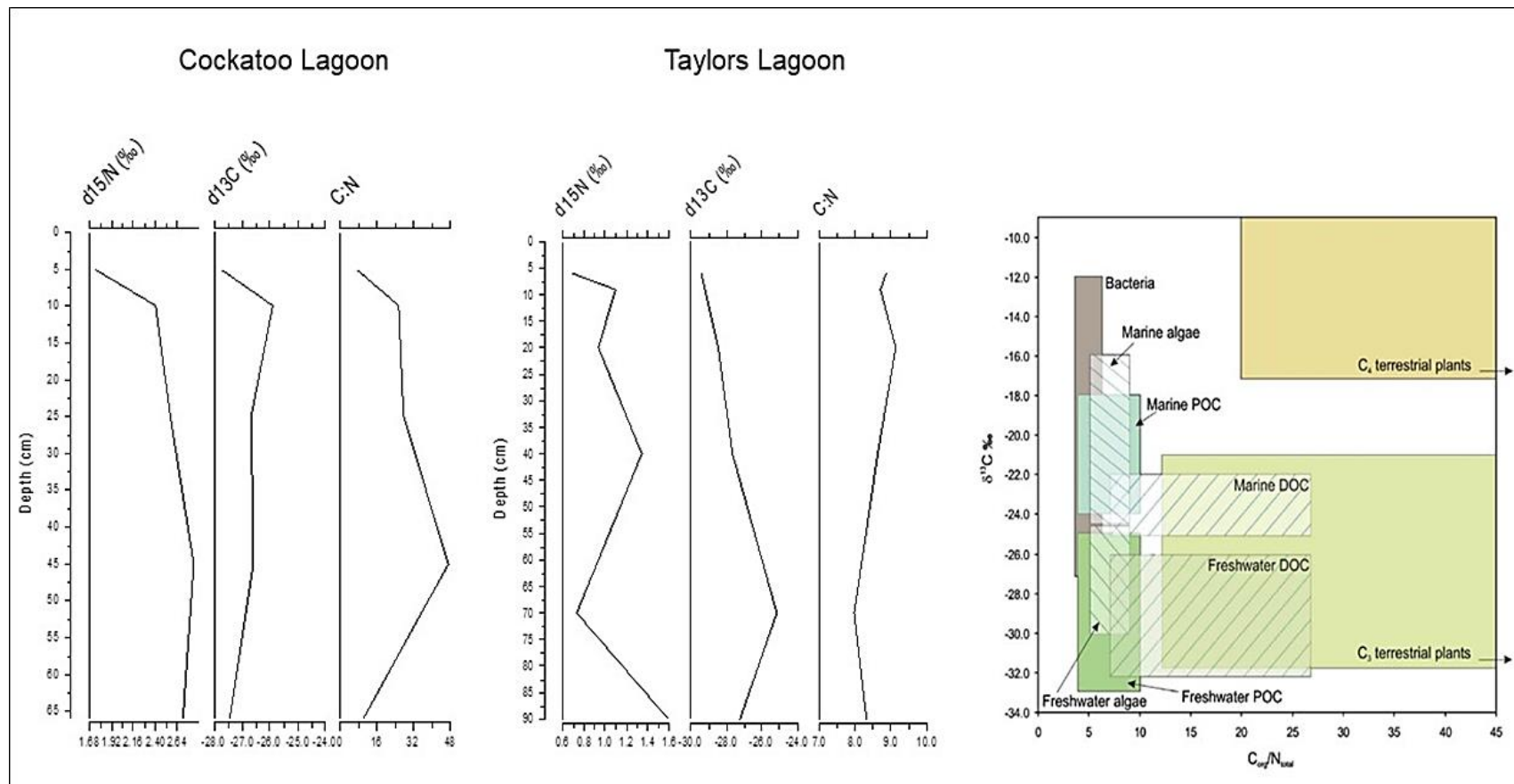


Figure 6.6 Variations in  $\delta^{15}\text{N}$ ,  $\delta^{13}\text{C}$  and C: N, with depth of Cockatoo and Taylors Lagoon and their identification of sources of organic matter with C/N vs  $\delta^{13}\text{C}$  (Lamb *et al.* 2006)

### **6.3.3.3 Summary - shift to a planktonic dominated state in Gunbower wetlands**

In the Gunbower wetlands, detrital and pedogenic inputs were the main sources of sediments yet there were high and low values for detrital material. The main organic inputs in all studied wetlands were from autochthonous sources except some inputs from terrestrial plants were observed in Cockatoo Lagoon. The nutrient levels were low as indicated by low values of  $\delta^{15}\text{N}$  and low numbers of eutraphentic diatoms. It is likely that the flux of nutrients was low, yet the plants may have played a role in recycling incoming nutrients. By 1955 CE the wetlands had become impacted by increased sediment loads that were associated with increased lithogenic rather than detrital input. This can be explained by the increased influx of fine clays which are known to carry attached phosphorus, although the intensification of dairying could represent a local source. Further, non-planktonic nutrient indicators (inferred from *Nitzschia palea*) were noticed after 1950 CE and such nutrient levels are known to trigger major shifts to phytoplankton dominance. In essence, flood events like those in the 1950s may be considered as a possible cause of the transitions in the Gunbower wetlands, while it may have been a general transition with increased dairying and land clearance activities in the Gunbower site.

Based on diatom assemblages, Itrax data,  $\delta^{13}\text{C}$  values and C/N ratios, it can be said that the density of submerged aquatic plants may have begun to decline after 1955 CE when sediment inputs were high. The appearance of *Lemnicola hungarica*, in Taylors Lagoon, Green and Black Swamp suggested the replacement of submerged macrophytes with floating plants. This diatom is an epiphyte on floating plant species such as *Lemna* which rapidly takes up nutrients from water and forms thick floating mats over the water surface. This has detrimental effects on other aquatic life (Emson *et al.* 2018; Gell *et al.* 2018a). Epiphytic algae also affect macrophytes, as greater densities may cover and shade their hosts (Brönmark 1989). The shade formed by the epiphytic coating on a plant surface caused by epiphytic algae suppress the growth of macrophytes. These epiphytes restrict the amount of light reaching the macrophytes, and thereby influence their plant distribution, biomass, productivity and reproductive capability (van Montfrans *et al.* 1984). Here, in the Gunbower wetlands, light limiting conditions could be a cause behind the loss of submerged macrophytes as seen elsewhere.

To conclude, it appears that a sustained transition to planktonic dominance occurred owing to changed hydrology or nutrient/sediment loads. There has been widespread shift to plankton dominance across southern MDB. Further, there is evidence of loss of aquatic plants and their replacement by floating plants. So, many wetlands have changed

ecologically, in a similar way and at a similar time. While the changes are consistent with regime shift models, it remains possible that the assemblage shifts here are responding to ongoing regional scale pressures.

#### 6.3.4 Recent eutrophication

In the Gunbower wetlands, even post the transition to phytoplankton, there has been a sustained rise in nutrient indicators to the top of sampled cores. This evidence is not much supported by isotope or Itrax data. All our samples have  $\delta^{13}\text{C}$  values that are below  $-25\text{‰}$  (Figures 6.5 & 6.6), indicating the dominance of C3 plants as the principal carbon source but organic input from terrestrial vegetation is likely to be low because of low C/N ratios indicating algae are a major component of organic matter. Almost in all Gunbower cores, eutrophic indicators (*Cyclotella* spp., *Nitzschia* spp.) increased in the upper sediments suggesting unprecedented nutrient concentrations in recent (~10-20) years. The abundance of nutrient indicators is particularly high in the dairy sites but also in Black Swamp, a forested site. So, there is likely an ongoing regional eutrophication effect, that is greatest in the dairy sites. This is coincident with the loss of *Epithemia* spp. As the *Epithemiaceae* have blue-green endosymbionts that fix nitrogen they are likely less competitive in high nitrogen conditions (Stancheva *et al.* 2013) and so their decline may be due to nitrogen inputs.

In the MDB, many floodplain lakes have experienced such eutrophic conditions. The nutrient indicator diatoms across Lower Murray sites include, *Cyclostephanos dubius*, *Cyclotella meneghiniana*, *Nitzschia* spp. (incl. *N. amphibia*, *N. palea*), and *Stephanodiscus* spp. Several records have shown substantial declines in *Epithemia* spp. for example; Lake Cullulleraine (Fluin *et al.* 2010), Sinclairs Flat (Grundell *et al.* 2012), Kings Billabong (Kattel *et al.* 2015), Loveday Wetland and Loch Luna (Gell *et al.* 2007), and upstream at Hogan's Billabong (Reid *et al.* 2007). Increases in nutrient indicators, including *Cyclotella meneghiniana* (Tareena Billabong, Gooragool Lagoon, Bomen Lagoon and Psyche Bend Lagoon), and *Nitzschia* spp. (Loveday Lagoon, Sinclairs Flat, Ral Ral Creek, Loch Luna, and Hogan's Billabong), were also reported (Figure 6.7). Hence, there is a regional scale continuation of eutrophication.



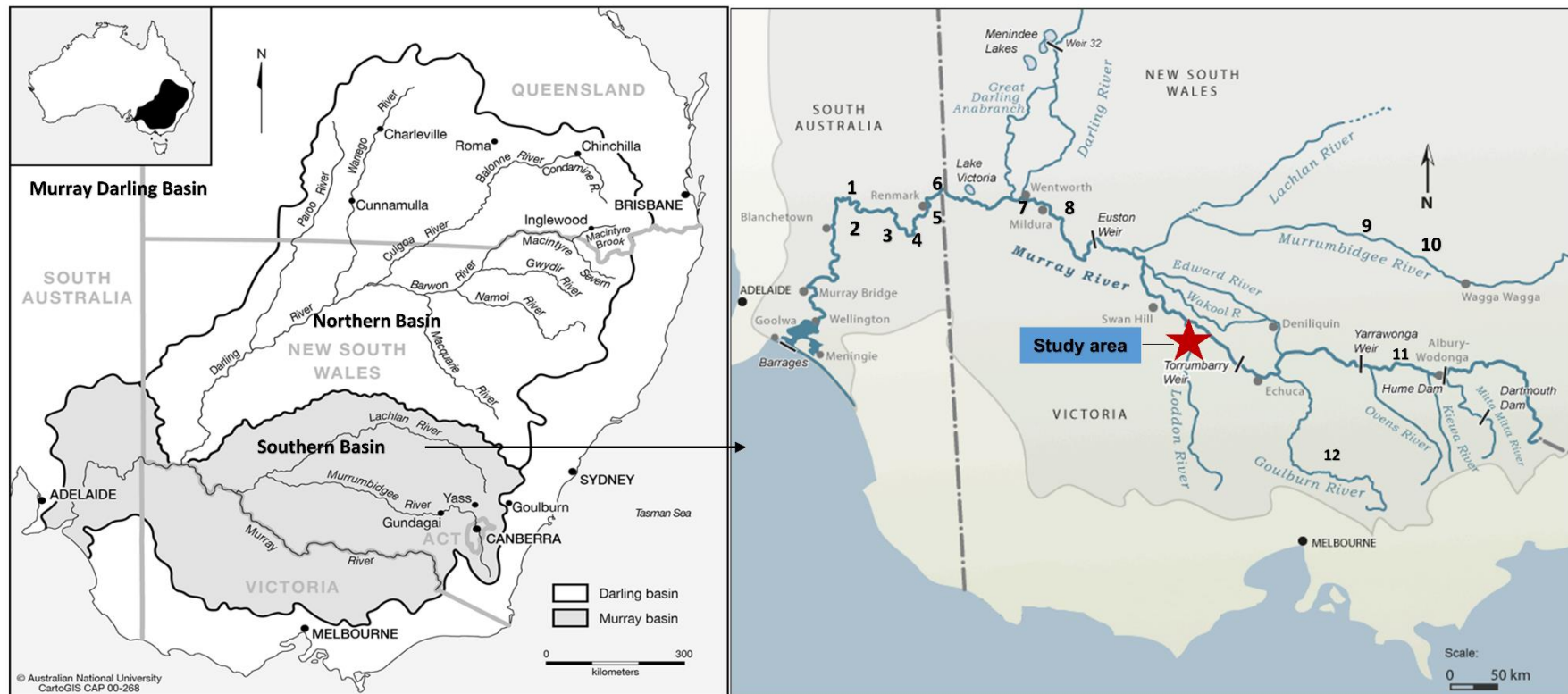


Figure 6.7 Location of the Murray Darling Basin, Australia (map source: CartoGIS Services, College of Asia and the Pacific, The Australian National University). Red star indicates study area: Gunbower Region near Torrumbarry weir. Black circles numbered from 1-11 refer to other sites of Murray Darling Basin. 1= Loch Luna; 2 = Sinclair Flat; 3 = Loveday Wetland; 4 = Ral Ral Creek; 5 = Lake Cullulleraine; 6 = Tareena Billabong; 7 = Kings Billabong; 8 = Psyche Bend Lagoon; 9 = Gooragool Lagoon; 10 = Hogan's Lagoon; 11 = Berry Jerry Lagoon; 12 = Lake Callamondah.

### 6.3.5 Regime shifts

The consistent rise in plankton across the four Gunbower wetland sediment records suggested a consistent cause. Typically, diatom plankton species are more abundant in deeper waters and so a shift towards plankton could suggest increased inflows. As the wetlands in the Gunbower Forest are shallow however, and water flows out after flooding, there is little capacity for them to attain depths that would see a sustained shift from benthic to planktonic species. As they are the dominant algae in the main River (Bormann and Likens 1967), they likely to represent an increase in connectivity with the river as suggested by (Kattel 2012b). They are also known to benefit from elevated nutrient levels and several taxa are considered to be indicators of eutrophic conditions (e.g., *Cyclotella meneghiniana*, *Aulacoseira granulata*). Several early studies (e.g., (France 1995; Doi *et al.* 2010) used the mere ratio of planktonic to benthic taxa to indicate elevated nutrients. Plankton is also advantaged, and benthic taxa disadvantaged, by a reduction in photic depth and this can be caused by the input of both nutrients and fine sediments which combine to increase turbidity (Reid *et al.* 2007). Finally, in the Murray system, fine clays carry high loads of Phosphorous and so there is a direct link between sediment and nutrient flux. So, the shift to plankton can be explained by nutrient and sediment flux and changed hydrology. Here, while inundation appears to have become permanent favouring plankton, the early-stage taxa are reflective of clear water conditions, suggesting that the water that came to fill the wetlands on a regular basis was of relatively high quality. The increases in aerophilous species (e.g., *Diadesmis confervacea*), suggesting erosional inputs, and non-planktonic eutraphentic taxa (e.g., *Nitzschia* spp.), coincident with the rise in planktonic species, are consistent with both nutrient and sediment flux being the main cause. That this change is common to all cores, both from forested and dairy lands, is evidence that the cause is at a regional scale – likely delivered via the main river or its tributary Gunbower Creek. The dairy zone sites however, tend to show higher proportions of clearly eutraphentic species suggesting that local sources are exacerbating the elevated nutrient status of the region's wetlands.

In the absence of clear drivers of a shift to phytoplankton dominance, it remains that there may be internal trophic feedback mechanisms at play whereby ongoing doses of pollutants such as nutrients or turbidity, have advantaged phytoplankton that have themselves increased to the point where the light environment has declined to the detriment of macrophytes. This is consistent with the model of Gell and Reid (2016) and is consistent with the slow changes documented in Kattel *et al.* (2020) at Kings Billabong.

## 6.4 Possible causes behind the wetland changes

### 6.4.1 Climate change

Climate change and variability can influence water quality, water quantity and habitat functioning of wetlands. The response of wetlands to climate change will depend upon their exposure to altered climate conditions, sensitivity and resilient capacity. Wetlands are known as dynamic systems that respond to wet/dry regimes, and paleoenvironmental and paleoclimate records integrate this variation. So, these paleo-records can examine decadal or annual changes in wetlands. The MDB is sensitive to changes in Interdecadal Pacific Oscillation (IPO) and *El Niño*-Southern Oscillation (ENSO) (Gallant and Gergis 2011). The Pacific Decadal Oscillation (PDO) caused multidecadal drought (1898-1946) and flood-dominated regimes (1946-1976). As a consequence, fluctuations in the pervasiveness of wet or dry states have been apparent (Hendon *et al.* 2007). While this study does not record detailed evidence for climate change, some climatic variations can be revealed from diatoms, elements and stable isotopes (see Figures 6.3 and 6.4). These records assess the degree to which the main phases of climate change may have added or reduced the impacts of catchment disturbance.

In the Gunbower Wetlands, a high abundance of benthic diatoms in the basal sediments of the collected cores suggested low water levels in the years following regulation, corroborated with high values of Ca, an indicator of dry conditions. Benthic communities survive on detritus aggregations which are evident from high detrital inputs in the lower sections of the cores, also corresponding to the Drought Dominated Regime (DDR) of 1898-1946 CE. These findings are supported by lower P/B ratios in lower sections of all Gunbower wetlands except Green Swamp, clearly suggest that wetlands were covered by submerged macrophytes between 1930 CE and 1946 CE when the human disturbance was low. During DDR (pre-1946 CE and 1997-2009 CE), in the Gunbower wetlands, the conditions were dry supported by  $\delta^{13}\text{C}$  enrichment while the appearance of some phytoplankton (*Aulacoseira italica* and *Aulacoseira granulata*) could be indicative of 1950s floods under the Flood Dominated Regime (FDR; 1946-1976). The abundance of *Aulacoseira granulata* in the Gunbower wetlands after the 1950s flood event is reflective of a discrete wet period or light limiting conditions from sediment loads associated with floods. The occurrence of planktonic diatoms suggested that the wetlands would have been impacted by the substantial flood events of the 1950s. However, the abundance of planktonic diatoms in the DDR contradicts their flood-induced occurrence. So, despite the climate variability, the dominance of planktonic diatoms, suggests reduced drainage and turbid water conditions as the main drivers. Under the influence of the FDR of 1946-1976 CE, enhanced abundance of *Diadesmis conferveacea* after 1980 CE could be influenced

by the FDR, hence justifying the interpretation for muddy and turbid water conditions. Of the benthic forms, the abundance of *Bacillaria paradoxa* was high in the surface sediments between 2009-2018 CE in Green Swamp. This may be reflective of salt input to the wetland, yet such salinity has not been observed in other Gunbower wetlands. As the sedimentation rates increased during the climatic phases of both DDR (1997-2009 CE) and FDR (2010-11 CE), it is suggested that increased sediment influxes cannot be only associated with climate variations but largely linked to catchment disturbance.

To summarise, the Gunbower system was under the influence of droughts (1898-1946 and 1997-2009), high floods (1950s) and rainfall events such as the extreme *La Niña* of 2010-11. As the MDB has experienced large discharges under strong *La Niña* events, causing floodplain inundation and damage to channels, so substantial changes in the wetlands are evident. So, climate induced flow variability has impacted dry and wet cycles of Gunbower wetlands. Further, between 1946 and 1976, the basin was under the influence of a negative PDO phase, so, many wetlands would have been connected more regularly. Based on the Gunbower cores analysis, during FDR, wetlands did not appear to be connected with the main River. So, while there has been multi-decadal and ENSO scale variability, there have been substantial changes in hydrology due to regulation and abstraction for regulation and increased pollution loads. These are sustained changes that are matched by the transition in the diatoms and elements (refer Figures 6.3 and 6.4). The shift to more regular inundation and planktonic dominance in the wetlands is likely a consequence of the artificial elevation of Gunbower Creek, maintained for water supply purposes, which limits drainage, rather than an increase in overbank inputs from the main River. While in Gunbower wetlands, the observed shifts are not episodic, so they are not responding to climate change-induced events but climate change may have added to the impacts of catchment disturbance.

#### **6.4.2 Landscape disturbance and hydrological changes**

In the Gunbower region, the expansion of European settlement in the 19<sup>th</sup> century led to significant changes in the local environment. The European colonisation is considered to be the first substantial disturbance to the local hydrology of the Gunbower site. Further, in 1923, the Torrumbarry weir at Gunbower Creek was constructed to support the intensification of irrigation and agricultural activities of the Gunbower region (see Figures 3.4 and 3.7). Widespread land development and clearing have caused increased soil erosion leading to increased sedimentation in wetlands. This increased accumulation of sediment alters the chemical and hydrologic regime of the wetlands in a relatively short time. So, the Gunbower wetlands, under the influence of permanent inundation by nutrient-

rich waters, became permanent wetlands. It is assumed that the initial land developments in the Gunbower region may have enhanced erosion in the lower sections of its core sequences.

River flow modification via dams and irrigation diversions are the likely causes of substantial transitions in floodplain hydrology. The construction of the Torrumbarry weir altered the natural hydrological patterns that resulted in the reduction of the frequency, duration and magnitude of natural floods (Thoms 2003). At the same time, the use of Gunbower Creek as a channel for the transfer of water for irrigation ensured persistently high-water levels across much of the region. As a result of this shift to artificial hydrology, the wetland ecosystems have been significantly impacted. Changing lake hydrology such as depth can also be inferred in the sediment cores from Fe/Mn ratios. In all studied wetlands, the values of Fe/Mn ratios were consistent and generally high, suggesting hypoxic conditions. However, variability in the wetlands can be inferred from pH changes in the wetlands. In the studied wetlands, species from several genera (e.g., *Eunotia*, *Pinnularia*) indicated circumneutral to acid water conditions suggesting that organic acids influenced the chemistry of the benthic waters. The fluctuations in pH levels are evident from online water quality monitoring data measured at Torrumbarry weir on the Murray River (see chapter 3). It is most likely that the organic acids are derived from the local leaching of humic material (dissolved organic matter) from the floodplain or from submerged macrophytes that host the epiphytic algae. These are balanced by the relatively neutral and alkaline (sometimes acidic) waters from the channels (Gell *et al.* 2018a). The results of pH changes strongly imply that the Gunbower wetlands transitioned to alkaline waters, as occurred at Callamondah billabong (Reid *et al.* 2002), may be suggesting natural succession in the vegetation or the onset of increasing salinity.

The abundance of planktonic diatoms in the studied wetlands at the same time may be attributed to changes in hydrology or sediment influx, suggesting a regional driver of change. For example, at many sites, the increase in planktonic diatoms (e.g., *Aulacoseira* spp.) or facultative planktonic taxa is linked to river regulation (Gell *et al.* 2007; Grundell *et al.* 2012). Unlike other MDB sites, in Gunbower wetlands, the occurrence of planktonic diatoms cannot be just linked with river regulation. The Green and Black Swamp had planktonic diatoms throughout the core, suggesting hydrological variability including deeper water and/or regular connection with the River or Gunbower Creek. However, planktonic diatoms in Cockatoo and Taylors Lagoon cores appear later, suggesting turbid or nutrient rich conditions that favoured plankton or that the river connectivity inferred in the forest was confined to the last ~ 50 years in the dairyland wetlands.

## 6.5 Catchment scale change

The records of the diatom assemblages from the studied wetlands of this middle-Murray site reveal a link between Murray River water input and elevated relative abundance values of *Aulacoseira* species, causing a transition to a turbid planktonic state. Similar transitions were observed in a synthesis of wetland change across the MDB. Many floodplain lakes of the MDB have shifted from clear-water, macrophyte-dominated sites, as evident in epiphytic diatom and cladoceran communities (Ogden, 2000; Reid *et al.* 2007; Kattel *et al.* 2017), to planktonic diatom flora and pelagic fauna. In particular, the large, deep wetlands of the middle reaches were found to be the most vulnerable to this change (Ogden, 2000; Reid *et al.* 2007), yet few studies have identified these changes. So, the present study fills a substantial geographical gap in previous paleolimnological MDB records. According to previous MDB records, the impact of industrial development on the condition of wetlands across the southern Murray-Darling Basin has been extensive and seemingly greater than the impact of millennial-scale climate change. However, many of these records lack multi-proxy reconstructions such as robust dating models, geochemical interpretations, stable isotopes and diatom analysis.

The diatom records of upper sections of Gunbower wetland cores reveal a rise in species that are clear indicators of nutrient enrichment (e.g., *Cyclotella* spp. and *Nitzschia* spp.). This kind of rise in nutrient indicating taxa has been reported for other MDB sites in recent times, reflecting widespread increase in nutrient flux. The water quality monitoring data available does not show these particular trends in nutrient flux over recent decades. Coincident with the rise of eutraphentic diatom taxa is the widespread decline in the relative abundance of *Epithemia* spp. Their decline is evident in wetland records from Mildura (Kings Billabong; Kattel *et al.* 2017) to Blanchetown (Sinclairs Flat; Grundell *et al.* 2012) and this is now also observed in the Gunbower wetlands. The relationship between the Epithemiaceae and nitrogen has not been explored in detail in an Australian context. However, the widespread decline in *Epithemia* species in floodplain records is supporting evidence for a weakening of their competitive advantage after industrial development and increase in ambient nitrogen (Gell *et al.* 2019). Nutrient conditions are shown by several records across the MDB (Figure for example; Loveday Lagoon, Sinclair Flat, Ral Ral Creek, Loch Luna, and Hogan's Billabong have shown substantial increase in *Nitzschia* spp. (Gell and Reid 2016). In addition to this, several other species reflected high nutrient levels such as; *Cyclostephanos dubius*, and *Stephanodiscus* spp. in the lower Murray system. However, in the Gunbower wetlands, the most common nutrient indicator diatoms are *Cyclotella meneghiniana*, *Nitzschia* spp. while *Bacillaria paradoxa* is a salinity indicator was found in surface sediments of Green Swamp core. The diatom assemblage mostly

found in the Gunbower wetlands include a mix of river plankton and local wetland species, and while they support a particular mix of taxa, they can be used to infer similar broad changes in wetland condition, and the drivers of that change, as all sites studied across the southern Basin.

The widespread increase in the flux of fine sediments from the floodplain or channel banks into rivers and then to wetlands is well known (Gell *et al.* 2009; Gell and Reid 2014). However, the actual source of these sediments remains largely unknown, with evidence for subsoil sources (Olley and Wallbrink 2004) and catchment erosion more broadly hypothesized (Reid *et al.* 2007). Certainly, the release of large volumes of sludge into the upper sections of the catchment during the gold rush ensures there is much mobilised sediment to explain these increases (Davies *et al.* 2018). Sedimentation occurred abruptly in some upper-catchment sites such as Hogan's Billabong, where planktonic diatoms appeared soon after increases in sediment flux, as revealed by peaks in magnetic susceptibility and aerophilous diatoms. There were gradual elevations in sedimentation at Sinclair Flat near Blanchetown, where the turnover from benthic to planktonic, turbidity tolerant flora took place from 1950 to 1990 CE (Grundell *et al.* 2012). While it is unclear whether these changes reflect a true state switch driven by internal feedbacks, it is clear that the systems are subject to a sustained increase in sediment input. Rates of sediment accumulation have increased from 5 to 80 times (Gell *et al.* 2006) since gold rush period and, given their shallow nature and the reduction in middle levels floods, the prospects for their long-term persistence are limited (Gell *et al.* 2009). This study shows that this is also the case in the middle Murray River, with increased sediment inputs observed after river regulation in the Gunbower wetlands.

Due to the increased nutrient flux and sediments infilling wetlands, the water clarity and macrophyte abundance has declined (Reid *et al.* 2007; Gell and Reid 2016). While *Vallisneria* appears capable of growing in low light conditions (Blanch *et al.* 1998), many other types are vulnerable. The emergent species around the margins that are evident today are floating (waterlilies), *Azolla* and *Lemna* which have an obvious relationship with *Lemnicola hungarica* (Round and Basson 1997; Emson *et al.* 2018). Like other sites around the Murray Darling basin, this is a substantial change in the character and trophic dynamics of the Gunbower system.

Based on Gunbower wetland sediment analysis, these changes are continuous from top to bottom and identify increased river connectivity, permanency, increased flux of sediments and nutrients as major causes behind the observed catchment scale changes. These systems may have undergone a critical transition whereby feedbacks have become

established that advantage phytoplankton and limit the capacity of submerged aquatic plants to recover. Were this the case, the wetland would remain in a phytoplankton-dominated state after the main stressor (e.g., suspended sediments) was removed (Capon *et al.* 2015). As the scale of the forces that have driven this change preclude the mitigation of these inputs, it remains uncertain whether a critical transition has occurred or whether the sites are impacted by the ongoing supply of nutrients and sediments (Kattel *et al.* 2017).

The structure of this study allowed for the comparison of land use impacts (forest v dairy) and so directly investigated local versus regional effects. The state of all four studied wetlands shifted to phytoplankton dominance, suggesting that all wetlands were affected by the changes witnessed elsewhere, and so responded to regional forces. The two dairy sites however, appear to have sustained higher levels of eutrophication, suggesting a compounding local effect. The shift to plankton occurred in all four sites, although likely not at the same time. This may suggest that they separately resisted a regional impact but ultimately shifted after some time. Given the widespread nature of the evidence for sustained conversion to phytoplankton dominance across Murray River floodplain wetlands, it would seem that there may be an even larger scale driver of wetland state change. In some instances, change was abrupt and early after European settlement (Reid *et al.* 2007), while elsewhere, the transition was slow and ongoing (Grundell *et al.* 2012; Kattel *et al.* 2017). It remains uncertain whether the impacts were imposed at different times across the Basin, or whether different wetlands were more or less vulnerable, and so responsive to those impacts (Gell and Reid 2016).

## **6.6 Implications for the Gunbower Forest Ramsar site**

The Gunbower Forest was listed by Australia in 1982 as a Wetland of International Importance (Ramsar Site) under the Ramsar Convention on Wetlands. The ecological character description at the time of listing was of a periodically inundated, forested floodplain. Its four listing criteria (Hale and Butcher 2011), highlight its quality as a freshwater tree-dominated wetland (criterion #1), its role in supporting five threatened wetland species (#2), its role as habitat for waterbirds and as a breeding site for birds and fish (#4), and its role in providing for the ecological requirements of populations of native fish (#8). Water resource development and climate change are identified as threats to the site, but nutrient and sediments inputs are not (Hale and Butcher 2011).

While the indicators analysed in these sediment records do not directly assess change in the biota that underpin this listing, they do reveal ecological change in the wetlands over time. The state of some of the wetlands at the site then appear to have already shifted, on



account of declining water quality, by the time they were listed under Ramsar. So, this site provides an example where paleoecology has revealed that the long-term changes to a wetland have been underestimated (Pritchard 2021).

Nations have an obligation to report to the Ramsar secretariat if a wetland “has changed, is changing or is likely to change” (Davidson 2016). Australia, being one of the prominent signatories to Ramsar, is obliged to adopt the Ramsar Convention Guidelines for maintaining the ecological character of the wetlands listed under the convention. Kumar *et al.* (2020) have explored some of the important issues of ecological character of wetlands undergoing change (whether human-induced or not). The ecological character is a description of the contemporary ecosystem components of the site and their interactions. While the Gunbower Forest description identifies its present state and contemporary or future threats, it does not record past changes to the water bodies and the influence of changing water quality. So, the longer-term record attests to considerable change, albeit in indicators not noted in the ecological character description (Davidson *et al.* 2019).

This evidence reveals that a deleterious, anthropogenic change has occurred to the ecological character of at least some of the site’s wetlands. That they occurred before listing entertains Pritchard’s (2021) question of what is the right baseline. Should managers adhere to the time-of-listing baseline, then no reportable change has occurred (Gell 2017). But, as Finlayson *et al.* (2016) note, if this was applied universally, it would excuse all the degradation that occurred before the signing of the Convention. In seeking to restore ‘the site’s true value as seen from a longer-term perspective’ (Pritchard 2021), it may be appropriate for the Australian government to identify water quality decline as a threat and to seek means by which the present phytoplankton state, in these and similar wetlands, is returned to one dominated by aquatic macrophytes. Once identified, the restoration process is likely to be challenging as, while the dairy sites are impacted by local sources of pollution, it is clear that there is a regional contribution to the sediment and nutrient load and hydrological regime, that has shifted the state of all wetlands.

## CHAPTER 7- CONCLUSIONS

The paleolimnological and radiochemistry records obtained from the Gunbower wetlands of the middle Murray River have provided high-resolution evidence for the nature of environmental change that occurred due to post-European impacts, particularly after the main phase of river regulation. An understanding of the nature and rate of environmental change is of considerable value for the region. The findings have shown a significant correlation with other paleolimnological studies that have been conducted on the MDB (see Chapter 6) and contribute to our knowledge to understand the basin as a whole system by extending the evidence to the middle sections of the Murray River. This study shows that considerable wetland change has been brought about by post-regulation impacts. The changes can be attributed to catchment scale impacts while local impacts are evident, likely derived from intensive dairy agriculture. This chapter synthesises the work presented in the preceding chapters outlining the extent to which the study aims were achieved. The main findings of the study are described in the following sections.

### **7.1 Objective 1- To use of multi-proxy approach to assess wetland change**

Previous paleolimnological analysis across MDB was mostly based on the proxies such as fossil diatoms, cladocerans and pollen. In this study, in addition to diatoms, the analysis is supported by robust dating, stable isotopes and X-Ray Fluorescence (XRF) stratigraphy records. Hence, with this multi-proxy approach, a detailed record of analysis from Gunbower wetlands has revealed changing wetland conditions before and after river regulation that would otherwise be difficult to recognise using only a single approach. The results obtained from Gunbower wetlands revealed firstly that after regulation, wetlands shifted from intermittent systems that did not continuously accumulate sediments to permanent or semi-permanent systems that have consistently accumulated sediment. Through this time, the sedimentation rates increased and the diatom assemblages revealed the replacement of epiphytic and littoral taxa with planktonic and facultative planktonic forms. This change mirrors the transitions that were observed in a synthesis of wetland change across the MDB (Gell and Reid, 2016). Moreover, due to sedimentation and increased nutrient flux, the water quality and macrophyte abundance declined and the trophic system shifted to a phytoplankton-dominated state as observed in all records of the Gunbower wetlands.

In essence, the possible drivers of the degraded water quality of Gunbower wetlands are identified as the influx of suspended sediment and dissolved nutrients from the main river

system, as well as from local sources. Furthermore, paleolimnological evidence from Cockatoo and Taylors Lagoon suggests that an increase in nutrients is associated with land use activities such as dairy farming, land clearing and associated erosion. Subsequently, increased turbidity of incoming water has caused a loss of submerged aquatic plants. This is further supported by XRF evidence of high allochthonous input to the accumulating sediments.

## **7.2 Objective 2 – To understand wetland changes in relation to past climatic variations**

Although some information about climate change can be interpreted from diatoms, elements and stable isotope records, this study lacks detailed evidence on past climatic variations. This study assesses the extent to which the drying and wetting phases of climate change may have added to the impacts of catchment disturbance. Based on the knowledge about historical river flows (Mallen-Cooper and Zampatti 2018), rainfall data (Bureau of Meteorology 2019), MDB Commission Monthly Simulation Model (Cooling et al. 2002), prior to regulation, under natural conditions, the Murray system experienced drying and flooding at regular intervals. However, post-regulation the system has faced many episodes of drought seasons in between the flood events of 1956-58, 1973-75 and 2010. The frequency of drought has increased recently resulting in dry phases shifting from 1-3 in 10 years to 3-7 in 10 years. Despite this increased drying frequency, the Gunbower Wetlands have been shifted to permanent wetlands owing to the artificial elevation of Gunbower Creek.

## **7.3 Objective 3 & 4 – To compare the ecology and water quality of four wetlands in the last 100 years and explore drivers of these changes: permanence of wetlands and sediment flux**

In the mid-1900s, post-European settlement impacts were observed throughout the MDB and in the Gunbower region that were indicated by major shifts in diatom composition. The records of Gunbower wetlands likely only extend until the post-regulation period except for Taylors Lagoon, which may extend until early 1900. In the cores from all sites, a substantial increase in planktonic diatoms occur that suggest a strong increase in connectivity between all wetlands and the turbid water of the River. So, this change is evident in cores from both forested and dairy lands and provides evidence of regional-scale change. It is likely that a large proportion of the sediment and nutrient loads to all wetlands are derived from the main River and its tributary Gunbower Creek. However, the dairy zone sites have shown greater increases in eutraphentic species (e.g., *Nitzschia* spp.) than the forest

wetlands indicating that local sources are intensifying the elevated nutrient condition of the wetlands.

The lack of sediments that pre-date regulation suggests that net accumulation commenced only after the system was regulated. At Gunbower, the main consequence of water management was to establish Gunbower Creek as a permanent stream maintained for the purposes of delivering water for irrigation purposes. This acted to provide a permanent source of nutrients and sediments to all wetlands, and limit the drainage capacity of wetlands once they became disconnected. So, these wetlands became permanent limiting their capacity to reset ecologically after drying (*sensu* Gell and Reid 2016). The net sedimentation was enabled by the new permanence of the wetlands as previously drying events likely allowed for loss or compaction of accumulated sediments. This would have been exacerbated by the greater influx of sediments, in combination leading to net sediment accumulation of ~ 10 mm/yr.

#### **7.4 Objective 5 - To combine data sets to identify regionally specific environmental changes: possibility of regime shift**

The use of multiple proxies, as demonstrated in this study, has identified clear environmental changes. The combined evidence for increased sedimentation and nutrient loads and change in diatom habitat groups bears witness to the rapid decline in the abundance of submerged macrophytes as the main wetland autotrophs. The shift to planktonic diatom assemblages, and depleted  $^{13}\text{C}$  values in bulk sediments, suggest these plants were replaced by phytoplankton as the basis for the wetland food web. This change is consistent with the shifts evident in large wetlands upstream (Ogden 2000; Reid *et al.* 2007) as well as those in the lower reaches of the Murray River (Grundell *et al.* 2012; Kattel *et al.* 2015; 2017). While such patterns may be explained by critical transitions in the trophic system as shown in wetlands elsewhere by (Wang *et al.* 2012), it remains that the documented changes may be in response to the ongoing chronic influx of sediment and nutrients (Kattel *et al.* 2018). While there may be other factors driving the loss of vegetation in these shallow lakes such as elevated nutrient loading (Jones and Sayer 2003; Sayer *et al.* 2010), increased water levels (Blindow *et al.* 1993), in the Murray-Darling Basin, as indicated by the work of Ogden (2000) and (Reid *et al.* 2002), land disturbance and its impact on water quality is considered as the dominant cause of plant loss in wetlands.

## **7.5 Objective 6 - The benefits of the long-term perspective - Management and restoration**

The retention of the Gunbower Wetlands in the state they were at the time of Ramsar listing (1982) would excuse Australia of rectifying the impacts of degradation that have occurred since regulation in the 1920s. This study suggests that their natural character is seasonal or intermittent as sediment accumulation appears to have commenced on account of more permanent inundation and likely also increased sediment inputs. While this permanent condition allowed for the accumulation of sediments which revealed quite clear, acid-neutral waters in the years after regulation, it is clear there has been a transition to eutrophic and turbid water states.

Given the scale of the catchment and the level of development, it is unlikely that a short-term solution can be implemented to mitigate the unnaturally high flux of nutrients and sediments into this system. Now, the phytoplankton is possibly impacting the light regime in the water column and thereby acting to suppress the recovery of submerged macrophyte plants. The relative permanence of the wetlands also, now, precludes the recovery of this vegetation through drying. If this state is now entrenched, with stabilising feedback effects, then the restoration of the condition of these wetlands poses a greater management challenge than the mere removal of condition stressors. However, the consistency of this transition, even though with variable timing, across most wetlands along the Murray River floodplain (Gell and Reid 2016), suggests that restoration requires basin-wide measures. Remediation at such a scale poses a greater challenge than when cause and effect are proximal.

The information retrieved from the current study has strengthened the spatial extent of previous paleolimnological assessments. With ongoing improvements in dating technology, paleolimnology will go beyond the limits of monitoring and provide evidence for early pre-European conditions and give information about the ecosystem response to long-term changes. This information should be crucial to acquire knowledge of the key mechanisms and processes that are responsible for driving ecosystem change. These records assess the degree and nature of the change since regulation on account of permanent inundation, nutrient and sediment increase, and maybe climate change. In addition to providing the information necessary to establish the degree of impact, knowledge of pre-regulation conditions can also help to establish restoration targets. Where it is not possible to set restoration targets, past conditions will help to better understand the drivers of change to the ecosystem to help establish realistic goals for management.

In essence, this multi-proxy approach provides important baseline data for assessing and managing future land use impacts on water quality and ecology of Gunbower wetlands and for understanding the broader regional effects of river regulation and other anthropogenic impacts. Such knowledge is essential to assess restoration requirements to inform better management practices of these wetlands if their natural pre-disturbance state is considered the appropriate target for management. By documenting the scale and nature of changes that have impacted these wetlands over a century, it may be seen that the original state is unachievable for management (Kopf *et al.* 2015). Here, an 'anthropogenic baseline' (sensu Kopf *et al.* 2015) may be deemed more appropriate; and management measures may be manipulated to recover some ecological elements and functions while falling short of the intermittent, shallow, clear water benchmark states identified for these, and other Basin wetlands.

## **7.6 Future study and recommendations**

This study has highlighted the significance of paleolimnological research in determining the nature of wetland change using a multi-proxy approach. This study has been successful in determining water quality change, elevated sediment accumulation rates and nutrients. However, the stable isotopes of bulk sediments did not provide particularly strong evidence about the source of nutrients. This is due to the limited analysis, with only a few bulk sediment samples processed for stable isotope analysis as well as bulk sediment samples have mixture of organic compounds which are usually difficult to interpret. So, a significant trend of nutrients flux is not identified. Therefore, it is recommended that more samples should be subject to stable isotope analysis to better understand the wetland's nutrient dynamics. Moreover, the chronology of the cores was not compared with the Caesium peak of 1960 that would come from  $^{137}\text{Cs}$  dating. Further, it would be appropriate to apply  $^{137}\text{Cs}$  dating for better identification of the timing of the observed changes and age-determination of core sequences. In addition to diatom analysis, local and regional ecological change can be better estimated by the examination of other indicators. Pollen and pigment analysis would be useful in refining diatom inferred environmental change to understand the impact of water quality and habitat changes. Also, it would be valuable to extract macrofossil remains of plants in the lower sections of the cores to ascertain the nature of the past ecosystems. It will also be beneficial in assessing the vulnerability of certain plant taxa to light regime changes. In essence, information from the plant remains will be a valuable approach to document the plant communities lost in order to evaluate their status, trajectory regionally and to consider them as targets for recolonisation under plans for system rehabilitation.

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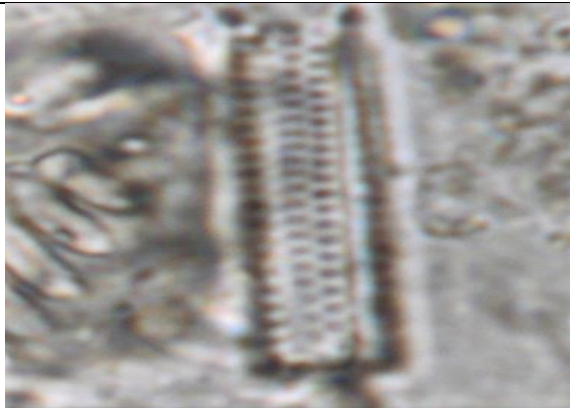
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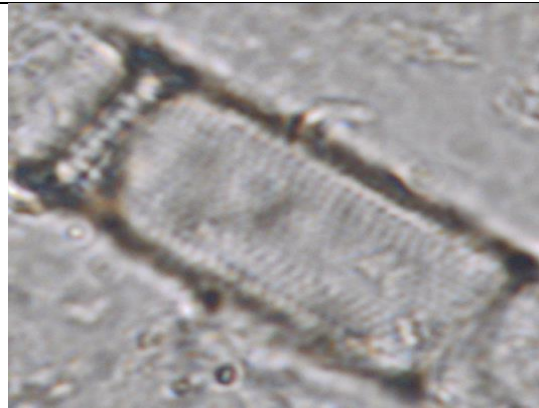
**APPENDIX-I:** List of all diatom species and their taxonomic authorities which attained a minimum relative abundance of 5% or more of the count from the four studied cores

<b>Abbreviation</b>	<b>Species Name</b>	<b>Taxonomic authority</b>
<i>AchnMint</i>	<i>Achnantheidium minutissimum</i>	(Kützing) Czarnecki
<i>ActinSpp</i>	<i>Actinocyclus</i> spp	Ehrenberg 1831
<i>AulcAlpg</i>	<i>Aulacoseira alpigena</i>	(Grunow) Krammer 1991
<i>AulcAmbg</i>	<i>Aulacoseira ambigua</i>	(Grunow) Simonsen 1979
<i>AulGrnSp</i>	<i>Aulacoseira granulata</i>	(Ehrenberg) Simonsen 1979
<i>AulGrnSp</i>	<i>Aulacoseira granulata angustissima</i>	(O. Muller) Simonsen 1979
<i>Aulcltal</i>	<i>Aulacoseira italica</i>	(Ehrenberg) Simonsen 1979
<i>BaclPard</i>	<i>Bacillaria paradoxa</i>	J.F.Gmelin in Linnaeus 1791
<i>CoccPlac</i>	<i>Cocconeis placentula</i>	Ehrenberg 1838
<i>CratcSpp</i>	<i>Craticula</i> spp	(Kützing) Mann
<i>CyclMeng</i>	<i>Cyclotella meneghiniana</i>	Kützing 1844
<i>CyclPseu</i>	<i>Cyclotella pseudostelligera</i>	(Hustedt) Houl & Klee 2004
<i>CyclStel</i>	<i>Cyclotella stelligera</i>	(Cleve & Grunnow) Houl & Klee 2004
<i>CymbAspr</i>	<i>Cymbella aspera</i>	(Ehrenberg) Cleve 1894
<i>DiadConf</i>	<i>Diadsmis confervaceae</i>	Kützing 1844
<i>EncySils</i>	<i>Encyonema silesiaca</i>	Krammer and Lange-Bert. 1986
<i>EpitAdnt</i>	<i>Epithemia adnata</i>	(Kützing) Brébisson 1838
<i>EpitSorex</i>	<i>Epithemia sores</i>	Kützing 1844
<i>EuntSerp</i>	<i>Eunotia serpentina</i>	Ehrenberg 1854
<i>FrgCapGr</i>	<i>Fragilaria capucina gracilis</i>	Desmazières 1830
<i>GomphSpp</i>	<i>Gomphonema</i> spp	Ehrenberg 1832
<i>GyrsAcum</i>	<i>Gyrosigma acuminatum</i>	(Kützing) Rabenhorst 1853
<i>GyrsSpen</i>	<i>Gyrosigma spenceri</i>	(W.Smith) Griffith & Henfrey 1856
<i>LemnHung</i>	<i>Lemnicola hungarica</i>	(Grunow) Round & Basson 1997
<i>NavcCryp</i>	<i>Navicula cryptocephala</i>	Kützing 1844
<i>NitzPale</i>	<i>Nitzschia palea</i>	(Kützing) W.Smith 1856
<i>PinnlSpp</i>	<i>Pinnularia</i> spp	Ehrenberg 1843
<i>RhopGibb</i>	<i>Rhopalodia gibba</i>	(Ehrenberg) O.Müller 1895
<i>SellLaev</i>	<i>Sellaphora laevisissima</i>	(Kützing) D.G.Mann 1989
<i>SellPupl</i>	<i>Sellaphora pupula</i>	(Kützing) Mereschkovsky 1902
<i>StauAncp</i>	<i>Stauroneis anceps</i>	Ehrenberg 1843
<i>StauConst</i>	<i>Staurosira constreuns</i>	Ehrenberg 1843
<i>StauPhoe</i>	<i>Stauroneis phoenocenteron</i>	(Nitzsch) Ehrenberg 1843
<i>UlnrUlna</i>	<i>Ulnaria ulna</i>	(Nitzsch) Compère 2001

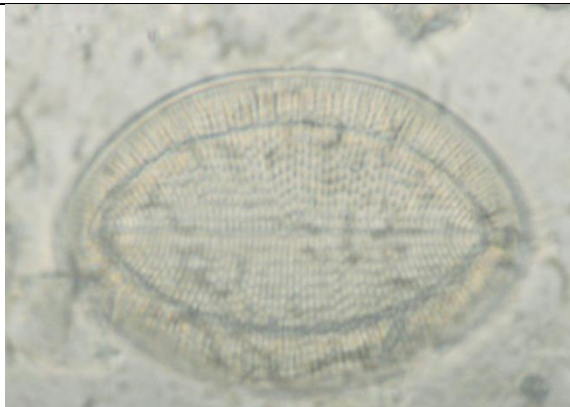
## APPENDIX-II: Diatom images of few dominant species



*Aulacoseira granulata*



*Aulacoseira italica*



*Cocconeis placentula*



*Cyclotella meneghiniana*

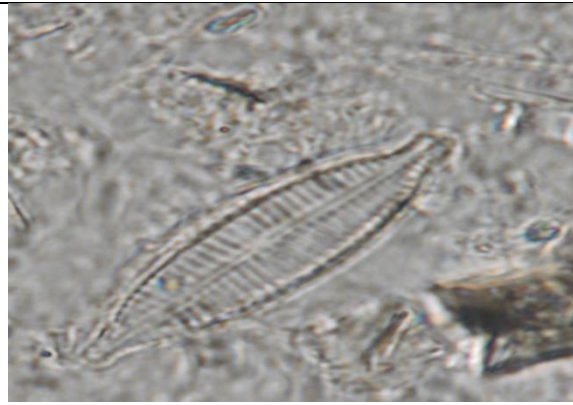


*Diadesmis confervaceae*



*Epithemia sorex*





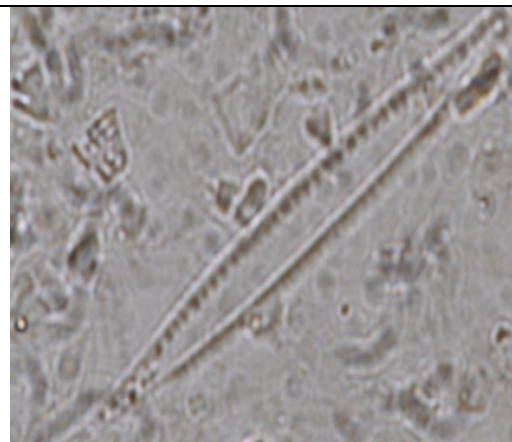
Gomphonema spp



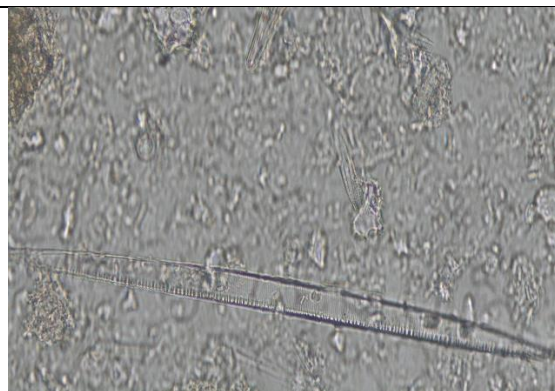
Gomphonema spp



Lemniscola hungarica



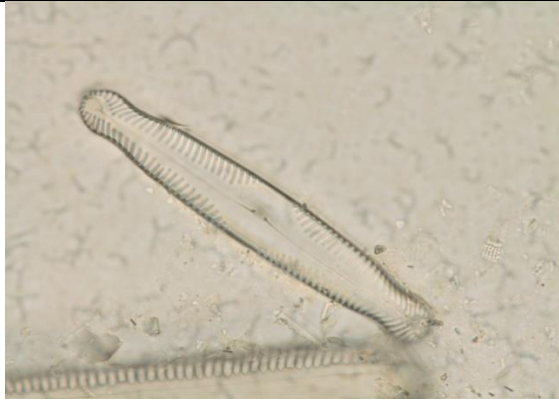
Nitzschia palea



Nitzschia spp



Pinnularia spp



*Pinnularia* spp



*Sellaphora* pupula



*Staurosira* anceps



*Staurosira* construens



*Rhopalodia* gibba



## APPENDIX-III: Pb-210 dating results of all four sites

### Black swamp

ANSTO ID	Depth			Total Pb-210		Supported Pb-210		Unsupported Pb-210 Decay corrected 16-May-18		Calculated CRS Ages	
	(cm)			(Bq/kg)		(Bq/kg)		(Bq/kg)		(years)	
U594	0	-	1	170	± 8	31	± 3	139	± 9	0.1	± 0.2
U595	2	-	3	157	± 7	32	± 3	126	± 8	0.8	± 0.9
U596	6	-	7	161	± 7	17	± 2	145	± 7	4.3	± 2.1
U597	10	-	11	117	± 5	29	± 3	88	± 6	8.7	± 2.9
V104	15	-	16	77	± 2	21	± 2	57	± 3	13.1	± 3.6
U598	20	-	21	94	± 4	29	± 3	65	± 5	17.6	± 4.2
V105	25	-	26	73	± 3	30	± 3	43	± 4	22.2	± 4.7
V106	30	-	31	79	± 3	37	± 3	43	± 4	26.4	± 5.1
V107	55	-	56	42	± 1	14	± 1	28	± 2	53.8	± 7.3
V108	65	-	66	42	± 1	21	± 2	21	± 2	68.8	± 8.3
V109	82	-	83	42	± 1	26	± 2	16	± 3	113	± 11

### Green Swamp

ANSTO ID	Depth			Total Pb-210		Supported Pb-210		Unsupported Pb-210 To Decay corrected 16-May-18		Calculated CIC Ages	
	(cm)			(Bq/kg)		(Bq/kg)		(Bq/kg)		(years)	
U599	0	-	1	115	± 5	35	± 3	80	± 6	1	± 1
U600	2	-	3	107	± 5	63	± 6	44	± 8	4	± 2
U601	6	-	7	35	± 2	14	± 1	22	± 2	13	± 5
U602	10	-	11	88	± 4	32	± 3	56	± 5	20	± 8
U603	25	-	26	72	± 3	36	± 3	36	± 4	45	± 17
V110	30	-	31	71	± 3	66	± 6	5	± 7	65	± 25
V111	45	-	46	73	± 3	17	± 2	56	± 4		
V112	65	-	66	84	± 4	50	± 4	34	± 6		
V113	75	-	76	94	± 4	35	± 3	60	± 5		

### Cockatoo Lagoon

ANSTO ID	Depth			Total Pb-210		Supported Pb-210		Unsupported Pb-210		Calculated CRS Ages	
	(cm)			(Bq/kg)		(Bq/kg)		(Bq/kg)		(years)	
U563	0	-	1	142	± 6	27	± 3	114	± 7	0.5	± 2.0
U564	2	-	3	124	± 5	35	± 3	89	± 6	2.0	± 2.0
U565	7	-	8	108	± 4	42	± 4	66	± 6	5.3	± 1.4
U566	14	-	15	92	± 4	31	± 3	61	± 5	12.5	± 1.7
U567	18	-	19	98	± 4	31	± 3	67	± 5	17.4	± 3.0
U568	24	-	25	95	± 4	29	± 3	66	± 5	24.4	± 3.7
U569	35	-	36	58	± 2	26	± 2	31	± 3	39.7	± 3.6
U570	50	-	51	36	± 2	24	± 2	12	± 3	70.4	± 12.4
U571	74	-	75	26	± 1	29	± 3	-3	± 3		

### Taylor's Lagoon

ANSTO ID	Depth			Total Pb-210		Supported Pb-210		Unsupported Pb-210 Decay corrected date 15 May 2018		Calculated CRS Ages	
	(cm)			(Bq/kg)		(Bq/kg)		(Bq/kg)		(Years)	
U589	0	-	1	175	± 8	12	± 2	164	± 8	0.1	± 2.1
U590	1	-	2	161	± 7	31	± 3	131	± 8	0.4	± 1.8
U591	5	-	6	182	± 8	25	± 2	158	± 8	1.9	± 1.8
V096	7	-	8	153	± 8	20	± 2	135	± 8	2.9	± 1.8
V097	9	-	10	172	± 9	29	± 3	144	± 10	3.8	± 1.8
U592	15	-	16	131	± 6	46	± 4	85	± 7	5.8	± 1.8
V098	21	-	22	136	± 6	45	± 5	92	± 8	8.4	± 1.8
V099	25	-	26	117	± 5	46	± 5	72	± 7	10.3	± 1.8
U593	30	-	31	157	± 7	44	± 5	113	± 9	12.6	± 1.9
V100	60	-	61	138	± 7	45	± 4	94	± 8	44	± 2.3
V101	70	-	71	47	± 2	17	± 2	31	± 3	72.4	± 3.2
V102	81	-	82	23	± 1	12	± 1	11	± 1	105	± 4.9
103	100	-	101	38	± 2	21	± 2	17	± 3		

## APPENDIX-IV – Research publication

publish.csiro.au — Private

### Just Accepted

This article has been peer reviewed and accepted for publication. It is in production and has not been edited, so may differ from the final published form.

A multi-proxy approach to track change in ecological condition of wetlands in the Gunbower Forest – A Ramsar site

NEERAJ MALL , Peter Gell, Professor Giri Kattel , Patricia Gadd, Atun Zawadzki

### Abstract

Gunbower Forest is bordered by the Murray River and Gunbower Creek, and hosts several floodplain wetlands listed under the Ramsar Convention. Sediment cores were retrieved from three wetlands to trace changes to their ecological state over time. The basal sediments of the wetlands date back to the beginning of river regulation in the 1930s suggesting that only after then were they inundated sufficiently often to allow for net sediment accumulation. The diatoms preserved in the lower levels of all cores suggest clear, freshwater conditions prevailed during that period. Increased sediment and nutrient loads are inferred by increased epiphytic forms and nutrient indicators. Over recent decades, the wetlands have transitioned to plankton dominance reflecting greater connectivity to the river and distributary, and a reduced light environment. This pattern resembles to that recorded both upstream and downstream suggesting a regional scale change in the wetlands of the southern Murray Darling Basin.

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