1 Background

1.1 Introduction

Wetlands are increasingly becoming valued and used for some of the functions or services they provide. Costanza et al. (1997) prepared a study on the value of the world’s different ecosystem services, wetland services scoring the highest of all terrestrial and aquatic ecosystems. The services and functions offered by wetlands can be broadly divided into 3 categories (Anonymous 1995; Morris 1991; Scheffer 1998). The first of these is hydrologic or flood amelioration, where wetlands can act in aid of short-term surface water storage, long-term surface water storage, or the maintenance of a high water table. The second is the preservation of flora and fauna habitat and associated food webs, through the maintenance of characteristic plant communities and characteristic energy flow. The third is biochemical or nutrient and sediment uptake, where wetlands can be involved in the transformation or the cycling of elements, the retention or the removal of dissolved substances, and the accumulation of inorganic sediments.

Not all functions of wetlands are regarded as an asset; the value of a wetland function is usually only then recognised when useful or required services have been identified. However, a wetland function that presently does not have a recognised value may obtain one in the future. For example, the value of maintaining water quality by a small wetland may not be recognised until it is acquiring a relative greater percentage of representation in the area, or if it is close to a drinking water source (Anonymous 1995). Maintenance and restoration of wetlands and associated aquatic environments should therefore have a high priority for sustainable development.

Whereas the flood amelioration and preservation of habitat and biodiversity has seen ongoing recognition, the nutrient uptake and sediment uptake has started gaining a greater significance than previously was the case due to the loss of wetland function. This project focused primarily on the nutrient uptake aspect of wetland function, although the potential management interventions simulated aimed at rehabilitating degraded wetlands are also expected to contribute to wetland biodiversity and habitat availability rehabilitation.
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From an anthropocentric standpoint, there are a number of reasons for improving the management of wetland function and the resources or services they provide. Freshwater habitats have a very important role in sustaining human activities (Burbridge 1994). The natural functions of wetlands produce a range of resources, which affect the economic and social welfare of a diverse range of people. With the degradation of wetlands these resources are being severely and adversely affected (Burbridge 1994).

One justification for reversing the trend of degradation of wetlands is that the sum of the services provided by the functioning of wetlands, which include economic and social values, is of a greater value than can be gained from degraded or converted wetland use (Burbridge 1994; Costanza et al. 1997; Pimm 1997). Furthermore, the function of a number of small wetlands may not be recognised until their cumulative capacity is fully understood. For example, swamp reclamation or flood amelioration can also lead to wetland reduction or even destruction; with a decrease in overall wetland area, reduction in average size, total numbers, linkage and density, the cumulative function of wetlands will decline (Anonymous 1995; Johnston et al. 1990; Preston et al. 1988). Therefore, the functions of wetlands, which include the uptake and storage of nutrients and sediment retention, will have an impact on a landscape scale through the improvement of water quality.

The primary driving force of nutrient exchange, being the flow of nutrient into and out of a wetland, is through the water flow between the wetland and the river. The model developed in this project used a nutrient balance simulation within a wetland to calculate this exchange rate, thereby elucidating a significant unknown for wetland management. The process, by which the model assesses the exchange rate, is simplified in Figure 1.
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Figure 1: Wetland exchange modelling

To understand the impacts that wetland functions have within a catchment and the implications of management of wetlands (or of reduction of wetlands, i.e. continued wetland loss or degradation), an evaluation of the cumulative impact of the functions of multiple wetlands is required. Landscape scale modelling of the wetland processes and associated functions, such as nutrient retention by healthy wetlands or lack thereof in degraded wetlands, would contribute to knowledge and understanding and therefore provide information for decision making. A model that can be applied generically across multiple wetlands can be used to assess the cumulative nutrient retention estimate on a landscape scale; Figure 2 represents the use of a model in such a scenario.

Figure 2: Cumulative assessment of wetland processes
The riverine ecology system is still inadequately understood (Young et al. 2000), complicating the issue of aquatic modelling. However, even with limited understanding and data resources, it is possible to develop an aquatic model to test hypotheses of wetland function and management; and to improve general understanding. To identify the processes required within a wetland model and be aware of the interactions these processes have both within the wetland as well as externally and appreciate some of the issues affecting water quality it is necessary to examine some of the wetland characteristics in detail. The complex interactions between sedimentation, re-suspension, turbidity, eutrophication, primary producers and consumers are to varying extent considered in the model developed during this project, and are therefore briefly discussed below in reference to the study area.

This project focuses on the floodplain wetlands of the lower River Murray, the South Australian section of the Murray-Darling Basin in Australia (see Figure 3). The catchment area is approximately 1 million km$^2$ or approximately one seventh of Australia (Hills 1974; Walker 1985; Walker et al. 1994). The headwaters comprise of only 500 km of the 2560 km of the river (Mackay et al. 1990; Roberts et al. 1991; Walker 1985), which has a total floodplain area of approximately 10,000 km$^2$ (Roberts et al. 1991). The approximately 2,000 km of river floodplain section has a very shallow gradient with a drop of mere centimetres over distances of kilometres (Mackay et al. 1990; Walker 1985). The average annual runoff is approximately 11,000 GL but can vary from 2,500 GL in a dry year to 40,000 GL in a wet year (Mackay et al. 1990; Walker 1985).
1.1.1 Wetland processes

Wetlands are complex ecosystems with numerous interactions which link to separate aquatic systems (river, creeks, drainage flow paths etc.), terrestrial systems such as the surrounding riparian zone and atmosphere. The complex interactions such as between primary producers, consumers, predators and their feedback loops; as well as the multiple sources and losses of nutrient and energy, can make full accounting an impossible task in wetland assessment and therefore modelling seem an impossibility.
Of the interacting facets within a wetland some can however be focused on to obtain an understanding of the function of the wetland. The ones that are seen as the major processes or facets within a wetland are discussed below.

**Sedimentation**

Any wetland processes that act to decrease waterborne sediment and nutrient concentrations are considered to benefit water quality (Johnston 1991). Sedimentation and sediment re-suspension are processes that operate continually in wetlands and can have an impact on the nutrient availability and wetland turbidity. Increasing sedimentation and decreasing sediment resuspension would, through their impact on improving water quality, be seen as part of rehabilitation. That is, wetland turbidity and consequent nutrient availability affect the state of wetlands and the primary producer (phytoplankton and macrophyte) composition. In a sequence of events, the state of primary producers, along with turbidity and nutrients, compound the impacts on water quality within wetlands, i.e. self regulating processes.

**Turbidity**

Turbidity in a wetland can effectively shade out the incoming light, thereby minimising the underwater light availability. Walker and Hillman (1982) have found that even in eutrophic waters of the River Murray high turbidity can restrict primary productivity. The high turbidity is therefore an important factor controlling plant growth in River Murray wetlands (Walker et al. 1982). The reduction of turbidity particularly within wetlands is consequently seen as a major management focus. The Secchi depth of water bodies (an indication of turbidity) is increased both through an increase in suspended matter and the high nutrient flux from the sediment, which also stimulate the algal production (Soendergaard et al. 1992).

**Nutrients**

Dissolved and particulate inorganic nutrients such as phosphorus, nitrogen and silica are a natural part of the water content in rivers. In excess, these substances become pollutants and contribute to growth of phytoplankton and other aquatic plants (Shafron et al. 1990). Laboratory studies have shown that the release of phosphorus can be increased 20-30 times in a resuspended sediment compared to that of an
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An undisturbed sample (Soendergaard et al. 1992). Such increased phosphorus levels can lead to eutrophication of wetland water.

Phytoplankton

The increased growth of phytoplankton caused by eutrophication contributes to an increase in turbidity and a decrease in water transparency of the water column, limiting light penetration and therefore submerged macrophyte growth. Phytoplankton, detritus and resuspended inorganic sediment particles thereby contribute to lake turbidity (Scheffer 1998). The algal blooms tending to increase turbidity of the water column both through their presence and because macrophytes are effectively shaded out, their contribution to sedimentation thereby is lost. Binding of sediment through compaction and/or minimisation of resuspension is therefore an important management objective.

Macrophytes

Macrophytes are not only a part of the primary productive activity of wetlands but also contribute to its self regulated maintenance. For example, established macrophytes have been said to function as biological engineers as they act as buffering systems in wetlands and have a large role in maintaining a clear state (Sand-Jensen 1998; Stephen et al. 1998). Some of the ‘engineering characteristics or mechanisms’ include the reduction in flow velocity, the stabilisation of the sediment and the provision of habitats for micro-organisms, invertebrates and fish (Carpenter et al. 1997a; Sand-Jensen 1998).

Biota such as macrophytes contribute to the long-term storage of nutrients, with some residual accumulating in newly formed soils (Graneli et al. 1988; Kadlec 1997). Further, macrophytes can become permanent sinks of phosphorus through the burial of plant litter (Graneli et al. 1988). Some benefits of macrophytes include habitat provision for zooplankton which feed on phytoplankton (Baldry 2000; Stephen et al. 1998; Timms et al. 1984), uptake of nutrients (Chen et al. 1988), including luxury uptake and enriched denitrification (Meijer et al. 1994; Stephen et al. 1998). Reduced chlorophyll-a (i.e. phytoplankton) has been found to occur close to macrophyte growth. This has been associated to the presence of zooplankton which find a refuge within the macrophyte growth (Stephen et al. 1998). The role of macrophytes in
supporting zooplankton and therefore control of phytoplankton can therefore be a significant aspect in wetland management.

Macrophytes also reduce water movement (turbulence) and therefore reduce re-suspension and increase sedimentation; they can also shade benthic algae and phytoplankton (Mitchell 1989; Sand-Jensen et al. 1988; Stephen et al. 1998). Sand-Jensen and Mebus (1996) showed a steep reduction in flow velocity within dense macrophyte growth. The lower energy environment above the sediment within the macrophyte patches leads to a retention of fine sediment and organic matter, carbon, nitrogen and phosphorus (Chambers et al. 1994; Sand-Jensen 1998; Sand-Jensen et al. 1992; Sand-Jensen et al. 1996). Effectively the sedimentation within macrophyte beds reduces the transportation of nitrogen, phosphorus and other particles downstream (Sand-Jensen 1998). Therefore, a healthy wetland with a large macrophyte biomass should self-propel a reduction in turbidity and nutrient retention. Due to the many and diverse mechanisms provided by the macrophytes they are recognised as a key step in restoring wetlands (Meijer et al. 1994; Stephen et al. 1998).

Macrophytes obtain phosphorus from the surrounding water and the substrate, with minimal release found in actively growing macrophytes (Graneli et al. 1988). However, decaying macrophytes can account for a substantial contribution of phosphorus to the open water (Graneli et al. 1988). The growth and decay of macrophytes will therefore have an impact on the phosphorus balance of an aquatic system.

Macrophytes affect nutrient levels in wetlands in more ways than just uptake and sedimentation. For example, phosphorus release may also be reduced through oxidation of the sediment (Stephen et al. 1998). Macrophytes readily take up soluble nitrogen from recycling processes (Stephen et al. 1998). Macrophytes also serve as a bottom up control mechanism of nitrogen both through uptake and denitrification (Carpenter et al. 1997a; Stephen et al. 1998). Macrophytes also influence the nitrogen cycle by increasing water residence time and therefore enhancing the denitrification cycle. This can be up to 3 times otherwise expected due to the organic enrichment among rooted macrophytes (Sand-Jensen 1998). Effectively, macrophyte biomass contributes to nitrification and denitrification within shallow water bodies and therefore plays a significant role in the nitrogen budget (Caffrey et al. 1992).
River flow

Seasonal changes in nutrient and turbidity levels are influenced by river flow behaviour. As a result of decreased flow and increased nutrient availability the impounding of water (e.g. instillation of the locks in South Australia) will possibly favour the growth of phytoplankton leading to algal blooms (Shiel et al. 1982; Walker 1979). However, despite the eutrophic conditions there may be some limiting of algal production due to the turbid waters of the lower River Murray (Walker 1985). The turbid conditions may however not be limiting to *Anabaena* as it is able to control its buoyancy thereby increasing its light harvesting potential (Baker et al. 2000). Nutrient control to manage algal blooms would in this case be a significant management achievement.

Zooplankton

Nutrient availability (N and P) may determine the potential algal biomass production, however zooplankton grazing can have a large role in determining the biomass balance (Stephen et al. 1998). Of the zooplankton in the lower River Murray, the most common are indicative of eutrophic conditions, some of which are influenced by temperature changes, turbidity and salinity (Shiel et al. 1982), reflecting the state of the system. The zooplankton grazing rates can be correlated positively to water temperature and have a negative impact on phytoplankton biomass, i.e. chlorophyll-a (Kobayashi et al. 1996; Schwoerbel 1993). Studies by Griffin et al. (2001) showed that zooplankton grazing had a significant impact on phytoplankton biomass. They found zooplankton biomass peaks follow that of the phytoplankton biomass peaks, which is a typical Lotka-Volterra predator-prey cycle (Griffin et al. 2001). The degree to which zooplankton impacts on phytoplankton biomass is dependent on the zooplankton species as well as the species and size of phytoplankton (Schwoerbel 1993). Zooplankton are therefore an important constituent within wetlands playing a role in stabilising phytoplankton growth. They are therefore a significant aspect to consider as part of management.
1.1.2 Spatial relationships of wetlands to transport processes

*Flow and flood regulation*

The seasonal distribution of flow has been changed by flow regulation of the River Murray. The winter flows have decreased as surplus water is taken into storage, and the summer flows have increased as irrigation demands are met (Walker 1979). There has also been significant flood amelioration; that is, through water retention of some of the surplus water followed by controlled release, the severity and incidence of flooding has been reduced significantly (Walker 1979).

Flood regulation drastically affects aquatic environments, including the reduction of interaction between all but the closest wetlands to the river (Walker 1979; Walker *et al.* 1993). This renders wetlands, which are not adjacent to the river, dry due to the lack of periodic flooding thereby virtually eliminating these wetlands. The wetlands closer to the river remain for the most part permanently inundated with associated consequences (e.g. lack of sediment compaction and lack of macrophyte regeneration). Flow regulation has consequently been widely recognised as a major contributor to river and floodplain wetland degradation (Arthington *et al.* 2003; Bunn *et al.* 2002; Walker 1979; Walker 1985).

*Nutrient retention and Exchange capacity*

The natural retention of nutrients in wetlands occurs by cumulative fluxes into storage compartments of the wetland ecosystem. These compartments include the soil, vegetation and plant litter (Johnston 1991). Through their retention of nutrient, wetlands act as sinks of waterborne nutrients and thereby act to improve the water quality (Johnston 1991). The impact that a wetland sink or storage compartment has on the water quality depends on both the rate of nutrient uptake and the retention time (turnover rate) (Johnston 1991; Kadlec *et al.* 2001). The flow of water through a wetland therefore controls the nutrient transport into the wetland as well the nutrient transport out of the system, see Figure 1. The nutrient retention of the wetland is significantly determined by the water residence time that is controlled by the flow speed, wetland size and linkage to the river.

The proximity of a wetland to the river as well as the wetland shape, size, depth and volume can have a substantial impact on the effectiveness of the function of the
wetland in the landscape; this impact can be influenced by the exchange capacity as well as residence time within in the wetland. The exchange capacity can be impacted on by channel volume, shape and length or by such factors as the location of the wetland in the landscape. The location of the wetland in the landscape relating back to variables such as wind direction, which in the case of the lower River Murray plays a significant role in the flow direction and flow rate of the river (Webster et al. 1997). The depth, area and volume of the wetland itself will also impact on the exchange of water between the wetland and the river, for instance wind can push the water in a large shallow wetland away from the connection channel; or evaporative processes can be influenced by the volume and surface area of a wetland.

The transport of material in and out of wetlands is primarily a function of water flow (Johnston 1991). That is, the exchange rate has an impact on the exchange of nutrients and transport of salinity between wetlands and the river. These aspects must therefore be taken into consideration for wetland management, however obtaining exchange data can be prohibitive due to cost or the complexity of environmental factors mentioned. Consequently, any method of obtaining an estimate of exchange between the wetland and the river will be a valuable tool in wetland management and a significant addition to budgeting aspects of nutrient and salinity impacts to the river. This estimation of the transport of material by river exchange has the potential of being the most significant external influence acting upon a wetland and a wetlands impact on the river, and its estimation is currently a significant data gap.

Despite the size, shape and position of wetlands in the landscape having a potentially large influence on the functioning of wetlands, these parameters are infrequently measured. These wetland properties can however be radically changed by human influence (Preston et al. 1988). Therefore, an understanding of how the wetland properties influence wetland functioning on a landscape scale is relevant to restoration and management decision-making. Scenario analysis of different wetland properties may therefore assist in increasing this understanding.
1.2 Degradation of floodplain wetlands

Water quality is a key indicator of river and wetland health, and of wetland functioning. Maintenance of good water quality helps to prevent further degradation of wetland and riverine ecosystems. River and wetland water quality need to be maintained in the interest of primary industry as well as water supply for urban environments.

The River Murray basin accounts for a large part of Australia's agricultural production. The demands of settlement and land use have placed considerable pressure on the river system, resulting in a decline in biodiversity and aquatic habitats and therefore altered the structure and function of river and wetland ecosystems. As a result the water quality of the lower River Murray, which covers an approximately 650 km stretch of the river in South Australia, has drastically diminished.

The River Murray is often viewed as the lifeblood of South Australia, the driest state on the driest continent, and water quality is a significant issue for its inhabitants. The River Murray is a significant water source for South Australia. The city of Adelaide derives between 55% and 90% of its water from the River Murray, and other South Australian towns, including those of the 'Iron Triangle' (Whyalla, Port Augusta and Port Pirie), receive up to 90% of their water supply from this source (Jacobs 1990). Agricultural areas along the River Murray use it as a primary water source for crop irrigation, as there is very little rainfall in these areas. Other uses of the River Murray within SA includes tourism (camping, fishing, house boats and other cruises) and commercial fishing.

Wetlands perform important services and functions for river water quality, such as accumulating nutrients and trapping sediments (Anonymous 1995; Johnston 1991; Mitsch et al. 2000). Wetlands also act as habitats for a wide range of flora and fauna (Boon et al. 1997; Recknagel et al. 1997). It is therefore imperative to restore and/or maintain the structure and functions of wetlands, such as nutrient retention.

Of the wetlands along the River Murray few, if any, can be considered to be pristine environments (Walker 1979). Due to the present regulation of the flow regime of the River Murray, the development of new wetlands (billabongs is reduced significantly (Walker 1979). Therefore, it is becoming increasingly important to preserve, maintain
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and manage the remaining diversity, and significant areas of flood-plain habitats (Walker 1979). It has become increasingly recognised that rivers and wetlands are legitimate users of water (Arthington et al. 2003; Naiman et al. 2002), with government departments, such as the South Australian Department of Water, Land and Biodiversity Conservation (DWLBC), recognising their role in preserving and restoring ecological processes and ecosystems. Legislation for the protection of aquatic ecosystems, such as the Water Resources Act 1997 and as amended by the Natural Resources Management Act 2004, shows the progress towards the recognition of the importance of aquatic ecosystems such as wetlands.

As it is, there have been biological changes to the lower River Murray and its floodplain wetlands due to the introduction of the locks (Walker 1985). Effectively the river has been replaced by a series of cascading pools, which due to their difference from the normal river flow encourage a change in biodiversity such as plant community composition towards exotic species. These fish and plant species being more accustomed to permanent inundation and slow flowing pools (Pressey 1987; Walker 1985). Along the River Murray there are more than 100 different storages (Walker 1985), the lower River Murray wetlands are therefore to a large extent now either permanently inundated, or above pool level left dryer for a longer period than before (Pressey 1987). The river regulation has affected the riparian vegetation by disrupting regeneration and affecting the mature period (Roberts et al. 1991). Due to the lack of periodic flooding black box (*Eucalyptus largiflorens*) communities are showing a reduction in numbers, the river red gum (*Eucalyptus camaldulensis*) is not regenerating in significant numbers and in many areas there is a significant dieback due to drowning (Walker 1985).

Some of the problems contributing to water quality degradation in the River Murray are associated with changes in catchment condition due to land use in the Murray Darling Basin over the past 100 years. The increased nutrient load in riverine water has led to an increase in algal growth and conversely a decrease in water quality. River and wetland management therefore has an important role in preserving a very significant resource for Australia.
1.2.1 Eutrophication of aquatic environments

A full understanding of wetland eutrophication is still in its infancy (Keenan & Lowe 2001). However, research has shown two alternate stable states exist for shallow water bodies; that of the turbid phytoplankton-dominated state and the clear water macrophyte-dominated state (Blindow et al. 1993; Boon et al. 1997; Scheffer 1998; Scheffer et al. 1993; Stephen et al. 1998). A wetland in one state will tend to remain so due to a number of buffer mechanisms (Boon et al. 1997; Moss 1990; Stephen et al. 1998), but with an increase in nutrient loading to a system, a wetland may change from a clear to a turbid state (Boon et al. 1997; Scheffer 1998). A reverse change can be difficult to obtain, however changes in water level and the removal of a part of the fish stock have been used as successful restoration approaches in returning wetlands to a clear state from a turbid one (Scheffer 1998).

In the River Murray catchment, agricultural development (land clearing, irrigation and pasture management) has caused substantial increases in the river sediment load (Walker 1979). There is also an increase in the organic, and nutrient load to the river brought on through agricultural practices and as a consequence of loss of buffering activity of the cleared vegetation (Lijklema 1994). This, combined with the turbidity and sediment deposition downstream, affect the water quality and habitat suitability of the river and its wetlands. Through leaching of nutrients from fertilised and irrigated surrounding farmland, some wetlands of the River Murray floodplain have become eutrophic.

This eutrophication, combined with turbid waters and degraded systems (e.g. by permanent inundation, or the presence of exotic species such as carp), has turned the wetlands into a turbid, algal dominated state where phytoplankton out-competes macrophytes, leading to algal blooms (Scheffer 1998). Some of the nutrients increased in the river are phosphorus and nitrogen. Both are essential nutrients for plant growth, but in excessive amounts they can reduce water quality through eutrophication, algal blooms, decreased light penetration and loss of dissolved oxygen of the water body (Marsden 1989). Eutrophication can also contribute to the reduction of macrophytes due to the shading impact of increased phytoplankton and can force a wetland into a turbid state (Asaeda et al. 2001; Graneli et al. 1988; Scheffer 1998). Therefore, increased eutrophication can alter the species composition of a wetland (Johnston 1991) and therefore change the function of a wetland.
In most wetlands of the lower River Murray, phosphorus and nitrogen concentrations exceed the limit of what is considered critical for eutrophication, reflected in the high chlorophyll-a concentrations in the water columns (up to 256 mg/l) (Boon et al. 1997; Goonan et al. 1992). Boon et al. (1997) concluded that nutrient enrichment poses a significant threat to the ecological integrity of wetlands throughout Australia. Management of nutrients in the landscape can therefore have an impact on a large range of ecosystems. Using wetlands or at least managing wetlands to fulfil the function of nutrient retention can thereby be a strong tool to their own preservation.

The sources of nitrogen to wetlands include external inflow as well as fixation of gaseous N₂ that is converted into organic nitrogen (Johnston 1991). The removal of nitrogen from wetlands however often occurs through a process called denitrification where nitrogen is released into the atmosphere (Bowden 1987; Kadlec et al. 2001; Mitsch et al. 2000; Morris 1991; Reddy et al. 1989; Scheffer 1998; Schindler 1977). Therefore, the concentration of nitrogen or NO₃ is in a continual state of flux depending on the rate of nitrification and denitrification.

Unlike nitrogen, phosphorus cannot escape a wetland system. It therefore remains in the system and is recycled. Phosphorus uptake in wetlands is regulated by physical, (e.g. sedimentation), and biological mechanisms, (e.g. uptake and release by vegetation) (Kadlec et al. 2001; Schindler 1977). The measurement of phosphorus within the wetland is therefore more stable and indicative of availability.

Nitrogen and phosphorus can play a significant role in the eutrophication of wetlands (Reddy et al. 1995). Phosphorus is the major limiting nutrient to nitrogen fixing algae such as Anabaena (Schindler 1977) whereas increased nitrogen concentration can contribute to a shift in species composition within wetlands (Morris 1991; Schindler 1977). They are also both the most likely nutrients to limit primary productivity within wetlands (Baker et al. 2000; Beardall et al. 2001; Hecky et al. 1988; Morris 1991; Oliver 1993; Schindler 1977; Walker 1979; Walker et al. 1982).

Total phosphorus includes crystalline, occluded, adsorbed, particulate organic, soluble organic and soluble inorganic phosphorus. However, not all of this phosphorus found in a water body is biologically available. The biologically available phosphorus includes the soluble reactive phosphorus (entirely biologically available), soluble unreactive phosphorus (available through enzymatic hydrolysis) and labile
phosphorus (available through desorption) (Holtan et al. 1988). The main anthropogenic input of phosphorus is through fertilisers and detergents (Holtan et al. 1988). In a system such as the lower River Murray, the phosphorus source to a wetland can be almost exclusively through fertiliser or irrigation drainage runoff, whereas sediments can act as sinks of phosphorus (Holtan et al. 1988). The phosphorus, which is found in the sediment, is to a great extent sorbed to soil particles or as part of organic matter (Holtan et al. 1988) reducing its availability. This biologically unavailable phosphorus, found in the sediment, can be released through various mechanisms such as turbulence, animal activity (bioturbation), and plant growth (Scheffer 1998), thereby becoming biologically available. Sediment can therefore in circumstances contribute to the maintenance of eutrophication in a water body where inflow may have been reduced (Lennox 1984; Lijklema 1994; Nürnberg 1984; Nürnberg 1998; Olila et al. 1995; Recknagel et al. 1995). Recknagel et al. (1995) found through simulations of Lake Mueggelsee that the best management for the reduction of eutrophication was in fact sediment dredging. In a different approach as in the case of lower River Murray wetlands, sediment compaction (drying of wetland beds) to minimise resuspension and consequent release of bound nutrients is a management strategy currently employed.

The internal phosphorus loading of the sediment is a significant factor in internal loadings once external loading has been reduced. However, van der Molen (1994) found that including this in the model of phosphorus concentration did not significantly improve the predictive capacity of their model for shallow lakes which experienced high external phosphorus loadings. In shallow lakes, where the external loading was reduced, the sediment water exchange of phosphorus became significant in estimating the variability experienced. This indicated that the external source was the significant contributor of phosphorus concentration. Their hypothesis is that in shallow lakes with significant external phosphorus loadings the sediment water interaction is held in an equilibrium (van der Molen et al. 1994). It can be assumed that this is also the case in wetlands, phosphorus simulation in eutrophic lower River Murray wetlands can therefore concentrate on the external and suspended phosphorus and disregard the impact of current sediment released phosphorus.
1.2.2 Alternate stable states and permanent inundation impacts on wetlands

To maintain natural ecological integrity rivers, floodplains and their wetlands need their natural flow regime in its full spatial and temporal variability (Arthington et al. 2003; Bunn et al. 2002; Poff et al. 1997). The wetlands, as part of their ecological function, provide resilience mechanisms by which extreme events are buffered. Some of these have been discussed above, such as phosphorus and nitrogen loads, the role of macrophytes, plankton (phytoplankton and zooplankton) and flow regime. All of these complex interactions, many of which have not been described, act to provide the wetlands with a certain resilience mechanism. However, with the destruction of these resilience mechanisms new resilience mechanisms develop to adapt to the new state of the ecosystem (Carpenter et al. 1997a; Ludwig et al. 1997; Scheffer et al. 1993).

The change of wetlands from one buffered state to another is due to the resilience being overcome by an extreme event. Such a change will transform an aquatic ecosystem, such as a wetland, from one stable state to another (Carpenter et al. 1997a; Carpenter et al. 1997b). The change can be driven by a complex interaction of eutrophication, loss of macrophytes, water regime and turbidity, or by extreme events for any of these (Carpenter et al. 1997a; Carpenter et al. 1997b; Scheffer et al. 1993). The two states can be seen as alternate stable states of clear and turbid (Scheffer et al. 1993). Through river regulation many of the lower River Murray wetlands have degraded to the turbid state reducing the function of the wetland in the landscape.

Returning a wetland to a clear state, once it has switched to a turbid one, can be more complex than reversing the cause (Scheffer et al. 1993). For example, eutrophication contributes to changing a shallow aquatic ecosystem such as a wetland from a clear stable state to a turbid one. Reducing the nutrients may however not bring the wetland back to a clear state due to the resilience of the alternate turbid stable state, which acts through the buffering release of nutrients from the sediment or resuspension as winds are not reduced by the otherwise present macrophytes (Scheffer et al. 1993). Management of these wetlands could lie in the forceful change from one state to another, such as the reduction of turbidity. This would induce macrophyte growth through increased light availability that then reinvigorate the resilience of the clear stable state (Scheffer et al. 1993). Scheffer et al. (1993) suggest the reduction of
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turbidity through either the management of fish stock such as carp or the management of water levels to induce macrophyte growth.

Australian wetlands do not need constant inundation, and in fact their constant inundation is detrimental. Drying and refilling of wetlands are natural processes in Australian wetlands to which the flora and fauna are adapted and dependent (Pressey 1990). Permanent inundation reduces the growth and regeneration potential of ephemeral vegetation common to River Murray floodplains (Nielsen et al. 1997), and the lack of periodic flooding, due to river regulation, may contribute to the lack of regeneration of terrestrial vegetation.

This permanent inundation of wetlands resulting from the regulation of river flow has favoured invasion by the common carp (*Cyprinus carpio*). The feeding habits of carp are thought to be a potential contributor to wetland turbidity further limiting macrophyte growth. Carp together with the lack of drying cycles in the floodplain wetlands are therefore believed to have contributed to the demise of wetland macrophytes (Blindow et al. 1993; Pressey; van der Wielen 2001; Walker et al. 1993).

Further macrophyte loss is due to a lack of dry periods in wetlands. The lack of drying cycles reduces sediment compaction leading to easier re-suspension and increased wetland turbidity (McComb et al. 1997). Through increased turbidity macrophytes can be shaded out causing their dieback. Their regeneration cycle, which is dependent on dry spells, is also interrupted through the permanent inundation. The lack of competition for underwater light due to the loss of macrophytes, as well as the loss of nutrient buffering actions of macrophytes, stimulates phytoplankton growth and increases the potential for future algal blooms (Carpenter et al. 1997a; Recknagel et al. nd). Although the introduction of drying cycles is partly expected to reduce wetland turbidity other causes for water turbidity also exists for the River Murray and have an impact on the potential reduction of turbidity possible in a wetland. Darling River water for example, which is known to be turbid and sodic soils, widespread in Australia (Rengasany et al. 1991), contribute to maintaining turbidity within wetlands.

In 2000 the Department of Water, Land and Biodiversity Conservation introduced a flood event to a section of the lower River Murray floodplain (DWLBC 2004; Siebentritt et al. 2004). A study on the impacts of flooding on riparian plants of the
River Murray showed an increase in flood dependent species, the reduction in flood intolerant species but no change in aquatic species (associated with an impoverished seed bank) (Siebentritt et al. 2004). The recommendation of this study was future repeat flooding to increase the aquatic species seed bank and enhance their regeneration. This flooding in 2000 also seemed to be effective in reducing exotic species numbers. This study confirms the hypothesis of the impact river regulation has had on riparian and aquatic species, with the reduction in aquatic species seed bank of the lower River Murray wetlands. The study also shows one method of influencing and improving species regeneration, i.e. flooding.

Nielsen and Chick (1997) conducted a study on sixteen artificial billabongs on the River Murray floodplain. Their findings were that the longer a billabong remained flooded the less diverse the plant communities became. The permanent flooding in their study did not allow ephemeral or terrestrial species to grow, whereas in billabongs where extended periods of drying followed by spring flooding was introduced more diverse plant growth including terrestrial taxa were seen as a consequence. This shows that should wetlands in the lower River Murray floodplain have a natural water regime a more diverse plant community should become evident. As a wetland management strategy the alteration of wetland inundation through the introduction of dry periods and consequent re-flooding should stimulate responses to species regeneration in turn returning a wetland to a stable clear state. In the lower River Murray this management response may however be reduced when the main water source is from the more turbid Darling River.

1.2.3 Irrigation drainage and constructed wetlands

Both constructed wetlands and natural wetlands can be used to improve water quality (Keenan et al. 2001). Braskerud (2002) found that constructed wetlands placed at first order streams removed between 21% and 44% of the phosphorus inflow. Constructed wetlands, in a study by Burgoon (2001), were found to remove from 50% to 99% of the nitrate inflow load. In a study of constructed wetlands in Flanders Belgium, the nutrient removal efficiencies ranged from 31% to 65% for nitrogen and 26% to 70% for phosphorus (Rousseau et al. 2004). Whereas Schulz et al. (2004) found constructed wetlands for the treatment of aquaculture runoff were able to remove 41% to 53% of phosphorus and 19% to 30% of nitrogen. Lüderitz et al. (2002), who
studied the effectiveness of constructed wetlands on sewage treatment, found removal rates of phosphorus to be between 27% and 97% depending on the constructed wetland design and management. Stormwater treatment, in Australia, with constructed wetlands were found to remove up to approximately 80% of phosphorus (Bavor et al. 2001). This shows that the reason for using constructed wetlands in the removal of the dissolved nutrients phosphorus and nitrogen can be diverse, the effectiveness of the constructed wetlands also varying significantly. The optimal design parameters of the constructed wetlands and the retention time required for nutrient removal depend on the nutrients being removed (Bavor et al. 2001; Hunter et al. 2001; Rousseau et al. 2004). However, all studies showed that wetlands can be used for nutrient removal.

One of the major contributors of nutrients to wetlands is irrigation drainage from agriculture. Constructed and natural wetlands are capable of absorbing phosphorus and can be used for phosphorus load reduction (Kadlec 1997). How they impact on a river system (i.e. capacity of wetlands to remove nutrients from the system) depends on their location in the landscape (Crumpton 2001). In identifying the best landscape position of restored wetlands Crumpton (2001) found that where wetlands are placed to intercept a higher load of nutrients there is an increased retention capacity.

Studies by Wen and Recknagel (2002) and Wen (2002a) at a wetland in the lower River Murray show that constructed wetlands can reduce wetland nutrient inflow from irrigation drainage by up to 90% (Wen 2002a; Wen et al. 2002). Therefore, in cases where the main wetland degradation impact comes from ‘reclaimed swamp’ or dairy pasture irrigation drainage outflow, the eutrophication source can be reduced substantially. Consequently, where possible the interception of irrigation drainage and treatment prior to its flow into wetlands could contribute considerably to the reduction of nutrient inflow loads into wetlands.
1.3 Restoration of degraded floodplain wetlands

1.3.1 Management strategies for restoration

Following wetland restoration, through the re-introduction of drying cycles and carp restriction during re-wetting of wetland degraded by permanent inundation, Recknagel et al. (nd) observed the recovery of wetland habitats and the improvement of water quality. By introducing drying periods or partial draw down, the germination and growth of macrophytes are stimulated allowing for a return of macrophytes in a reflooded wetland. Although initial conditions following re-wetting show increased nutrient availability and therefore algal growth in the wetlands, macrophytes once established out compete the algal community for nutrients (Recknagel et al. nd). Where possible, such as in constructed wetlands, the harvesting of macrophytes can partially remove the nutrients from the system (Hunter et al. 2001).

The main benefit of the drying of a wetland is the consolidation of the wetland sediments, which reduces re-suspension, minimising turbidity and release of nutrients from the sediment (McComb et al. 1997; Recknagel et al. 2000; van der Wielen 2001). Therefore, re-introducing a dry period to a wetland can have the impact of switching a wetland from a turbid stable state to a clear stable state (Scheffer 1998; Scheffer et al. 1993) as discussed above. Consequently, the reintroduction of dry phases has been recommended as a management strategy to improve or restore wetlands of the Murray-Darling Basing (Scholz et al. 2002).

Equipping wetland inlets with grills will prevent large carp from entering the re-flooded wetland. It is assumed that this will protect macrophytes from being uprooted by carp, as well as reducing the re-suspension of sediment expected as a consequence of their feeding behaviour.

As a summary of the above discussed issues of wetland degradation, the potential management strategies therefore available to improve water quality of degraded wetlands, which have been considered in this project, are as follows:

1. The reintroduction of drying and wetting cycle’s thereby reducing turbidity of wetlands. Through this measure, the function provided by emergent and submerged macrophytes can be reinstated revitalising a degraded wetland (Recknagel et al.; Recknagel et al. 1997; Recknagel et al. 2000; Scholz et al.}
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2002; van der Wielen 2001). Drying consolidates the sediment and therefore reduces the quantity of suspended solids in the water column. The re-emerging macrophytes act to improve water quality by nutrient uptake, reduce flow speed increasing sedimentation (Sand-Jensen 1998) and by out competing phytoplankton for nutrient (Recknagel et al. nd). Experiments have shown that water quality in wetlands managed in this manner can improve (Recknagel et al.; Recknagel et al. 1997; Recknagel et al. 2000; van der Wielen 2001). There are two possible mechanisms for introducing dry periods; these are through the construction of regulators at individual wetlands or to implement it at a broader scale through the change in water retention and therefore river height at individual locks.

2. Irrigation drainage nutrient reduction through constructed wetlands. In wetlands where external point nutrient sources such as irrigation runoff contribute to the wetland nutrient load, there is an opportunity for the construction of artificial wetlands where macrophyte harvesting can be used to reduce nutrient loads. An example of this management strategy, in the lower River Murray, was a research project situated between the Reedy Creek wetland and the Basby farm near Murray Bridge (Wen 2002a). Here an experimental pond system was constructed to eliminate inorganic phosphorous from agricultural drainage by native water plants (Wen 2002a). This research may lead to the design of constructed wetlands for the treatment of agricultural drainage water before it enters floodplain wetlands.
1.4 Predictive modelling of wetland processes and services; current state and potential alteration due to management

Real environmental systems are complex and it is therefore extremely difficult to measure the parameters with accuracy (Parsons et al. 1995). The predictive ability of water quality models is seriously limited by the difficulty in identifying complex environmental processes and defining these within parameters (McIntosh 2003; McIntyre et al. 2003; McIntyre et al.; Reckhow 1994). One method to overcome this is to invest in monitoring and field based research. However, this quantitative understanding and data are difficult and expensive to obtain. Ecologists and resource/land managers cannot always employ traditional quantitative simulation because of financial and temporal constraints (McIntosh et al. 2003), and therefore need to use alternative approaches such as modelling.

Clearly substantial and complex data are required in order to assess and understand the processes within a wetland, the interactions of these processes within the wetland, and processes having influences upon wetlands, let alone assessing the implication of potential management strategies. Assessing such a substantial and complex data set is therefore outside the capacity of an individual. To facilitate the understanding of processes operating on such a large scale computer models can be created to assist in evaluating the wetland processes. This enables an assessment of management scenarios as well as the testing of hypotheses of wetland function (Caswell 1988; Goodall 1972; McIntosh 2003; McIntosh et al. 2003; Oreskes et al. 1994; Rykiel 1996; Wallach et al. 1998). As a consequence of the complexity of assessing such a vast and complex data, there has been an increasing use of simulation models in the study of aquatic and other ecological systems over the past couple of decades (Elliott et al. 2000; Oreskes et al. 1994; Wallach et al. 1998).

There are two strategies for the management of degraded wetlands considered for this modelling work, the choice being dependent on the reason underlying the degradation. For wetlands where the main degradation is the inflow of nutrients constructed wetlands would be considered. These constructed wetlands would eliminate nutrients by absorption to nutrient poor sediments and nutrient uptake by macrophytes. Simulation of the management of these wetlands would help determine the impact of successful nutrient removal on the wetland and its exchange rate of
nutrients with the river. Where permanent inundation is the primary cause of wetland degradation, a model could help determine the impact of the introduction of drying and wetting cycle, on internal nutrient dynamics and wetland nutrient uptake. Both of the simulations would provide assistance in decision support by providing an estimate of:

- overall wetland recovery and restoration
- wetland specific responses to restoration management, and
- degree of response required from either nutrient removal (constructed wetlands) or from turbidity reduction (sediment compaction)

Modelling can be useful in understanding ecosystem processes and predicting intervention outcomes. There is an inherent value in the analysis of past and present in setting goals and objectives for the future (Thomann 1998). That is, modelling can be used as a tool in predicting the ecological consequences of restoration plans or management scenarios (Costanza et al. 1998; DeAngelis et al. 1998), which are vital factors in environmental decision-making. Recognition of model validity, and transparency to stakeholders, increases understanding and contributes to informed dialogue, thereby enhancing decision making by consensus (Thomann 1998). A model can also be helpful in a situation where non-linear mechanisms cause unexpected patterns that cannot be grasped intuitively (Scheffer et al. 2000), or where systems are too complex or cumbersome for human interpretation alone. For example, computer models can be used to predict wetland response to environmental change (Sklar et al. 1993).

A model is not expected to achieve exact predictions of ecosystem function, but its development provides a tool for an approximation of outcomes. After all, modelling often involves stressed systems with a view to return them to a natural state (Beck 1997). Not all potential impacts can be modelled successfully following intervention, as there is always some lack of knowledge. However, modelling can help minimise (but not eliminate) the variability of potential outcomes (Beck 1997).

Simulation models have become widespread and are playing an increasingly important role to assist in the decision making process (Griffin et al. 2001; McIntyre et al.; Sullivan 1997). With the fragmentation and lack of wetland specific data and knowledge in the lower River Murray region, it is difficult for managers and other
stakeholders to make decisions in the management of wetland restoration. This is particularly true in assessing the highest return of investment (cost-benefit analysis). Managers ideally desire models capable of quantitative predictions of restoration scenarios, taking into account hydrology and ecological processes of aquatic environments (Arthington et al. 2003). However such models are not yet available for Australian conditions as their development is prohibitively expensive, particularly models that take into account catchment or regional scale (Arthington et al. 2003). A simplified model capable of regional scale scenarios may however be able to answer some of the questions posed by managers.

1.4.1 Complexity and feasibility of modelling

There are two important factors which will dictate the complexity of any model of an ecological system. The first is the purpose of the model, dependent on the aims of the potential user, e.g. flexibility may be an important issue. The second factor is the feasibility of a model. This can be dependent on the understanding or knowledge available for a system. That is, to what extent can the system be explained within a modelling framework based on the current knowledge (McIntosh et al. 2003; Reckhow 1994; Young et al. 2000). Furthermore, the incorporation of too many factors into a model can obscure the action of some processes and render the model mathematically inflexible (Caswell 1988). Caswell (1988) even suggested omitting important factors to avoid obscuring the focus of the model. De Wit and Pebesma (2001) compare four models of increasing complexity to assess the value of complex models versus simple models. Their conclusions are that the complexity of models may not improve the modelled results if the data quality is restrictive.

A model does not have to be extremely complex as good data for model development may be all that is required to produce a simulation that will answer questions (Gibbs et al. 1994). Taking this further, the simplest possible model, which can accurately predict an observed phenomenon, provides a valuable contribution to ecological knowledge (Caswell 1988). It provides a starting point on which there is a possibility to build on observations and develop new theories (Caswell 1988). Whereas unnecessarily complex models may lack the flexibility that may be required and may contain inherent flaws (Wood 2001).
The choice between simple and complex models is affected by knowledge and data availability. Young et al. (2000) found aquatic ecology to be complex and dynamic with a multitude of interactions. However limited data, such as for the lower River Murray wetlands, and limited detailed knowledge and understanding, of aquatic ecology (Keenan et al. 2001; Young et al. 2000) argues for a simple model structure (de Wit et al. 2001; Li et al. 2002; Li et al. 2003; Reckhow 1994). Reckhow (1994) claims that limited data and knowledge are incompatible with high detail and large models. The lack of detailed understanding of each process required to develop a holistic quantitative model of an aquatic ecosystem restricted the modelling by Young et al. (2000) to a few parameters. Young et al. (2000) therefore adopted a simplistic modelling approach. The degree of knowledge is therefore an important determinant of the level of complexity allowable within the model to achieve a meaningful and accurate scenario (Wood 2001).

Wetlands are variable ecological systems and can be complex to model. This is due to their morphology, susceptibility to sporadic external influences such as wind, temperature, river flow (directional change is a possibility in the case of lower River Murray wetlands (Webster et al. 1997)) and a multitude of complex dynamics and interactions that cannot be monitored and studied without disturbing (and therefore influencing) the system. Modelling wetlands can therefore become a complex venture often hampered by the lack of detailed monitored data as well as rapid and sporadic change in condition such as water availability, weather etc.

Despite the lack of knowledge, many complex descriptions of wetland behaviour and nutrient cycling have been developed for modelling purposes. However, the complexity complicates and often defies calibration and validation. Through this complexity, the ecosystem can behave counter-intuitively despite individual components being well understood. Therefore, wetland models are often kept simple, with well-understood parameters and processes assigned a defined value (Kadlec et al. 2001). Additionally, simple models are easier for non-modellers such as water quality managers and the general public to understand (Murray 2001), which contributes to consensus building.

Young et al. (2000) began with a simple model, intending to extend model complexity in the future. The key to their model development was keeping the degree of complexity consistent with the current level of understanding. Therefore, the model
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can be developed as the understanding develops. This premise was also used in WETMOD development (Cetin and Recknagel, pers. Com). Therefore, WETMOD can be further developed with increasing understanding and data availability, and is therefore a basis from which to conduct further research for lower River Murray wetlands. Another example of simple wetland nutrient retention models, simple due to limited data and knowledge, are described by Li, Xiao et al. (2003) and Li et al. (2002) and discussed in relation WETMOD below. Different ways how the accuracy of models that simulate complex systems can be assessed are examined below.

1.4.2 Qualitative and quantitative assessment of model accuracy and generic applicability

As the output requirements for models can vary depending on their intended application and purpose; further differentiated by data availability by which to run scenarios, many opinions on the need for quantitative vs. qualitative modelling output have developed. Different methods of assessment of model performance have therefore been developed. Judgmental terms such as excellent, good, fair, and poor are useful because they can invite, rather than discourage, contextual definition (Oreskes et al.). It is not uncommon for water quality models to have a small amount of data available for model development, leaving even less for model evaluation and testing. In this situation, rigorous testing and assessment of model predictions is rare and has little meaning (Reckhow 1994). Water quality model calibration should compensate to some degree for errors arising from model limitations (spatial averaging, model structure errors and numerical dispersion) (McIntyre et al.).

Data restriction to modelling

Due to the limit in data availability “exemplar” data have been used to develop predictive models. The model output along with continued monitoring can then be used for adaptive management relating model outcomes with real occurrences (Young et al. 2000). It must however be understood by both the model developer and future users that the level of assumptions regarding the use of “exemplar” data will affect the modelling accuracy in a quantitative way (Beck 1997; Wood 2001). In using assumptions within models some otherwise unsolvable process given the current data availability or knowledge can be resolved.
McIntosh (2003) states that there is no reason why relationships between abiotic quantities such as soil and nitrogen, or between biotic and abiotic quantities such as vegetation biomass and soil or water, cannot be modelled imprecisely if such an approach is required by the level of available knowledge/data or the model purpose. The model output in such a case should however not be expected to be quantitatively accurate. However, despite a lack of quantitative accuracy, qualitative results can be used as a guide in future monitoring, research needs and further model development. The argument may be that qualitative models outputs have an intrinsic uncertainty due to the imprecision of the outcomes. In fact stakeholders and managers are often aware of model uncertainty, however they do not see this as detrimental to the value of models in decision support (Andersson 2004). That is, the role of the model may be such that the only output possible is a qualitative one due to data limitations and therefore inherent assumptions. However, such a qualitative output can be informative and therefore assist in decision-making, even if this decision is of the necessity of further research. A qualitative model therefore fulfils its function where inadequate data is available for quantitative predictions.

Assessing accuracy

Many studies have discussed imprecise and qualitative modelling techniques (Dambacher et al. 2003; Guerrin et al. 2001; McIntosh 2003; McIntosh et al. 2003; Parsons et al. 1995; Wood 2001). An understanding of, and rigorous comparison to monitored data, can be used to assess a models qualitative and quantitative accuracy. Wetland management decision support models are not necessarily dependent on optimal statistical accuracy, and may fulfil their role when assessed as qualitative models (McIntosh et al. 2003), despite their quantitative output and need therefore not be validated as rigorously as purpose built quantitative models. Some models such as WETMOD (Cetin 2001) are developed as quantitative models that provide qualitative outcomes.

The comparison of model output, to establish modelling accuracy, can be performed in a qualitative or a quantitative manner, the decision of the methodology being dependent on circumstances (Wood 2001). The qualitative approach assumes that if the composition and structure of the ecosystem is known, it can be encompassed in models qualitatively (Dambacher et al. 2003). Users of such a model must however
understand that qualitative predictions are only a relative benchmark of expected system behaviour (Dambacher et al. 2003). In fact, Andersson (2004) found that stakeholders and decision makers were more interested in relative changes over long periods than prediction of exact time-series. Andersson (2004) found that the use of a simple qualitative model to assess future environmental conditions depending on management strategies was effective in stimulating a three-way communication between model developers, stakeholders and decision makers, the stakeholder however were cautious about regional information. The modelled results were found to provide a common base for understanding the impact of management. Andersson (2004) also found that qualitative information based on a generic environment was an effective model output for stakeholders and that quantitative information could be seen as confusing.

A descriptive analysis of ecological model output compared to monitored data is a valid method of assessment (Wood 2001). The assumptions required in the development of ecological models also cause a mismatch between model output and monitored data, therefore reducing the potential of statistical accuracy (Wood 2001). Dambacher et al. (2003) suggest that an over-emphasis on precision in ecological research is neither necessary nor essential for mathematical rigor. They argue that an emphasis on statistics and precision may detract from a valuable qualitative understanding of the system. Another view is that the pursuit of optimal quantitative mathematical programs is not necessarily the primary concern of modellers. Beck (1997) argues that rather than asking what numerical optimisation can do for us, we should be asking how we can use our understanding of a system to successively improve numerical optimisation. That is to say, an obsessive pursuit of optimal numerical precision is not necessarily the role of a modeller. The successive development of models or model parameters should be seen as a step forward in the modelling process. The identification of good candidate models or equations may assist in directing research, leading to the discovery of better future potential models (Beck 1997).

**Generic applicability**

McIntosh et al. (2003) present the view that flexible and cost effective models are more beneficial than one-off models, which perform very well for one ecosystem
only. In an extension of this concept, the applicability of a model as a testing tool in wetlands where minimal or virtually no data exist, can expand the potential of management understanding and thus advance the decision making process as well as guide future research needs. Goodall (1972) and Rykiel (1996) make a distinction between testing the adequacy of a model’s predictions for a particular ecosystem, and generalisation of its applicability to a range of ecosystems. A model might be applicable and accurate for one particular ecosystem, however it may not be generic and therefore applicable to other ecosystems (Rykiel 1996). This is clearly important in determining whether predictions from a given model can be generally applied to decision-making and management strategies in other ecosystems. WETMOD (Cetin 2001) sacrifices some quantitative accuracy for a few wetlands, in favour of flexibility, applicability and cost. The point should be taken, as stated by Rykiel (1996), that a model may accurately simulate the qualitative behaviour of the system without the quantitative accuracy. In the development of WETMOD it was decided that the qualitative assessment was the more appropriate assessment approach, so as to maintain the generic applicability required in the region (Cetin 2001).

1.4.3 Validation

The necessity of validation is an issue that is the subject of considerable controversy in the literature. To increase the understanding of the approach used for a given project, and to potentially minimise conflict, a modeller should clearly state what the validation criteria are in the context of the model. Modellers should also state any restrictions and limitations of the application of the models. For the modeller to fulfil this obligation, the purpose of the model, the criteria the model must meet to be declared acceptable for use, and the context in which the model is intended to operate, must be specified (Rykiel 1996).

Rykiel (1996) discusses that models should be judged on usefulness rather than validity. However, model validation is required regardless of whether a model is expected to produce quantitative or qualitative outputs. Model validation is also important for end-user acceptance in the decision-making processes (Power 1993; Rykiel 1996). Mayer and Butler (1993) relate validation to the potential application and users of the model, where the validation is a comparison of model prediction to real world monitoring, to determine whether the model is suitable for its intended
purpose (Mayer et al. 1993; Rykiel 1996). Rykiel (1996) states that a valid model is one whose scientific or conceptual content is acceptable for its purpose. According to Goodall (1972), validation is testing to determine the degree of agreement between a model and the real system, that is, how good is the prediction, not whether it should be accepted or rejected (Goodall 1972; Rykiel 1996). Caswell (1988) argues against the case of validation being the decisive part of a successful model. His view is of the role of a model in expanding understanding and contributing to knowledge in a similar vein to experiments contributing to empirical problems.

Validation procedures range from general qualitative tests to highly restrictive quantitative tests (Rykiel 1996). Rykiel (1996), Oreskes et al. (1994) and Tsang (1991) all examine and discuss different validation concepts. Although a detailed discussion of this topic is beyond the scope of this thesis, it is necessary to state how the term “validation” is understood in the context of this project. WETMOD is assessed on the basis of Rykiel’s (1996) description of validation, where validation is a test to confirm that the model is acceptable for its intended use and meets its specified performance requirement (Rykiel 1996). For pragmatic purposes “a model only needs to be good enough to accomplish the goals of the task to which it is applied” (Rykiel 1996)

1.4.4 Modelling role in environmental decision-making

Scale

The study of ecological function and the management of natural resources have often been at a local scale, even though the ecological processes within wetlands, streams, and rivers occur at a larger (catchment) scale. One of the reasons for this local scale approach has been an inability to manage and analyse large and complex data sets. However, there has been a gradual recognition that management must be handled at large spatial scales to obtain meaningful results (Crumpton 2001; Fitz et al. 1996; Johnson et al. 1997). Fortunately, technology, spatial data, and software tools have advanced to such an extent that landscape-scale studies are now feasible (Johnson et al. 1997). As discussed in the previous chapter, to fully understand management implications and evaluate options the full impacts of restoration of wetland functions will need to be assessed on a landscape scale rather than at an ecosystem scale.
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Individual wetlands, through the food web, provision of habitat and flood mitigation, have an impact on surrounding wetlands, on the surrounding ecosystems and local land use (Bedford et al. 1988). Without consideration of wetland processes at watershed, landscape, and ecosystem scales, the most effective management strategies cannot be assessed (Lemly 1997). That is, the spatial modelling of ecosystems is necessary to develop a description of past behaviour, or to predict impacts caused by alternative management strategies (Mitasova et al. 1998; Sklar et al. 1993), and their impacts beyond their boundaries.

The benefit of landscape models is the ability to use them for the prediction of management impacts on wetlands, without actual alteration or potential destruction (Sklar et al. 1993). Spatial variation is important in assessing the response of a system to excessive nutrient loads and the impact on the system (Murray 2001). Specifically, landscape models can be used to study ecological principles, evaluate cumulative impacts, mitigate environmental alterations, and prevent large-scale anthropogenic mistakes from degrading wetland functions (Sklar et al. 1993). Models can also be used to predict “missing” data that can further be used in management decisions (such as flow exchange). Part of the strength of landscape models is the integration of disciplines due to their ability to handle large amounts of data and information, and provide output that is simple to convey (Boumans et al. 2001). Perhaps the major advantage of landscape models in catchment management is their comprehensive and systematic integration of knowledge and data for a specified region (Voinov et al. 1999a). Thereby a model user can be forced to view and interpret data normally not considered.

Cumulative impacts

As mentioned above, environmental impacts have often been assessed in the past on the local scale, and have not considered the broader scale impacts (Bedford et al. 1988). However, in a cumulative approach, the different external activities that impact upon a study area are considered. Therefore, on a landscape scale cumulative impacts from processes or activities external to the project area may become apparent that otherwise were not apparent using a local scale approach.

The cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time (Preston et al. 1988). When assessing
cumulative impact, the impacts caused by external activities and projects set the assessment boundaries i.e. the landscape scale (Bedford et al. 1988; Preston et al. 1988). Therefore, the area considered in cumulative assessment can expand from the wetland scale to catchment or regions. Only by allowing all external activities and processes that affect a wetland to determine the project boundary, can cumulative impacts be monitored or measured (Bedford et al. 1988).

The ultimate aim of a cumulative impact assessment is to evaluate the impacts that may result from change. These impacts include the physical, chemical, and biological changes to an environment (Abbruzzese et al. 1997). The cumulative impact of nutrient uptake due to management, whether improved or degraded, falls within the scope of impact assessment of potential landscape scale wetland management application. Accordingly, the cumulative impact observed due to the simulation of multiple wetland management scenarios can be viewed as a cumulative impact assessment of the proposed management strategies.

Models role in estimation of nutrient retention

Simulating nutrient flux within a river environment using models taking into account pollution sources through to river outlets should be able to assist managers to target intervention options for nutrient load reduction (de Wit 2001). There are models of various complexity which attempt to provide this capability such as PolFlow by de Wit (2001), which is based on physical laws and is embedded in a GIS (geographical information system). As well as a model by Crumpton (2001) who attempt to identify the position in the landscape of wetland restoration sites for optimal nutrient (Nitrogen) removal. Peijl et al. (1999) developed a model that was able to describe the carbon, nitrogen and phosphorus dynamics and interactions in riverine wetlands, and Muhammetoglu et al. (1997) developed a dynamic three dimensional water quality model for macrophyte dominated shallow lakes. An example of a simple spatial wetland model which simulates nutrient retention of wetlands is described by Li et al. (2002) and Li et al. (2003).

These models all try to simulate the nutrient retention capacity of wetlands and relate this back to the landscape scale, e.g. downstream nutrient load. The model by Crumpton (2001) attempts to direct management for an optimal return on investment,
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i.e. by addressing the question of where in the landscape investment in a wetland would deliver the greatest return as far as nutrient retention is concerned.

The PolFlow model was designed as a complex model. In following investigations de Wit and Pebesma (2001) found that under given circumstances, where the available data quality is the limiting factor in model development, simple models may in fact provide as accurate model results as complex models. Crumpton (2001), who study wetland restoration on a catchment scale, found through the application of their model that the location of restored wetlands was decisive in the ability of the wetland to effectively remove Nitrogen loads of the system. They showed that the interception of nutrients by the wetland should be a focus by managers in deciding on wetland restoration sites. Wetland managers lack the information to make any such decisions for lower River Murray wetlands. Landscape scale assessment through modelling can be used in a similar manner to Crumpton (2001), with due consideration given to the availability of data for the lower River Murray catchment.

Peijl et al. (2000b), who investigated the importance of landscape geochemical flows using a dynamic model, that simulated carbon, nitrogen and phosphorus cycling in riverine wetlands, show an example of a wetland model that did not manage to predict the field experiment. However this model did contribute to their understanding of the system (Peijl et al. 2000a). This shows that a model can be counted as successful simply based on the improvement of knowledge or understanding.

Muhammetoglu et al. (1997) developed a dynamic three-dimensional water quality model for macrophyte dominated shallow lakes. The model simulates the interactions between macrophytes and water quality parameters. The parameters, which are considered, include dissolved oxygen (DO), organic nitrogen, ammonia, nitrate, organic phosphorus, orthophosphate, biochemical oxygen demand, phytoplankton and the sediment. The model has been successful in prediction compared to measured values and can be used as a eutrophication management tool (Muhammetoglu et al. 1997).

The model developed and described by Li et al. (2002) and Li et al. (2003) is an example of a simple wetland model which simulates nutrient retention of wetlands and relates this to a landscape scale. Their stance is similar to that of research needs identified for the lower River Murray wetlands model; in that the data availability and
knowledge of the system being modelled was limited, the model was therefore impacted upon by a number of assumptions. As a consequence they opted for a simple model. Their model outcomes are in some instances contrary to those anticipated. But they point out that the trends displayed by the model are useful in guiding land use planning (Li et al. 2002). Due to the simple structure of their model Li et al. (2002) and Li et al. (2003) claim that it is applicable to other areas and therefore not location specific. The model output, from which management recommendations are made are only indicative of a trend (Li et al. 2003). This shows that in circumstances where limited data is available, model scenarios of wetland nutrient retention can be used for land use and other environmental management decision-making.

*Sediment compaction*

Sediment resuspension accounts for a large part of wetland turbidity influenced by climatic factors. To study the impact within one wetland a model could be made to account for wind direction and speed, and macrophytes role in sediment resuspension. An example of a project which accounts for these influences is a study performed by Hamilton and Mitchell (1996). The objective of the study performed by Hamilton and Mitchell (1996) in shallow New Zealand lakes was to derive relationships between suspended sediment concentrations and the physical forces caused by wave action, and to quantify the factors responsible for the differences between a number of lakes. They were successful in obtaining statistical evidence of the stabilisation mechanism of sediments and the inhibition of resuspension posed by macrophytes. One of the major causes degradation of the wetlands of the lower River Murray is their turbidity as discussed previously. Climatic data on a regional scale could be an option to be included in future monitoring studies of select wetlands thereby providing a representative case for the region. Wetland specific issues such as vegetation cover would also have to be included, complicating a landscape model. In the meantime a case exists for the development of a model capable of simulating the impact turbidity has on wetland ecosystem process and thereby assist managers in evaluating wetland rehabilitation needed to achieve a set turbidity reduction target.

*River Murray models*

The Murray-Darling Basin Commission (MDBC) has been using computer models for more than thirty years for water resources planning, development of operating rules,
development of salinity and drainage strategies and forecasting of flow and salinity (Close 1996). The history of mathematical modelling for the MDB to evaluate management options dates back to 1902 (Close 1986). From 1965 a computer water supply model was being used. Since then flow and salinity models have been created and their interactions improved (Close 1986). In 1996 a model called BigMod was taken up by the MDB, which replaced the older models and had the role of salt routing prediction in planning studies, short term flow and salinity forecasts, calculating solute loads based on historical data and modelling daily flow variations (Close 1996). Its role is to estimate electrical conductivity (EC) and track parcels of water throughout the system so that salinity load can be defined anywhere in a reach (Close 1996). The use of models by the MDB has been a successful venture. Due to the complexity of the Murray Darling Basin, and therefore the difficulty to qualify the impacts of changes to the system and the impossibility of quantification without modelling, the developed models have been extremely useful to the MDB to aid in management decisions (Close 1986).

The Flood Inundation Model (FIM) is based on historical flood inundation extent extracted from satellite imagery, known flow at the border, flood levels and lock levels (Overton 2000; Overton 2005). The FIM takes into consideration backwater curves. It provides managers of lower River Murray assets, such as wetlands and floodplains, with a tool to simulate potential inundation areas by changing lock levels at given flows across the SA border (Overton 2000; Overton 2005). The model output is for example used for assisting wetland management by simulating their inundation at given flow levels and relating this back to a potential hydrological regime. The FIM however identifies neither the flow paths connecting the wetlands and the river nor the turnover rate (water volume exchange) within wetlands.

A salinity model was developed for The Department of Water, Land and Biodiversity Conservation (DWLBC) to account for salinity impacts of wetlands on the lower River Murray, i.e. salinity accounting (Murdoch et al. 2004; RMCWMB 2002). The use of the model was intended provide a generic daily salt water balance as a consequence of wetland hydrology regimes. This model Salinity Impacts of Wetland Manipulation (SIWM) is a generic model relying on “exemplar” data and qualitative outcomes for generating quantitative assessments. The hydrology estimations within SIWM were taken from BigMod which propagates inaccuracies based on BigMod.
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assumptions. Consequently SIWM output quality is degraded (Murdoch et al. 2004). Although a novel approach the accuracy of the model output was not adequate for the purposes intended, i.e. salinity accounting for individual wetlands. It was therefore withdrawn from use by DWLBC. A replacement model is currently being developed by DWLBC (Croucher 2005). Further models for the lower River Murray are a number of groundwater models. These models simulate groundwater sources and impact on floodplain and river salinity and have been combined to make up a Floodplain Risk Methodology (FRM). The FRM is a collection of models used to assess the impact of groundwater on floodplain vegetation and the impacts periodic flooding and weir manipulation would have (Holland et al. 2005).

As discussed, to fully understand management implications and evaluate options the full impacts of restoration strategies on wetland functions will need to be assessed on a landscape scale rather than at an ecosystem scale. This having been identified as a research need for the wetland processes of the lower River Murray, a wetland ecosystem model called WETMOD 1, initially developed by Cetin (2001), was identified as a first step from which landscape scale could be built on. The aim of WETMOD 1 was to simulate macrophytes, phytoplankton, zooplankton and nutrients in the open water of wetlands. Cetin (2001) based the structure of WETMOD 1 on the Patuxent Landscape Model (PLM) (see Maxwell et al. 1997; Voinov et al. 1999a; Voinov et al. 1999b), and the lake ecosystem model SALMO (Simulation by means of an Analytical Lake Model) (see Benndorf et al. 1982; Recknagel et al. 1982).

The PLM is a complex landscape scale model of aquatic ecosystems including wetlands where ecosystem processes are simulated. This model allows simulations of a catchment using detailed morphological data (digital elevation models DEM’s) of the catchment and wetlands as well as time series of point source nutrient inflow. The model simulates an entire catchment using raster GIS data as the main driving variables where the model is run for each cell at each time step, propagating the results to the next cell for the next time step (Boumans 2001; Voinov et al. 1999b). The complexity of the PLM particularly the requirement of a detailed DEM prevents it from being applied in the lower River Murray system. An added complication for the lower River Murray system for a linear model is the non linear nature of flows in the lower River Murray. This can be through the reversing of flows “upstream” by wind and the bypassing of the main channel through rapidly flowing anabranches.
such as those of the Chowilla floodplain. Therefore, cell to cell modelling such as the PLM methodology would not be easy to adapt or implement. Part of the PLM could however be adapted to the lower River Murray system. Particularly when adapted with equations such as from time series dependent models such as SALMO.

SALMO was designed for the management of lake ecosystems, based on state variables phytoplankton, zooplankton and orthophosphate time series data. The SALMO model allowed for management simulations of nutrient cycles within lakes and the consequences of different management strategies for the control of eutrophication in lake and reservoir ecosystems (Benndorf et al. 1982). Using select equations from both as well as further literature Cetin (2001) was able to develop a generic model (WETMOD 1) for simulation of internal nutrient dynamics. The WETMOD 2 model described in the remaining chapters, built on WETMOD 1, is a contribution to the simulation of the lower River Murray system to aid informed decision making research and management of the lower River Murray wetlands.
2 Aims and objectives

The main aim of this work is to develop a model, which facilitates the analysis of management options for informed selection of wetlands requiring restoration, with the aim of re-establishing wetland landscape function through optimal means. To fulfil this aim the following objectives must be met:

I. Adapt a generic wetland process model for the lower River Murray floodplain wetlands and improve the resolution of the spatial influences acting upon a wetland

II. Evaluate these spatially relevant impacts on wetland nutrient uptake

III. Appraise the potential river nutrient-load buffer capacity of wetlands both pre- and post-management, on a landscape scale.

It is generally expected that restoring wetlands will reinstate their river-nutrient buffering capacity, consequently improving water quality in rivers by reducing nutrients otherwise available to support algal growth. The model is expected to deliver an estimate of the potential nutrient reduction in the lower River Murray following management intervention.

This project focuses on the lower River Murray wetlands and relies on previous work done in that area. Some of the research in the Lower Murray area has focused data collection and survey work, and has been summarised in the Wetlands Atlas of the South Australian Murray Valley by Jensen et al. (1996). Other projects in the Murray Darling Basin have been compiled and catalogued by the Murray Darling Basin Commission (Kirk 1998). However, the work this project mostly depends on are projects in the lower River Murray that have had objectives of producing solutions for particular problems. These past projects include for example the creation of weirs at individual wetlands for the introduction of drying cycles, and the construction of wetlands for nutrient removal from agricultural drainage water (prior to being released into the system). Recent baseline surveys (SKM 2004; SKM 2006) have added to the information available on the condition of individual wetlands for the purpose of wetland management. This data provides a simplified snapshot of the current condition of a few wetlands. However, a key lack of data, which impacts on a
number of research projects (such as fish habitat) and management decisions, is the exchange of water (turnover rate) between wetlands and the river.

Hypotheses:

I. A simplified generic wetland model can be used to realistically simulate multiple and different wetlands qualitatively. It is the premise of this project that a simple model will, with the available data, produce results that are sufficiently accurate so as to aid in decision-making (see 1.4.1).

II. A simplified generic wetland model can be used to answer “what if” questions for landscape scale scenarios, and

III. A simplified generic wetland model can be used to assess the cumulative impact of managing multiple wetlands.

This project adopts a generic wetland process model WETMOD 1 to account for wetland local external influences. These influences include improvement of the resolution of spatial influences such as river nutrient content, river flow volume and where appropriate external irrigation drainage inflow, which act upon a wetland. The model will evaluate the impact these influences have on wetland uptake of PO$_4$-P, NO$_3$-N, and production of phytoplankton, as well as how uptake can change at different locations. To be able to apply the model at different locations, despite restricted data availability, a wetland classification system incorporating the use of monitored data from intensely studied wetlands as regional-scale exemplars will be adopted. Therefore, the model will be applicable on a regional (landscape) scale providing qualitative understanding of the cumulative impact of wetland management.
3 Materials and Methods

3.1 Model Description

For application of the model in decision-making, managers and stakeholders need to understand the models purpose as understood by the developer, any assumptions made during its development and the model structure (Bart 1995). The following chapter describes the assumptions of wetland behaviour as well as the model structure. The model used for the basis of extended development is WETMOD 1 developed by Cetin (2001), which was based mainly on literature data.

3.1.1 Design Considerations

Problems current for wetland management is the acute lack of awareness of impact of management on a regional scale. Given that wetlands will have a varying nutrient retention capacity depending on the turnover rate, i.e. longer turnover rate will allow for more nutrients to be absorbed, finding an optimum turnover rate to maximise the nutrient retention capacity of wetlands could be a management aim for river nutrient reduction. To assess the impact of wetland management for nutrient reduction in a river it is necessary to assess the capacity of a wetland to retain nutrients individually and cumulatively at the landscape scale. Therefore, multiple variables come into play to assess the capacity of wetlands to retain nutrients on a landscape scale.

The first step of assessing individual wetland nutrient retention was addressed in part by WETMOD 1 (Cetin 2001). The limiting factors are, as is often the case, the acute lack of sufficient data when the model is to be applied to a landscape scale. The wetland model WETMOD 1 has the ability to simulate the general internal dynamics of a wetland with minimal monitored driving variables, therefore allowing the model to be applicable at sites with minimal data. With site-specific data on water exchange, nutrient through flow and wetland morphology, introduced during the development of WETMOD 2, the modelling of wetland dynamics becomes more specific for an individual wetland although using landscape scale available data. None the less, with the limited resources invested in the monitoring of wetlands, only very few can reliably be simulated. To overcome the restriction hindering the testing of
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management strategies for wetlands and assessing the potential cumulative impact.

two options remained:

1. The substantial investment of resources in monitoring of wetlands, prior to the extensive development of a model. Such a model would potentially be capable of the simulation of each of the monitored wetlands in detail, thereby providing managers with a robust and comprehensive decision making tool. This option has the drawback of the investment of substantial resources, loss of valuable time in monitoring and model development. The largest drawback being that the model would still be restricted to the monitored wetlands.

2. The development of a modelling tool capable of assisting in the development of understanding of potential management decisions, which would be based on current available data and knowledge. The restriction on the complexity of such a model would be ensuring its applicability to all wetlands within the catchment area based on the current data availability. Therefore, as such a model would need to rely on available data some of the accuracy of model results would be dependent on the range of data quality and quantity.

Going with the second option, a developed generic model which allows the assessment on a landscape scale of wetland function and cumulative impact, the simplification of wetland into wetland classes becomes necessary, to such an extent that no wetland is seen as unique nor all wetlands as equal (Bedford et al. 1988). If this simplification is not introduced, the data required for a landscape scale assessment becomes insurmountable.

There are multitudes of ways to classify wetlands. The system that is chosen is dependent on the purpose of the classification, the time available, the data and the knowledge available, as well as the preconception of the classifier, which will affect any wetland classification. In a general sense, there are 2 approaches, one through geomorphology and the other through the hydrological relationship of the wetland to the river (Bedford et al. 1988; Pressey 1990).

As an example of a classification procedure Strager et al. (2000) used a landscape based approach to classify wetlands and riparian areas based on habitat requirements of amphibians and reptiles. This classification also included forested and non-forested groupings (Strager et al. 2000).
The classification used in this project is partially driven by the limited data availability for both geomorphology and hydrological relationship between the wetland and river. The approach was therefore a very simplified hydrological connectivity classification, which will be discussed in more detail below.

The description of the model is broken down into two segments, WETMOD 1 and WETMOD 2. The first description, WETMOD 1, relates to the model sections developed by Cetin (2001) that relate to internal nutrient dynamics. The second section, WETMOD 2, relates to the redesign of the model to account for external influences acting upon a wetland. The methodology for the application of WETMOD 2 to assess cumulative impact of wetland management is discussed at the end of this chapter.

The macrophyte biomass module is described in the macrophyte sector below. The phytoplankton and zooplankton biomass module are described together as part of the plankton sector, and the $\text{PO}_4$-P and $\text{NO}_3$-N module is described as part of the nutrients sector. The “Fitted River exchange and Irrigation Drainage Inflow” was, due to its complexity, split into separate modules within WETMOD 2, which are described as Flow Exchange Sector and External Nutrient Source Sector. Both of these relate to the significant addition to the model where internal nutrient dynamics are related to external and therefore landscape scale impacts such as river nutrient load. The output of both of these sectors contributes significantly to management considerations on a landscape scale. The sources of differential equations are described in Appendix A. The descriptions of the macrophyte, plankton and nutrient sectors have been adapted from Cetin (2001).

Units of input data (conversions are performed within the model, descriptions of which can be found in section 3.3);

- MDBC river data;
  - Nitrate + Nitrite as N: mg/L
  - Filterable Reactive Phosphorus as P: mg/L
- Reedy Creek river data;
  - Nitrate as $\text{NO}_3$-N: mg/L
  - Soluble Reactive Phosphorous as $\text{PO}_4$-P: mg/L
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Miscellaneous;

- Turbidity: NTU
- Temperature: °C
- Secchi depth: metres
- Chlorophyll-a: μg/L
- Solar Radiation: MJm²
- Wetland Volume: cubic metres (m³)

Units of output data;

- Nutrients (PO₄-P and NO₃-N): g/L
- Phytoplankton biomass: cm³/m³
- Zooplankton biomass: cm³/m³
- Macrophyte biomass: kg/m³

Principal Model Assumptions (for simplification of model design)

The following assumptions were made at the commencement of the project to compensate for a general lack of data, and data quality for the lower River Murray. These were needed as part of the simple model design strategy employed.

1. As the considered wetlands are permanently inundated wetlands, it is assumed that as a result of lock management, where locks are maintained at a constant level, all wetlands included in potential management scenarios have a constant volume as well as a permanent connection with the lower River Murray. Consequently, there is a bi-directional and permanent exchange of water and nutrient with the river, the exchange volumes (in- and out-flow) being equal.

2. There were no data on exchange flow or channel morphology; therefore it was assumed that the exchange volume was solely dependent on the river flow volume.

3. It was assumed the wetland is homogeneously mixed for each modelling time step. Simulated wetland nutrient data would therefore represent the concentration throughout the wetland.
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4. South Australia is a dry state, and there generally are no significant catchment areas for individual lower River Murray wetlands. It was therefore assumed, that there would be only low or insignificant nutrient inflow though precipitation runoff for most wetlands. The exception is Reedy Creek wetland, and therefore by extrapolation, all category 4 wetlands (wetland classification is described below).

5. For management simulation purposes, it was assumed that the introduction of dry periods to wetlands would compact the sediment and reduce the turbidity within the wetland during the next wetting event. The inherent assumption is that the turbidity is caused by suspended sediment and is not significantly contributed to by phytoplankton. However, phytoplankton will increase the turbidity in proportion to its own growth. A future user of the model must therefore take this into account when assessing model output. The management scenarios also assume that the turbidity is independent of the potential inflow of suspended sediment from the river as the river turbidity fluctuates dependent on the water source of upper River Murray versus the Darling River. A modeller must therefore take into account the realistic potential reduction in turbidity for a given wetland dependent on external sources as well as its internal dynamics, i.e. resuspension and sedimentation.

6. For management simulation purposes it was assumed that all same category wetlands resemble each other in exchange volume. In an operational application local knowledge of the exchange volume for simulated wetlands would assist in improving potential modelling output.

3.1.2 WETMOD 1

The WETMOD 1 model (Cetin 2001; Cetin et al. 2001) is a generic wetland ecosystem model. WETMOD 1 simulates internal wetland nutrient dynamics, i.e. the growth of macrophytes, phytoplankton and zooplankton through mass balance equations (Figure 4). WETMOD 1 simulates internal wetland nutrient processes using water temperature, turbidity, Secchi depth and solar radiation as driving variables (model time-series input). Phosphorus as PO$_4$-P, nitrogen as NO$_3$-N, macrophytes, phytoplankton and zooplankton are state variables (model output). This section, represented as WETMOD 1, was rigorously adapted into the WETMOD 2
environment to account for advancement in data and addressing model limitations. The description in this thesis is therefore of the model sections, WETMOD 1, as they are found in the WETMOD 2 model. Any generic wetland ecosystem model such as WETMOD 1 could be adapted in a similar fashion to account for the river, wetland interaction and cumulative impact on a landscape scale.

The Model WETMOD 1 was developed and implemented by means of the modelling developmental software STELLA (2000). STELLA provides an intuitive user interface for domain experts with little modelling experience. Models developed within STELLA are, due to its rigid structure, transparent.

![Diagram of WETMOD 1 model interactions](image)

**Figure 4: Driving Variables, State Variables and Major Interactions in WETMOD 1**

Nutrient contribution to the wetland occurs through sediment release, surface runoff, irrigation drainage and river inflow. The data used being either sourced from literature or approximate values obtained from expert recommendation. Nutrient loss occurs through uptake by macrophytes and phytoplankton as well as sedimentation and wetland outflow, this data being mainly calibrated or based on expert recommendation. Zooplankton increase is through growth, and reduction is through mortality. Macrophyte and phytoplankton increase is through growth (phytoplankton inflow being introduced in WETMOD 2). Macrophyte and phytoplankton biomass loss is through respiration and mortality, phytoplankton additionally through sedimentation, zooplankton grazing and outflow.
The model is divided into individual modules where related process equations are grouped together. The descriptions below are of the individual modules as they appear in WETMOD 2.

*Macrophyte Sector*

The macrophytes are simulated within WETMOD 1, and represented by their photosynthetic biomass in the open water (Cetin 2001). To obtain a simple model structure, all submerged macrophyte species found within the wetland were aggregated and represented as macrophyte biomass. Emergent macrophytes play an important role in the lower River Murray ecosystem, not only through their nutrient retention but also as habitat and sediment traps. WETMOD 1 however did not consider emergent macrophytes in the model and consequently neither will WETMOD 2.

The increase in macrophyte biomass within the model is controlled by the productivity of the photosynthetic biomass (‘Mac Gross PP’ Figure 5). The loss of macrophyte biomass is through mortality and respiration of photosynthetic biomass (‘Mac mortality’ and ‘Mac respiration’).

The growth of macrophyte (photosynthetic) biomass is influenced by the growth rate, turbidity and nutrients, underwater light and water temperature. In Australian waters turbidity can be a controlling factor in macrophyte growth (Roberts *et al.* 1986), with the growth rate reaching a maximum when the turbidity is below 70 NTU (Shiel *et al.* 1982). Therefore, the reduction in turbidity is seen as a major aim of wetland management and is consequently the main focus of management scenarios of the model.

Productivity of the photosynthetic biomass (‘Mac Gross PP’), i.e. macrophyte biomass growth, is contributed to by the macrophyte production coefficient (‘mac prod cf’), Gross Primary Production rate for the total plant biomass (‘Mac GPP’) and can be limited by turbidity if it surpasses the 70 NTU threshold. The production coefficient (‘mac prod cf’) is calculated from the availability of nutrients, underwater light and water temperature (see Appendix A for equations).

The underwater light coefficient calculation is based on the Beer-Lambert Law for light attenuation, where the data required is Secchi depth and solar radiation. Solar radiation input is MJm$^{-2}$/day. The equation used in WETMOD 1, which was obtained
from literature, demands units in Jcm$^2$/day, therefore, in WETMOD 2 MJm$^2$/day is multiplied by 100 to convert to Jcm$^2$/day. Where Secchi depth data are missing or of very poor quality, the Secchi depth is calculated based on the turbidity within the wetland. The equation for calculating the Secchi depth from the wetland turbidity is discussed in detail in section 3.2.1. Therefore, the Secchi depth data source can be either monitored, calculated from the turbidity or fixed manually.

The water temperature is one of the driving variables of WETMOD (1 and 2). The macrophyte temperature coefficient (‘mac temp cf’) is based on the optimum water temperature for macrophyte growth (Boumans 2001). The macrophyte nutrient coefficient (‘mac nut cf’) is based on the Michaelis-Menten expression, where the nutrient uptake is dependent on the concentration of the nutrient in the water and the nutrient half saturation constant (‘mac Ks N’ and ‘mac Ks P’, see Appendix A) for uptake.
The plankton sector, seen in Figure 6, (labelled as phytoplankton module) comprises both the phytoplankton and zooplankton equations within the model. Both phytoplankton and zooplankton have, for the sake of model simplicity, been aggregated into their respective state variable, i.e. phytoplankton biomass and zooplankton biomass.

The phytoplankton biomass can have two sources of input. One is the wetland growth of phytoplankton expressed as the phytoplankton gross primary productivity (‘pht...
Gross PP’), which is dependent on the phytoplankton production coefficient (‘pht prod cf’), and limited by the maximum biomass of phytoplankton (‘pht max’), i.e. the carrying capacity which is calibrated for each wetland. The phytoplankton production coefficient (‘pht prod cf’) is obtained in a similar manner to the macrophyte sector. The second input source of phytoplankton is from external sources such as the river or irrigation drainage inflow (‘Phytoplankton In’), the load is fitted with the exchange estimate; see External Nutrient Source sector description below.

The phytoplankton biomass reduction is through five sources, mortality (‘pht mortality’), respiration (‘pht respiration’), sedimentation (‘pht sedimentation’) outflow (‘Phytoplankton Out’) and zooplankton uptake (‘Pht grazing’). The phytoplankton respiration is dependent on water temperature and the temperature limitation coefficient (‘pht temp cf’). The phytoplankton temperature coefficient equation is adapted from Hamilton and Schadlow (1997), which relates the growth to a constant multiplier of the water temperature.

Within wetlands there is a net increase in effective sedimentation as a consequence of increasing turbidity. That is, due to the increase in suspended particles available there is a net increase in sedimentation as when compared with a clear wetland. The sedimentation is controlled through a calibrated sedimentation rate, which is altered by the turbidity of the wetland. In WETMOD the change in sedimentation rate is controlled through a calibrated turbidity threshold at 95 NTU. Mortality of phytoplankton is controlled through a set mortality rate and the respiration through a respiration rate. The outflow is dependent on the fitted (estimated) exchange rate of the wetland, described in Flow Exchange and External Nutrient Source Sectors. The zooplankton uptake is dependent on the zooplankton growth rate controlled through the grazing rate of phytoplankton by zooplankton (‘Pht grazing’).

The zooplankton equations are adapted exclusively from SALMO (Simulation by means of an Analytical Lake Model) (Recknagel et al. 1982). The zooplankton outflow is simplified with the zooplankton mortality, controlled through a calibrated mortality rate for each wetland, accounting for all sources of zooplankton biomass reduction. The phytoplankton biomass growth is controlled through the phytoplankton grazing equation (‘Pht grazing’), which is the grazing of phytoplankton by zooplankton. Affecting the grazing, and therefore zooplankton growth, is the zooplankton respiration rate and the zooplankton growth rate. The zooplankton
growth rate is a function of both the macrophyte biomass and the phytoplankton grazing rate. The phytoplankton grazing rate is a function of the day length and the water temperature. The macrophyte biomass has an influence on the zooplankton growth rate due the assumption that it provides a shelter for zooplankton (Asaeda et al. 1997). Therefore, if the macrophyte biomass is low, the zooplankton biomass will reduce. The zooplankton respiration rate is controlled by the phytoplankton grazing rate and the water temperature.

**Figure 6: Plankton Module**

**Nutrients Sector**

Both of the nutrient equations consist of similar inflows and outflows, Figure 7. As discussed in section 1.1.2 and 1.2 the main contributors of nutrient inflow to wetlands
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are external sources such as the river or irrigation drainage inflow. As with phytoplankton, the inflow rate is determined by the fitted rate for the particular wetland, which is described in Flow Exchange and External Nutrient Source Sectors. Other inflows include ‘P loading’ and ‘N loading’ respectively, as well as ‘P sediment’ and ‘N sediment’ release. Nitrate flux is also potentially affected by nitrification and denitrification. However, due to insufficiencies in data, the sediment dynamics could not be modelled within WETMOD. The nutrient dynamics of the wetland are for the open water only, with sedimentation rates calibrated to adjust for missing complexity. This has simplified the model, but future research may need to invest in expanding this section of the model despite increasing complexity, as the present simplification does account for some model limitation.

The outflows include ‘P soil coprecipitation’, or the sedimentation of PO₄-P and NO₃-N ‘N soil coprecipitation’, P or N uptake and nutrient outflow as per the fitted exchange estimate, described in Flow Exchange and External Nutrient Source Sectors. The sedimentation rate accounts for the coprecipitation of nutrients, (which is the sorption of nutrients to suspended soil particles that then precipitate to the wetland floor). The coprecipitation is more pronounced at high turbidity due to the high availability of suspended soil particles, and can account for significant nutrient uptake by wetlands. The model assumes a calibrated sedimentation rate (calibrated for wetland categories) for both PO₄-P and NO₃-N of 50% at turbidity levels 70 NTU or above (the 70 NTU being a calibrated estimate that acts as a threshold), and 10% below 70 NTU (for Lock 6 wetland) or 50% vs. 20% (for Reedy Creek wetland), wetland classification is discussed below.

The uptake of nutrients by macrophytes and phytoplankton was adapted from the PLM (Patuxent Landscape model) (Boumans 2001). The uptake is dependent on nutrient to carbon ratio and the net primary productivity of both macrophyte and phytoplankton biomass.
Figure 7: Nutrient Module
3.1.3 WETMOD 2

External influences were not well constructed, or represented, in WETMOD 1, therefore to study the impact of respective external influences for differing wetlands these need to be added to the model. WETMOD 2 is a modification of the original WETMOD 1 model to suit the requirements of this project. As accounting for external influences is essential in regional scale scenarios, a modified version of WETMOD had to be developed, i.e. WETMOD 2. The capacity of the model to simulate potential impacts of restoration scenarios for the different wetlands would, through improved local relevant data, also be enhanced. The first two aims in the modifications of WETMOD 2 playing a part in fulfilling the third. These aims are listed below.

I. Overcome shortcomings in knowledge due to limited data and incomplete system understanding.

II. Address processes requiring further development, which were identified at the beginning of the study. These included river and wetland water exchange, nutrient exchange, and irrigation drainage data influence, and

III. Adapt and test the application of the model on a regional scale; i.e. develop a cumulative assessment of potential management impacts of multiple wetlands on the river nutrient load.

One challenge in modifying WETMOD to simulate landscape scale scenarios was to account for external influences acting on wetland water quality. This involved the further development of estimates of the inflow and outflow of nutrient to the wetlands. To accomplish this, the following sources were included in the model:

I. Irrigation (Reedy Creek wetland and Sunnyside wetland)

II. River Exchange modelling (Lower Murray River flow dependent)

During these key WETMOD modifications the following two principles were maintained:

I. Model transferability (Model Generic Nature)

II. Model Expectations (Reliable prediction of system dynamics, i.e. trends)

The model overview, WETMOD 2, and the data flow between modules are presented in Figure 8. The sections initially sourced from WETMOD 1 are represented as the
three green modules where internal wetland processes are simulated. The newer modules, which encompass the major modifications of WETMOD 2, include wetland data updates (yellow, rigorous reconstruction and update of the data base), new wetland specific morphology data (white) and wetland external nutrient sources (blue).
Figure 8: WETMOD 2 Structure and Data Flow
Flow Exchange Sector

As discussed in section 1.1.2 the transport of material in and out of wetlands is primarily a function of water flow (Johnston 1991). Therefore, the transport of material through water flow has the potential of being the most significant external influence acting upon a wetland. The process of fitting the exchange volume to a particular wetland based on Equation 6 which is discussed in detail in section 3.3. Essentially Equation 6 relates the exchange volume to the wetland nutrient concentration, river nutrient concentration and the river flow rate. This sector shows the adjustment of exchange volume estimation (Percentage of River Flow regarded as exchange) based on the river flow ML per day (converted to appropriate units as required within the model) (Figure 9). As the wetland is assumed to maintain a constant volume, any irrigation drainage inflow into the wetland is included in the respective wetland outflow volume.
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Figure 9: Volume Exchange Module
External Nutrient Source Sector

This sector encompasses the Nutrient Exchange module, as seen in Figure 10, where both the river exchange and the irrigation drainage inflow are introduced. That is, the inflow load of the nutrients and phytoplankton can be from two sources. The first is the river and the second, when applicable, the irrigation drainage inflow. The calculation of the individual loads is discussed in section 3.4 and calculated as per Equation 9 (irrigation drainage load to the wetland) and Equation 10 (nutrient load to the wetland from the river). Both are described in section 3.4.1. The sum of both loads is fed into the relevant modules described in the Plankton Sector and Nutrients Sector.

Within the nutrient exchange sector the irrigation drainage concentration and volume are selected and adjusted based on the wetland being simulated. Time-series for both irrigation affected wetlands Sunnyside and Reedy Creek wetland (described in section 3.2.1) are selected if either is being simulated; the option of testing for irrigation drainage affecting Paiwalla wetland was included in the model as it was also potentially impacted by irrigation drainage. The irrigation flow volume is manually set for Sunnyside as accurate volume data were not available, see section 3.2.1. Reedy Creek wetland irrigation flow was fixed at a set volume. The calculated irrigation drainage load (‘PDrainLoad’, ‘NDrainLoad’ and ‘Chla DrainLoad (Reedy or Sunnyside)’, see Figure 10) is distributed for each of the wetlands (‘P Drain Water Inflow’, ‘N Drain Water Inflow’ or ‘(Reedy or Sunnyside) Chla divided into wetland’) as per the seasonal flow pattern (‘Seasonal Flow Pattern SunnyORReedy’) described in section 3.2.1. The methodology of conversion of Chlorophyll-a to phytoplankton biomass is discussed in section 3.3, and performed within the model in ‘Phytoplankton Inflow cm3m3’.

The outflow module, Figure 11, is where the outflow of PO₄-P, NO₃-N and phytoplankton are calculated based on the fitted exchange volumes from the Flow Exchange Sector, expressed in terms of Equation 11 or Equation 12 (both equations for estimating the nutrient load from a wetland to the river, Equation 12 taking into account irrigation drainage, see section 3.4.1). These outflow concentrations are then fed back to the relevant modules described in Plankton and Nutrient Sectors.
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Figure 10: External Nutrient Module
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Figure 11: Outflow Module

Miscellaneous Sectors

Other modules (Sectors) within the model contain data handling such as data source selection (including driving variables based on wetland category), wetland volume calculation, and appropriate solar radiation and river data selection. In certain circumstances data conversions between units are handled within these modules e.g. river data conversion.

Model accuracy is tested using MS Excel based on the evaluation criterion $D$ described in section 3.3.2 (statistical estimation of the accuracy of model output in comparison to monitored data). Excel was also used to calculate the retention capacity of the wetland, the potential impact that this retention capacity has on the river nutrient load, the potential impact of management scenarios (Equation 13, Equation 14, Equation 16 and Equation 17), and the cumulative impact of multiple wetland management. Equation 13 and Equation 14 both relate to the change in nutrient
outflow from a wetland to the river. They are both described in section 3.4.1. Equation 16 and Equation 17 calculate the change in river nutrient load following wetland management and its percentage change in river nutrient load respectively. Both Equation 16 and Equation 17 are used to calculate the impact the management of a wetland or multiple wetlands has on the river nutrient load (they are both described in section 3.4.2).
3.2 Data: Model Driving Variables

WETMOD 1 was developed to study the impacts of internal wetland nutrient dynamics. At the time of its development, most of the external nutrient inflow data had not yet been collated, nor was all the wetland data available. Literature and other “exemplar” data were used to supplement the datasets used as the driving variables of the model. A working model was therefore developed which could be improved through the introduction of appropriate real data.

As this (WETMOD 2) project was to focus on regional scale applications of wetland models and to develop scenarios to study the potential impact of management, WETMOD 1 was deemed to be an acceptable basis from which to continue development. By using WETMOD 1, the time otherwise invested in redeveloping an internal wetland nutrient dynamic model was saved. However, rigorous data pre-processing was required to adapt WETMOD 1 to both the regional data set requirements and the updated data set. This describes the data set used and it’s pre-processing for WETMOD 2.

This project used several different monitoring databases as summarised in Table 1. Processes and conditions within wetlands have been monitored in several studies focusing on select wetlands. These studies occurred between 1997 and 2001 for periods ranging from 9 months to 2 years. Data were collected approximately every two weeks (Bartsch 1997; Marsh 1997; van der Wielen nd; Wen 2002a; Wen 2002b). These case studies are the source for all type-specific wetland properties as well as most abiotic and biotic time-series that are used as the main driving variables in the model, i.e. “exemplar” data. The monitored wetland locations, used in the modelling, were from open water sampling in the centre of the wetland. Error Bars in the data graphs displayed below were calculated from all monitoring sites within a particular wetland.

The exchange of nutrients between the river and wetlands depends on the river flow and river nutrient load. River flow data and water quality data (nutrient load) are collected at all locks and were obtained from the Murray Darling Basin Commission (MDBC) and the South Australian Department of Environment and Heritage (DEH). The flow data included in the model were collected at Locks 1 through to 8 (Figure 12), therefore for the model the most appropriate river data can be chosen for a given
Regional Scale Modelling of the lower River Murray wetlands

scenario. The main climatic driving variable is solar radiation data obtained from BOM (Bureau of Meteorology).

To apply the model to all wetlands along the river, location specific data have been incorporated. These include wetland size, depth, influence of irrigation drainage and connection to the river; and were obtained from Planning SA and Wetland Care Australia. From this morphological data the wetland could be assigned to categories, depending on hydrology and irrigation drainage influence.

Table 1: Data Sources, Type & Monitoring Frequency

<table>
<thead>
<tr>
<th>Data Type</th>
<th>Monitoring Frequency</th>
<th>Data Included</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetland, Drainage Inflow &amp; River (water quality)</td>
<td>Fortnightly</td>
<td>NO₃, PO₄, Turbidity, Temperature, Chl-a &amp; Secchi depth</td>
<td>University of Adelaide</td>
</tr>
<tr>
<td>River Monitoring</td>
<td></td>
<td>Temperature &amp; Turbidity</td>
<td>DEH</td>
</tr>
<tr>
<td></td>
<td>Weekly</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Fortnightly</td>
<td>Chl-a</td>
<td>DEH</td>
</tr>
<tr>
<td></td>
<td>Monthly</td>
<td>PO₄ &amp; NO₃</td>
<td>DEH</td>
</tr>
<tr>
<td>River Flow Volume</td>
<td>Daily</td>
<td>Water Flow</td>
<td>MDBC</td>
</tr>
<tr>
<td>Solar Radiation</td>
<td>Daily</td>
<td>Solar Radiation</td>
<td>BOM</td>
</tr>
<tr>
<td>Wetlands Management Study Report</td>
<td>N/a</td>
<td>Wetland Depth</td>
<td>Wetland Care Australia</td>
</tr>
</tbody>
</table>
3.2.1 “Exemplar” Wetland Sites

The study area, which contains the modelled wetlands, covers a length of the River Murray of just over 600 km from the South Australian and Victorian border to the entry of the river into Lake Alexandrina (Figure 12).

Figure 12: “Exemplar” Wetlands & River Monitoring Sites

The overall purpose of the model is to simulate as many wetlands as possible along the lower River Murray in order to identify management strategies that may potentially improve wetland state and function. A range of different wetlands have
been studied in the past. Of these, selected wetlands that best represent the range of
wetlands, based on hydrological connections, act as “exemplars” of driving variable
data time-series used in management simulations. The assumption is that if physically
similar wetlands respond in the way “exemplar” wetlands do it will be possible to
expand the model application and simulate the cumulative impact of multiple
management intervention. The wetlands for which data was available and serve as
“exemplar” data sources are Paiwalla, Sunnyside, Lock 6, Reedy Creek and Pilby
Creek wetlands. Their locations within the lower River Murray catchment are shown
in Figure 12.

The classification of wetlands is based on the hydrological relationship of the
wetlands to the river. This has been divided into 2 basic types, 1. through-flow
wetlands and 2. dead-end river connections. Through-flow wetlands occur where river
water can flow through the wetland, either as the wetland has one complete side open
to the river or the wetland has two distinct flow channels acting as water inflow and
outflow channels. Dead-end river connected wetlands occur where the river water
flows in and out at the only available flow channel in the wetland (i.e. one channel
only connects the wetland to the river). Both of these have furthermore been divided
into the following two categories, permanent inundation (with carp presence) and
permanent inundation (without carp presence) as well as being influenced by
agricultural drainage. In the fifth category are the managed wetlands, which are
controlled through drying and wetting cycles. These managed wetlands could be made
of either through flow or dead end wetlands, in both cases carp restriction would be
built into the wetland control barriers to restrict large carp from entering during re-
flooding of the wetland and potentially disturbing the sediment. This study addressed
five categories of wetlands as follows:

- Category 1, Through flow, permanent inundation (Paiwalla wetland)
- Category 2, Through flow, permanent inundation & irrigation drainage
  (Sunnyside wetland)
- Category 3, Dead end, permanent inundation (Lock 6 wetland)
- Category 4, Dead end, permanent inundation & irrigation drainage (Reedy
  Creek wetland)
Category 5, Managed - Dry periods & carp restriction (Pilby Creek wetland, in this case a dead end wetland)

Category 1: Through flow wetlands permanent inundation and no irrigation drainage (Paiwalla wetland)

Paiwalla and Sunnyside wetlands are situated approximately 14 km North of Murray Bridge in the lower River Murray region. Prior to swamp reclamation, the two wetlands were a part of the same riparian wetland. Selricks swamp was reclaimed in 1967 (Bartsch 1997) and was until recently used as irrigated dairy pasture. Due to the nature of the reclamation, the retired pasture area was situated lower than the average pool height of the River Murray (Philcox 1997). Both water seepage from the river and irrigation, necessitated the construction of drainage channels within the reclaimed area to remove excess water and prevent water logging. The collected irrigation drainage water was pumped into the Sunnyside wetland.

Paiwalla wetland is situated directly north or upstream of Selricks swamp (Figure 13). For the purpose of this study, as in Bartsch (1997), it was assumed that Paiwalla wetland was not influenced by irrigation drainage discharge. This assumption was justified by Paiwalla being upstream of Selricks swamp and did not receive direct irrigation drainage through active pumping. Paiwalla acts as an “exemplar” of category 1 wetlands; permanently inundated through flow wetlands with no irrigation drainage.

Category 2: Through flow wetlands with irrigation drainage (Sunnyside wetland)

Sunnyside is south of and downstream from Selricks swamp (Figure 13). Like Paiwalla wetland, Sunnyside was considered to be a through flow wetland, the main difference between the two wetlands being the influence of Selricks swamp irrigation drainage outlet that flowed directly into the northeast corner of Sunnyside. Sunnyside was used in the study as an “exemplar” for category 2 wetlands; through flow permanently inundated wetlands with irrigation drainage.
Category 3: Dead end wetlands with no irrigation drainage (Lock 6 wetland)

Lock 6 wetland (Figure 14) is a dead end wetland situated immediately upstream of Lock 6 in the Riverland region of the River Murray. Due to the controlled and constantly maintained volume of Lock 6, the wetland is permanently inundated. As with all unmanaged wetlands directly connected with the lower River Murray, there is carp presence potentially contributing to resuspension of sediment and therefore wetland turbidity. There is no irrigation drainage directly affecting this wetland. Permanent inundation and high turbidity levels have led to a reduction in macrophyte growth and therefore nutrient uptake. Lock 6 wetland is therefore, considered to be in a degraded state, with an increased possibility of blue green algae growth (Blindow et al. 1993; Boon et al. 1997; Scheffer 1998; Scheffer et al. 1993; Stephen et al. 1998), see section 1.2.

Lock 6 wetland was used in the modelling project as an “exemplar” for category 3 wetlands; dead end wetlands with no irrigation drainage. It served, in the modelling of potential management strategies (described in section 3.4), as a prime example of a
Regional Scale Modelling of the lower River Murray wetlands

wetland that has the potential of being improved through management. The management considered in modelling scenarios was the construction of a wetland weir, as found in neighbouring Pilby Creek wetland, for the introduction of dry periods.

Figure 14: Lock 6 and Pilby Creek wetlands

Category 4: Dead end wetlands with irrigation drainage (Reedy Creek wetland)

Reedy Creek wetland (Figure 15) is permanently inundated and situated approximately 6 km south of Mannum in the lower River Murray region. It is influenced by irrigation drainage runoff from surrounding agricultural areas. Intensive monitoring of this wetland over a 2-year period provided data for wetland internal nutrient dynamics, river nutrient load and irrigation drainage from Basby farm. It was used in the project as an “exemplar” for category 4 wetlands; dead end permanently inundated wetlands with irrigation drainage. The management strategies employed in simulations for this wetland (described in section 3.4) are based on nutrient reduction of irrigation drainage using constructed wetlands.
Pilby Creek wetland is found directly north of Lock 6 wetland (Figure 14). A minor through flow creek “Pilby creek” feeds into the wetland at the northern end. As this creek feeds in and out at one point of the wetland only, Pilby Creek wetland is considered to be a dead end wetland with no through flow (any wetland managed wetland is considered to fall within this category for the purpose of this project although it is recognised that through flow wetlands can also be managed). There is no irrigation drainage considered to influence Pilby Creek wetland. The introduction of a control structure and the consequent management with dry periods has dried and compacted the sediment and returned the wetland to a clear stable state (discussed in section 1.2.2). A further advantage of the management has been the exclusion of large carp through screening off of the inflow channel. The potential re-suspension of sediment by bottom feeding carp has therefore been reduced.
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Pilby Creek wetland was used in the model to simulate an ideal target condition wetland, which is considered to be in a natural, clear, non-degraded, stable state. Pilby creek was used as an “exemplar” for category 5 wetlands; dead end wetlands managed through implementation of dry periods with carp restriction and no irrigation drainage.

3.2.2 Wetland Data

The data presented in this chapter were used in developing the model as well as serving as data “exemplars” for each wetland category. The main driving variables of the model are turbidity, water temperature, solar radiation, Secchi depth and the morphological data; wetland volume and surface area. Spatially relevant driving variables include external sources of the nutrients Nitrate (as NO$_3$-N), Soluble Reactive Phosphorous (as PO$_4$-P) and phytoplankton, the external sources being river exchange; and if applicable irrigation drainage. Additional monitoring time-series of wetlands, not used in WETMOD 2 development, were used for validation and confirmation. The validation data were prepared in the same manner as the driving variable data as described below.

Time Series of Wetland Physical Condition

One of the key driving variables is wetland turbidity, which affects PO$_4$-P and NO$_3$-N sedimentation and re-suspension, as well as macrophyte and phytoplankton growth. The turbidity time-series are provided in Figure 16A, D and G. Most of the wetland data was monitored in 1997 however, Reedy Creek wetland (category 4) was monitored between 20/10/1999 and 16/09/2001, and represents the most complete and reliable study in the database.

Wetland water temperature data can be seen in Figure 16B, E and H. This driving variable affects zooplankton and phytoplankton growth, grazing and mortality, and macrophyte growth.

The Secchi depth is another driving variable required for the modelling of macrophyte growth. Secchi depth was not monitored constantly for either category 1 & 2 (Paiwalla & Sunnyside) wetlands, but assumed to be constant at 0.7 metres due to the wetland depth. In Reedy Creek wetland, a turbid wetland, the Secchi depth was assumed to be constant at 0.2 metres. The Secchi depth for Pilby Creek wetland,
being in a stable clear state where the bottom could be observed, was assumed to be at a constant 1.8 metres. The Secchi depth for Loch 6 was considered to be variable and was therefore calculated from turbidity data. Equation 1 was used to calculate Secchi depth from turbidity data and was derived from the power regression of Secchi data versus turbidity data from van der Wielen’s time-series (van der Wielen nd), where the only reliable monitoring of both had been undertaken. The $R^2$ of the power regression was 0.7748.

**Equation 1:** \[ \text{Secchi} = .4355 \cdot \text{Turbidity}^{-0.5675} \]
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Figure 16: Wetlands (Categories 1 to 5) Driving Variables Turbidity, Water Temperature & Solar Radiation (see also in Appendix B)
Climatic Time Series Solar Radiation

The solar radiation data used in WETMOD 1 were obtained from literature (Bowles et al. 1979; Cetin 2001). This literature data were adequate in the early development of the model. However, the source area of the radiation is somewhat remote from the lower River Murray and did not provide the model with reliable daily values. A CD containing solar radiation data was obtained from the Bureau of Meteorology (Forgan 2001).

Solar radiation time-series were obtained from BOM solar data, as derived from the processing of Japanese Geostationary Meteorological Satellite (GMS) imagery. The data is essentially exposure data from Meteorological Satellite Imagery collected daily. Data for any given location is obtained for the pixel encompassing the given area and is not interpolated. The resolution of each pixel is between 6x6 to 24x24 km (Forgan 2001). The BOM model calculated the surface insolation (solar radiation) from the measured upward solar radiation measured by the Visible and Infrared Spin Scan Radiometer (VISSR) taking into account atmospheric influences such as the absorption by water vapour and ozone, cloud reflection and absorption. Effectively the solar radiation is modelled for hourly images from which a daily total is derived.

For a detailed account of the model used to calculate the solar radiation refer to (Weymouth et al. 1994).

Figure 16C, F and I show the solar radiation used as driving variables in the model. Solar radiation is used in the model to calculate macrophyte and phytoplankton productivity. Unfortunately, no data were available for the period between February 1994 and July 1997, which is the period that Paiwalla, Sunnyside, Pilby and Lock 6 wetlands were monitored. However, South Australia is a very dry State with minimal cloud cover; therefore the seasonal pattern of the solar radiation for 1998 is similar to what would be expected for 1997. The intensity of the solar radiation, which impacts on macrophyte and phytoplankton biomass growth, follows such a seasonal pattern. It was found during simulation test runs of WETMOD 2 that slight variation in the solar radiation time-series does not have a noticeable impact on the simulation output. As the use of 1998 solar radiation pattern is assumed to have minimal impact on the modelling accuracy, the Solar Radiation for 1998 is used in WETMOD 2 for the simulation of Paiwalla, Sunnyside, Pilby and Lock 6 wetlands. The solar radiation
data were available for the period where Reedy Creek wetland was monitored, and was used accordingly. The solar radiation at two locations, one at either end of the study area, was adopted into the model. Simulation of either solar radiation positions did not alter the modelled output significantly. Solar radiation from the northern end of the study area was therefore used for all modelling scenarios, as this contained the least amount of missing daily values and therefore represented the most complete seasonal range of solar radiation.

*Wetland Morphology - Spatial Data*

In WETMOD 1, subjective estimated wetland volume was used in each wetland simulation. One modification for WETMOD 2 was to use a more correct wetland volume during wetland scenario modelling. The wetland volumes for all wetlands that can be potentially simulated by WETMOD 2 were obtained or estimated using the wetland surface area multiplied by the wetland depth.

The GIS data covering the wetlands and the lower River Murray and Locks were used for a number of data extractions. These GIS data reflect the wetlands as shown in the “Wetlands Atlas of the South Australian Murray Valley” (Jensen et al. 1996). The data extracted, related to wetland morphology (surface area, depth and river connection), as well as the geographical position of the wetland in relation to the river. The wetlands data sets, “Locks”, and “lower River Murray”, were also used in determination of regional scale scenarios.

Wetland volume was used in the model to calculate nutrient concentration as well as the nutrient and water exchange capacity of the wetland. Therefore, relatively accurate wetland volume estimation was required. As no DEM’s were available the surface area in conjunction with the wetland depth provided the necessary wetland volume estimation. The surface areas of the wetlands were obtained from the digitised version of the SA Wetland Atlas. The “Wetlands Management Study report” (Nichols 1998) surveyed many of the lower River Murray wetlands, and contains some data relating to average wetland depth. Many wetlands in the lower River Murray are regular in depth (Recknagel nd; van der Wielen nd), it therefore seemed justified, given the lack of better data, to assume each wetland to be a basin of uniform depth and the “Wetlands Management Study report” (Nichols 1998) depth data used in the model.
Although many of the wetlands described in the “Wetlands Management Study report” (Nichols 1998) had an estimate of their average depth, some did not have quantitative data for depth, and were simply referred to as shallow, deep, or unknown. Given that the model needs quantitative data for depth, assumptions had to be made regarding descriptive terms. For all the wetlands described as:

- shallow a depth of 0.3 metres was used in the model,
- deep a depth of 2 metres was used, and
- “unknown” an average value of 0.92 metres was used.

This last figure was calculated from the actual wetland depths presented in the “Wetlands Management Study report” (Nichols 1998).

The wetland volume was calculated using Equation 2, the results, for “exemplar” category wetlands only, are presented in Table 2. The wetland volume was used in the nutrient sector of the model (section 3.1.2) and the nutrient exchange sector (section 3.1.2).

Equation 2: $\text{Wetland Volume} = \text{Wetland Surface Area} \times \text{Wetland Depth}$

<table>
<thead>
<tr>
<th>Wetland Category</th>
<th>Wetland Name</th>
<th>Area Hectares</th>
<th>Depth m</th>
<th>Volume m³</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Paiwalla Wetland</td>
<td>49.009</td>
<td>0.7</td>
<td>343061</td>
</tr>
<tr>
<td>2</td>
<td>Sunnyside Wetland</td>
<td>27.309</td>
<td>0.7</td>
<td>191160</td>
</tr>
<tr>
<td>3</td>
<td>Lock 6 Wetland</td>
<td>17.92</td>
<td>0.92</td>
<td>164860</td>
</tr>
<tr>
<td>4</td>
<td>Reedy Creek Wetland</td>
<td>98.633</td>
<td>0.8</td>
<td>789064</td>
</tr>
<tr>
<td>5</td>
<td>Pilby Creek Wetland</td>
<td>11.991</td>
<td>1.8</td>
<td>215843</td>
</tr>
</tbody>
</table>

Time Series Irrigation Drainage

A number of the wetlands under consideration are influenced by irrigation drainage runoff to varying degrees. As irrigation drainage can be a source of high nutrient loads, this may have a significant impact on wetland nutrient content and must therefore be taken into consideration in wetland modelling. In WETMOD 1, a constant nutrient load contribution from irrigation drainage flow was assumed for all irrigation affected wetlands. The extended model includes time-series data for irrigation drainage nutrient contribution, and therefore nutrient and flow variations within the irrigation drainage.
“Wetlands Management Study report” (Nichols 1998) was again used to identify features of wetlands. In this instance where Nichols (1998) identified wetlands subject to 10% irrigation inflow or more (where 10% of inflow into a wetland can be assumed to be from irrigation areas) were considered as irrigation impacted wetlands during modelling scenarios.

Two of the wetlands (Sunnyside and Reedy Creek wetlands) considered in the development of the wetland category structure, have irrigation drainage inflow. For both of these wetlands, monitoring of the drainage inflow was included during the wetland-monitoring project. These data were used in simulation modelling of these wetlands and their respective categories.

The pump supplying the irrigation drainage to Sunnyside wetland was not observed at every monitoring date. The pumping of irrigation into Sunnyside wetland may have occurred either intermittently or daily. In either case, the volume pumped will have varied with requirements. In a situation of intermittent pumping, it is not possible to retrospectively estimate when pumping occurred, nor the nutrient concentration of the drainage water. In the absence of better data constant daily pumping was assumed based on the agricultural need to prevent water logging of reclaimed dairy pasture and the raising of water tables that can cause damage to pasture growth (Harrison 1994). The data shown in Figure 17 provides the model with additional input of NO$_3$-N & PO$_4$-P and phytoplankton to Sunnyside wetland, received as irrigation drainage. The inflow amount into the wetland can be set at a constant volume, the units being in litres per day.

The supply of irrigation drainage to Reedy Creek wetland was monitored at one inlet. The flow volume at this inlet was not monitored and an annual rate of 600 ML was estimated for this inlet into Reedy creek wetland (Wen 2002b). The inflow amount is controlled by an estimate where the volume distribution pattern is based on the relative average monthly precipitation, the distribution pattern having a mean of one over a one-year period. Therefore, the monthly drainage pattern resembles that of the average precipitation pattern. The irrigation drainage flow pattern for Reedy Creek wetland was adopted to account for the estimated load of 600 ML per annum. The irrigation and drainage multiplication factor chosen during modelling, in the case of Reedy Creek wetland, is therefore a direct multiplication of estimated nutrient inflow loads. The Reedy Creek wetland base irrigation rate of 600 ML per annum is included.
once irrigation drainage flow simulation is selected for the wetland category. Figure 17D shows the irrigation and drainage inflow pattern developed and Figure 17E, F and G the additional input of NO$_3$-N & PO$_4$-P and phytoplankton loads supplied as part of the irrigation and precipitation drainage.

**Surface Runoff Data**

As the lower River Murray flows through a predominantly arid landscape water contribution through precipitation does not account for a significant nutrient or water source for most of the wetlands, the exception being Reedy Creek wetland. Therefore, to maintain the generic nature of this model site-specific surface flow would unnecessarily complicate the model with no significant advantage to modelling scenarios. Precipitation and consequent surface flows were ignored for most wetlands in this generic model, with the exception of Reedy Creek wetland that had a separate contributing minor catchment.

Annual average rainfall in the east Adelaide hills was used to provide the seasonal precipitation pattern. This was believed to be the most appropriate source of a rainfall pattern as the east Adelaide hills is the source of surface runoff flowing into Reedy Creek sub-catchment, see Figure 17D.
Figure 17: Sunnyside Irrigation Drainage PO₄-P, NO₃-N, Phytoplankton and Estimated Flow Volume (see also in Appendix B)
3.2.3 River Data

External sources, such as river exchange, precipitation, and irrigation drainage, impact upon wetlands. The most important of these for most of the considered wetlands is river exchange. Although the river flow data are limited to the Lock locations, using this data for relatively long stretches of the river is more appropriate than using models of river flow and inundation. Around Mildura the river fall is less than 5cm per kilometre and near the sea is as little as 1.6 cm per kilometre (Mackay et al. 1990). Therefore, due to the shallow gradient of the river as it flows in its course through South Australia (Walker 1985) with alternating flow direction based on wind direction, the development of a rudimentary flow model becomes difficult and would add a complexity and inaccuracy that would compound in the generic ecosystem model WETMOD 2.

In the past, series of aerial photographs and satellite images have been used to develop a flood inundation model (FIM) (Overton 2000; Overton 2005). The data required which is water exchange between a wetland and the river, is dependent on river flow and could not be extracted from FIM for individual wetlands. The development of an estimation of the exchange volume between the wetlands and the river was achieved using WETMOD 2 in combination with river flow and nutrient load. The methodology of estimating the exchange volume, between the river and individual wetlands, is described below and is a major output of the model.

River Flow Volume

As all the wetlands considered in this model are permanently inundated and have a constant connection with the river, it was assumed that the controlling factor for nutrient exchange between the river and the wetlands is river flow volume. The river flow is monitored at each river lock and this data is presented in Table 1. The flow volume of the River Murray has been monitored daily since the construction of the Locks in the late 1920’s (Lock 6 being completed in 1930), the relevant time-series for this project was obtained from the Murray Darling Basin Commission (MDBC). This provides an important source of information that can be related to the connection between wetlands and the river and consequently exchange of nutrients.
On occasional days where river flow data were unavailable, a linear interpolation between the monitored dates was performed. However, for a fortnight in December 2000, a number of the locks failed to monitor the flow volume due to particularly high flow levels during a flood. Fortunately, the locks at the beginning and end of the river stretch under consideration recorded the flow through their location. Regression equations, based on the correlation of flow during simultaneously monitored dates in the weeks preceding the flood, were used to estimate the missing flow data based on the nearest lock with monitored flow volume. The $R^2$ values for these regression equations ranged from 0.95 to 0.99. To corroborate these estimated flow volumes, the data were compared to the flow levels monitored at an independent lock. Through this methodology, it was possible to reconstruct a probable flow volume pattern during the fortnight of high flow event through all the relevant locks. Figure 18 shows the river flow pattern and volume used in the modelling.

*River Water Quality*

The River Murray is the major nutrient source or deposit area for wetlands within the study area. The river nutrient data, monitored simultaneously with wetland data, provides a more accurate representation of the modelled situation and also a more accurate comparison of wetland vs. river nutrient load than lock monitored data. This is due to river nutrient data monitored simultaneously with wetland data was a direct indication of the river nutrient load at that time and not at a location further from the site such as at locks. In those wetlands where the river and wetland were monitored concurrently (Reedy Creek, Sunnyside and Paiwalla wetlands) only the Reedy Creek river nutrient data were comprehensive enough to be considered for the wetland modelling. Consequently, for Reedy Creek wetland it is possible to simulate the wetland with either the MDBC data or the concurrent monitored river data.

Other sources of river data were required for the remaining wetlands. River data from the same time period as wetland monitoring that could be included in the model were acquired from the sources listed in Table 1. The river data quality, obtained from DEH and MDBC see Table 1, were suitable for use in the model, see Figure 18. As with the wetland data, all the river data were extrapolated linearly to obtain daily values.
River concentrations of both NO$_3$-N and PO$_4$-P were generally higher than the concentration within the wetlands (Figure 18 and Figure 19), exceptions occurred where wetlands had a high-modelled river exchange volume. This suggests that where there is an inflow of nutrient from the river to the wetland, the river will act as a source of both NO$_3$-N and PO$_4$-P to the wetland. If the wetland processes manage to take up the nutrients in macrophyte and phytoplankton growth, and these are retained within the wetland, the water outflow from the wetland into the river would contain lower nutrient concentrations. The wetland would therefore act as a nutrient sink. For wetlands with higher concentrations of nutrients than the river, the wetlands may act as point sources of nutrients to the river.

Figure 18A and E contain the MDBC river time-series of PO$_4$-P and Figure 18I contains the Reedy Creek river time-series of PO$_4$-P. River Filterable Reactive Phosphorus as PO$_4$-P monitoring was discontinued early in the wetlands study time period, whereas Filterable Reactive Phosphorus as P was continued. As WETMOD requires phosphorus as PO$_4$-P, a linear regression was calculated between Filterable Reactive Phosphorus as P and Filterable Reactive Phosphorus as PO$_4$-P, from a time when both were monitored. Equation 3 was used to convert the monitored Filterable Reactive Phosphorus as P to PO$_4$-P. The $R^2$ for Equation 3 was 0.9988.

Equation 3: \[ PO_4 - P = 3.0575 \cdot P + 0.0004 \]

The Reedy Creek river monitored Filterable Reactive Phosphorus time-series (PO$_4$-P) could be used in the model without any conversion.

Figure 18B and F contain the MDBC and Figure 18J the Reedy Creek time-series of river Nitrate as NO$_3$-N. As with PO$_4$-P, river Nitrate as NO$_3$-N monitoring was discontinued early in the wetlands study time period. As the model input required is Nitrate as NO$_3$-N, a linear regression to obtain estimated NO$_3$-N was calculated using Equation 4 to convert from Nitrate as N to Nitrate as NO$_3$-N. The $R^2$ for the linear regression was 0.9998.

Equation 4: \[ NO_3 - N = 4.412 \cdot N + 0.0044 \]

The Reedy Creek river monitored Nitrate time-series (NO$_3$-N) could be used in the model without any conversion.
Chlorophyll-a is used as a surrogate for phytoplankton. The conversion between the two is given in Equation 5 below where \( C \) is Chlorophyll-a in \( \mu g/L \) and \( P \) is phytoplankton in cm\(^3\)/m\(^3\) (Recknagel nd).

**Equation 5:**

\[
P \approx \frac{C}{2.5}
\]

The MDBC could not supply river Chlorophyll-a, Figure 18C and G for the entire study area, as monitoring ceased in early 1998 for some locations, therefore Chlorophyll-a was not available at all monitoring locations. Only the Reedy Creek project monitored river Chlorophyll-a concurrently and comprehensively for the entire study period, see Figure 18K. However, as phytoplankton exchange plays an important role in the wetland modelling it was opted to use data from further downstream rather than none at all. Therefore, for all other wetlands the MDBC river Chlorophyll-a time-series monitored at Murray Bridge was used in the model for all river to wetland inflow. The remoteness of Murray Bridge from Pilby and Lock 6 wetlands must be taken into consideration when assessing the model simulation performance for these wetlands. The modelling of Category 4 wetlands used the Chlorophyll-a time-series obtained from the Reedy Creek wetland data. This approach was far from optimal. However, as the data was not central in the development of the model and it could be ignored during validation it was deemed acceptable during this stage of the modelling process. Through future river Chlorophyll-a monitoring this discrepancy could be remedied.
Regional Scale Modelling of the lower River Murray wetlands

Figure 18: River Murray Nutrient & Phytoplankton Time Series as well as River Flow Volume (see also in Appendix B)
3.3 Data Handling

Calculating River and Wetland Exchange

One of the major attributes of WETMOD 2 is its ability to calculate exchange rate (turnover) of water and nutrients between the wetlands and the river. The nutrient exchange between the river and the wetland is calculated for each time-step in the model. The net outflow of nutrient from the wetland is subtracted from the net inflow of nutrient. The equation for the bi-directional exchange between the wetland and the river \( N_{R} [\text{mg/day}] \) (Nutrient Retention) can be expressed as per Equation 6 with \( C_{R} \) and \( C_{W} \) denoting concentrations of nutrients in the river and wetland respectively, and \( f \) being a fraction of river flow rate \( R [\text{L/day}] \), see Figure 1.

\[
\text{Equation 6:} \quad \frac{N_{R}}{\text{V}} = \frac{(C_{W} \cdot f \cdot R)}{R}
\]

The factor \( f \) quantifies in a simple way, how the wetland is connected to the river. It summarises the complex morphology of linkage of wetlands and the river through channels, topographic conditions and distance.

The factor \( f \) is varied for each modelling scenario, and the model performance with respect to PO\textsubscript{4}-P and NO\textsubscript{3}-N is tested. The best performing scenario is chosen to represent the optimum exchange volume for a given wetland. An example of the exchange volume estimation is provided in section 4.1. The methodology for the assessment of model performance is discussed in section 3.3.1.

Based on the modelled exchange volume it is possible to estimate the wetland water turnover rate where the turnover rate (\( \tau [\text{1/day}] \)) relates to the factor \( f \), \( R \) and \( V_{W} \) as per Equation 7, \( V_{W} \) being the wetland volume.

\[
\text{Equation 7:} \quad \tau = \frac{f \cdot R}{V_{W}}
\]

The turnover rate gives a secondary method to assess the potential accuracy of the rate of exchange expected for a given wetland, see section 3.4.2. As mentioned in section 1.1.2 the potential nutrient uptake of wetlands is related to the turnover rate, i.e. the retention time.
Model Expected Simulation Output (monitored data)

As mentioned in section 3.1 the model simulates the PO$_4$-P and NO$_3$-N concentration in a wetland and the phytoplankton, macrophyte and zooplankton biomasses. The wetlands used as “exemplars” were monitored for the outputs PO$_4$-P, NO$_3$-N and phytoplankton. These output data were used to test, develop, and calibrate the model, and to adjust the exchange volume and nutrient inflow to achieve a best fit. Neither zooplankton nor macrophyte biomass were used in the first instance as these data were unavailable for comparison with model outputs and thus could not be used to assess the model. Any discussion and conclusions made based on macrophyte and zooplankton modelled biomass is limited by this lack of data and may not necessarily reflect what may occur in a natural setting. Validation of the model continued as discussed in section 4.2.

The monitored data for the different wetlands representing the categories, i.e. providing the “exemplar” data, are presented in Figure 19, which an ideal model would simulate accurately. For Paiwalla and Sunnyside wetlands Figure 19A, B and C represents the monitored PO$_4$-P, NO$_3$-N concentration and phytoplankton biomass respectively. Figure 19D, E and F represent the Lock 6 and Pilby Creek wetlands monitored PO$_4$-P, NO$_3$-N concentration and phytoplankton biomass respectively. Figure 19G, H and I the Reedy Creek wetland monitored PO$_4$-P, NO$_3$-N concentration and phytoplankton biomass respectively. At least three monitoring sites were used for each of the wetlands, usually one close to the inlet (or the river), one in the littoral zone of the wetland, and one in the open water of the wetland. The model however uses the driving variables from the open water monitoring site of the wetland. The monitored data used to test and validate the model were also derived from the open water location. To represent the variability of the wetlands and therefore the potential variability of the modelling outcome, the Standard Error was calculated for each sampling date and is displayed along with monitored concentrations in Figure 19. Only data from one sampling location in Reedy Creek was obtained (i.e. only one measurement per monitoring date), the Standard Error for the entire monitoring period had been calculated based on all sampling dates Figure 19G, H and I.
Figure 19: Wetlands (Categories 1 to 5) Monitored Nutrients and Phytoplankton
3.3.1 Model Calibration

As WETMOD 1 was substantially adapted and the driving variable database rebuilt with updated data for WETMOD 2, the model calibrations needed re-evaluation. The model was run based on its original calibrations and the optimal exchange rate established. Once the optimal exchange rate had been estimated the model output was assessed to identifying discrepancies such as unexpected trends. The model parameters identified to be adversely affecting the model output were recalibrated to account for the new data set. Many parameters calibrated in the original WETMOD 1 model were unaltered with only the following parameters being recalibrated.

- Turbidity sedimentation threshold for phytoplankton was recalibrated from 70 NTU to 95 NTU. The ones for PO4-P and NO3-N were unaltered at 70 NTU.
- The sedimentation rate for phytoplankton (pht sed) was recalibrated
- Zooplankton mortality rate (ZooMortRate) was recalibrated
- The maximum phytoplankton growth rate (Phyt max) was recalibrated

Once the model had been recalibrated the exchange rate between the wetlands and the river was reconfirmed and readjusted as appropriate.

3.3.2 Validation Procedure

It was found during the initial validation procedure that squared error estimates over-represented errors at peaks in the model output. This was seen as an inaccurate representation of a generic model where short term peak fluctuations can not be modelled. Therefore, an evaluation criterion where the average linear deviation from the measured values as a fraction of the average observed values was used and is referred to as $D$ (Equation 8). The index $D$ is derived as per Equation 8 with $M$ being the modelled and $E$ monitored PO4-P, NO3-N concentrations or phytoplankton biomass at the monitoring dates.

**Equation 8:**

$$D = \frac{\sum_{i}^{ABS} M_i - E_i}{\sum_{i}^{E}}$$

Any reduction in $D$ was considered to be an improvement in performance of a model scenario, however some improvements had a greater impact than others and should be emphasised. The following descriptive grades of improvement were adopted to better
convey the importance of each improvement. Improvements of 10% or greater were regarded as noteworthy, improvements of 20% or greater to be considerable and 30% or above to be a significant improvement to the modelling performance.

When assessing modelling performance the PO$_4$-P $D$ was valued prior to NO$_3$-N $D$ as PO$_4$-P does not escape to or return from a gaseous phase, like NO$_3$-N does in a wetland environment, and is therefore more constant in the system, see section 1.2. In scenarios where PO$_4$-P $D$ optimum performance could not be achieved due to data or modelling peculiarities, NO$_3$-N optimum $D$ performance was strived for. As mentioned in section 3.2.3 the Chlorophyll-a data, used to calculate phytoplankton biomass, were sourced from Murray Bridge. Due to this limit of location specific concentrations of phytoplankton, and the methodology for calculating the phytoplankton from Chlorophyll-a concentration, phytoplankton was never used to assess the model performance.

3.4 Wetland Management

3.4.1 Options

In the application of the model there were two management strategies simulated by WETMOD for the wetlands of the lower River Murray, turbidity reduction and irrigation drainage reduction. Scenarios were developed for potential turbidity reduction management for both Lock 6 wetland and Reedy Creek wetland. Scenarios of the second management strategy were developed for only the Reedy Creek wetland; however, it was applied both with and without the management strategy of turbidity reduction. The management strategies have two different approaches to nutrient reduction within a wetland, therefore potentially reducing both nutrient and phytoplankton outflow from the wetland.

- **Strategy; Turbidity reduction** - Construction of wetland flow control structures and grids for introduction of wetland dry periods and consequent carp restriction.

The presumed wetland response expected is sediment compaction as a consequence of a wetland drying, see section 1.3. Through grids being constructed at the wetland flow inlet large carp re-colonisation would be avoided minimising any bioturbation impact i.e. sediment re-suspension and therefore turbidity. Management simulations were
Regional Scale Modelling of the lower River Murray wetlands performed for 0% reduction in turbidity, 25%, 50% and 75% (100% reduction in turbidity regarded as unattainable).

Secchi depth increases with the reduction of turbidity; therefore the Secchi depth was altered appropriately for each assumed turbidity reduction scenario. In Lock 6 wetland management scenarios, where turbidity was reduced by:

- 25% the Secchi depth was set at 0.3 metres,
- 50% the Secchi depth was set at 0.6 metres, and
- 75% the Secchi depth was set at the wetland depth of 0.9 metres.

Strategy; Irrigation drainage nutrient reduction - Constructed wetlands for nutrient removal.

Nutrient normally entering the wetland through irrigation drainage would be diverted into constructed wetlands, where macrophytes would assist in nutrient uptake. Theoretically the harvesting of the macrophytes would remove the nutrients permanently from the system. The effective removal of nutrients can be variable, as discussed in the introduction, see section 1.3. Therefore, variable nutrient removal successes were modelled with scenarios representing 0% nutrient reduction, 25%, 50% and 75%. An example of “fully” restored wetlands with 85%, 90% and 95% nutrient reduction was also simulated.

To examine the impact of a two pronged management strategy a combination of both management interventions was simulated for Reedy Creek wetland. It was assumed that in the period prior to simulation the wetland had been dry and therefore resulted in turbidity reduction. The scenarios of the Reedy Creek twin management strategies were simulated for twelve months with no allowance made for a second dry period. Simulations were made for 25, 50, 75, 85, 90 and 95% irrigation drainage load reduction (nutrient reduction scenarios). High irrigation nutrient reductions were performed to display the hypothetical impacts of a nearly fully restored wetland. For each of the nutrient reduction scenarios, scenarios of 25, 50 and 75% turbidity reduction were also simulated. In Reedy Creek wetland simulations, the wetland Secchi depth during the turbidity reduction scenarios were adjusted to 0.2, 0.3, 0.6 metres and the maximum wetland depth of 0.8 metres used in the 75% turbidity reduction scenario.
Assessment of Management Scenario Impact

To assess the management scenario impact on nutrient retention capacity of a wetland, a comparison between the change in inflow and outflow was made, see Figure 20. The percentage Reduction of Inflow (%RI) is calculated as per Equation 13 where ID is the total Irrigation Drainage load (calculated using Equation 9). C_I denotes the concentration of irrigation drainage nutrient and I the Irrigation Drainage flow in litres/day. ΔID is the change in total Irrigation Drainage load after management and RF the total River Inflow load, RF is calculated as per Equation 10. Equation 14 calculates the percentage Reduction in Outflow (%RO), where OF is total Outflow load (calculated as per Equation 11), and ΔOF is the change in total Outflow load post management. The OF for Reedy Creek wetland and category 4 wetlands is calculated by Equation 12 to account for the additional irrigation flow volume exiting the wetland. As in Equation 6, C_R and C_W denote concentrations of nutrients in the river and wetland respectively and f represents a fraction of the river flow rate R.

Equation 9: \[ ID = C_I \cdot I \]
Equation 10: \[ RF = C_R \cdot f \cdot R \]
Equation 11: \[ OF = C_W \cdot f \cdot R \]
Equation 12: \[ OF = C_W \cdot (f \cdot R + \Delta ID) \]
Equation 13: \[ %RI = 100 - (\Delta ID + RF \cdot C_I + RF \times 0) \]
Equation 14: \[ %RO = 100 - (ΔOF + OF \times 0) \]

The %RO Equation 14 above is therefore, the change in outflow due to management when compared to the status quo (no management). With a positive %RO there is a net improvement of the nutrient or phytoplankton retention of the wetland due to management. The %RI Equation 13 only applies to Reedy Creek and category 4 wetlands and represents the effective change in wetland nutrient inflow due to nutrient reduction scenario as compared with the status quo.

The impact of water loss through other means, specifically evaporation, has not been included in the mass balance equations. The current method of evaporation estimation is itself inaccurate and would have added further complications, to model calibration and validation, than is acceptable at such an early stage of the model development. This is an aspect that can in future be included in the model when full monitoring (of
at least one wetland) including all water sources, sinks (including evaporation) and nutrient balance becomes available to effectively calibrate and validate the model.

Figure 20: Wetland exchange modelling

3.4.2 Management scenarios for cumulative assessment

Wetland candidates for simulations

There are more than a thousand individual wetlands in the lower River Murray, ranging from small, temporary wetlands to large and more permanent examples. However, of this multitude of wetlands, only 250 individual wetlands or groups of closely related wetlands (complexes) are identified in the ‘Wetlands Atlas of the South Australian Murray Valley’ (Jensen et al. 1996). For the purposes of this project, the 250 identified wetlands were perused with the intent of consideration for management. In the cumulative assessment of management scenarios two wetland categories were considered, these being category 3 (wetlands resembling Lock 6 wetland) and category 4 (wetlands resembling Reedy Creek wetland). Identified wetlands were assigned to a particular category, depending on their similarities to the
Lock 6 and Reedy Creek "exemplars", with each category having a defined management strategy.

In the lower River Murray 54 of 250 wetlands (wetland groups) were identified as being similar to Lock 6 wetland and therefore classified as category 3 wetlands. Including Lock 6 wetland, 35 were found to be over 0.6 metres depth, the minimum depth of wetlands found to be effectively simulated by WETMOD 2. These 35 wetlands and wetland groups, make up a total of 57 individual lagoons that can be simulated within WETMOD 2 (a list of these wetlands is provided in Table 18 in Appendix C). The method for Secchi depth adjustment in cumulative wetland management scenarios was handled in the same way as for Lock 6 wetland simulations discussed in section 3.4.1.

Due to the nature of Reedy Creek wetland, more stringent restrictions had to be placed on the wetlands that could be regarded as potential category 4 wetlands. If wetlands less than half the volume of Reedy Creek were simulated using the exchange volume found for Reedy Creek then the average volume exchanged per day would exceed the total wetland volume. When the exchange volume exceeds the wetland volume the nutrient retention time within the wetland is reduced below that of the model time-step. WETMOD 2 has not been developed nor calibrated for such a continual high exchange volume. WETMOD 2 was therefore restricted to simulation of wetlands where the average exchange volume is below that of the wetland volume. Consequently, due to the high river exchange volume estimated for Reedy Creek wetland, category 4 modelled wetlands are restricted to those with a volume greater than half the volume of Reedy Creek wetland.

A further restriction, for wetlands to be considered for management scenarios of category 4 wetlands, was based on the irrigation flow volume. Reedy Creek wetland was estimated to receive a high volume of irrigation drainage flow. Therefore, wetlands that were deemed to receive only a low volume of irrigation drainage flow were also excluded from management consideration. Therefore, 7 of the 250 wetlands (wetland groups), including Reedy Creek wetland, were identified as being category 4 wetlands for which WETMOD 2 had the potential capacity to reliably simulate (a list of these wetland is provided in Table 19 in Appendix C). This did not include the potential irrigation drainage nutrient concentration that these wetlands may receive, as
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this information was unavailable, Reedy Creek irrigation data was therefore used as the driving variables.

Exchange volume

During the simulation of wetland management scenarios using “exemplar”-driving variables, the wetland volume is changed to reflect the wetland that is being simulated. However, the exchange volume between the wetlands and the river was maintained at the same percentage of river volume as was estimated for the “exemplar” wetland (i.e. the wetland which provided the driving variable data). For category 3 wetlands the river exchange was maintained at 0.1% of the river flow volume per day, this being the volume fitted for Lock 6 wetland. For category 4 wetlands the fitted exchange rate for Reedy Creek wetland of 3.5% of the river flow volume per day was used. This fitted volume for each of the two categories was maintained based on the assumption that all category wetlands resemble each other unless specific data is available. Consequently future improvement of the model could be achieved with a proper estimate of individual wetland water exchange with the river, thereby providing improved wetland scenario accuracy.

For each category wetland scenario the driving variables for the river data are sourced from the nearest upstream monitoring location, the exception being Reedy Creek wetland which has its own monitored nutrient river data set. Therefore, the flow volume was adjusted below each successive lock and the river nutrient data was adjusted to each individual nutrient monitoring locations. The behaviour of wetlands of a particular category was expected to be similar, particularly where the only major difference between the wetlands is the morphology.

Implication of the change in nutrient retention capacity on river nutrient load

Through the management of both category 3 and category 4 wetlands, a cumulative impact on the river nutrient load would become evident. Although the modelling accuracy of category wetlands allows only a qualitative understanding of the trends expected due to wetland management and not quantitative accuracy, the model results will, for this section, be assumed to be quantitatively accurate. The rationale is two fold.
First, although the results are not quantitatively accurate the assessment of the quantitative output helps to develop a qualitative trend analysis of the cumulative impact of management.

Second, although this model, due to the poor data quality, is of low quantitative accuracy the methodology of assessing the cumulative impact could be applied in the same manner should the model quantitative performance improve through future data improvement.

However, this assumption is made in order to understand and discuss the potential cumulative impact on nutrient loads within the river, and should only be seen as a trend analysis.

To understand the cumulative impact that the management of multiple wetlands would have on the river nutrient load, the change in wetland nutrient retention capacity was compared to the river load. To this purpose the initial river nutrient load ($L_R$) was required see Equation 15, see Figure 21.

**Equation 15:** $L_R = C_R \cdot R$

The initial river load is calculated from the first available monitoring locations post inflow into South Australia, i.e. the flow volume data is obtained from Lock 6 whereas the river nutrient concentration is obtained from Lock 5. The calculation of the river nutrient load based on the earliest available monitoring locations was chosen so that the river data would not reflect the status quo impacts of the wetland that are simulated, i.e. wetland impacts would otherwise be counted both status quo and as per management scenario.

The wetland nutrient retention calculation is similar to Equation 6 (see Box) where the retention in the wetland is calculated per day. Equation 16 needs to calculate the sum over the modelled period for each of the management scenarios. The status quo (i.e. no wetland management) subtracted from the nutrient retention in the wetland as per a management scenario, gives the change in nutrient retention ($\Delta R_N$) due to management. Where, $\Delta R_N$ is the change in wetland retention due to management and is calculated as per Equation 16 where $\sum_{\Delta t} N_{sq}$ is the nutrient retention at the status quo scenario and $\sum_{\Delta t} N_{ms}$ the nutrient retention at the respective management scenario.
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### Equation 6:
\[
\frac{N_x}{\Lambda} = (C_s - W_r) \cdot \frac{F_r}{W_r}
\]

### Equation 16
\[
\Lambda \Delta i_x = \sum \Delta \xi \cdot ms - \sum \xi aq
\]

The \( \Delta i_x \) was used to calculate the change in river load where the % River Load removed due to the wetland management (%RL) is calculated as per Equation 17, see Figure 21.

### Equation 17:
\[
\% RL = \frac{\Lambda i_x}{L_R}
\]

Equation 16 and Equation 17 are used to calculate the impact of a single wetland on the river nutrient load as well as the cumulative impact the management of multiple wetlands would have on the river nutrient load, see Figure 21.

---

**Figure 21: Cumulative assessment of wetland processes**

\[
\frac{L_R}{\% RL} = \frac{1}{L_R} \left( \frac{1}{i_{R1}} + \frac{1}{i_{R2}} + \frac{1}{i_{R3}} + \cdots + \frac{1}{i_{Rn}} \right)
\]
4 Validation of the model WETMOD 2 and Discussion

During the development of WETMOD 1, neither river flow nor nutrient load data were available. To varying extents the wetlands are reliant upon the exchange of water and nutrient with the river. The addition of river flow and nutrient load as well as the exchange volume between the wetlands and the river is therefore a significant improvement of the WETMOD 2 model. This chapter will test the first hypothesis of whether “a simplified generic wetland model can be used to realistically simulate multiple and different wetlands qualitatively”.

4.1 Fitting and Validation based on calibrated (“exemplar”) wetlands

The results presented in this chapter show the validation steps used for WETMOD 2 using data from the five different wetlands. The validation of WETMOD 2 is based on $D$ (Percentage Deviation of modelled time-series from monitored time-series) for $\text{PO}_4$-$\text{P}$, $\text{NO}_3$-$\text{N}$ and phytoplankton, and is represented in Table 3.

River water quality is influenced by adjacent wetlands. The water exchange estimate is a step in the process of developing a model capable of simulating management strategies for wetlands of the lower River Murray and their impact on nutrient load in the river. WETMOD 2 was used to find the water exchange between wetlands, where there is a lack of channel morphology data and no measured wetland water turnover. The added spatial driving variables for WETMOD 2 are used to account for local variations and inflow into a wetland, particularly to reflect bi-directional water and nutrient exchange between the River Murray and the wetlands (see section 3.3). This was based on a combination of the river flow volume and the wetland specific budget of $\text{PO}_4$-$\text{P}$ or $\text{NO}_3$-$\text{N}$ simulated by WETMOD 2. Through this methodology it is possible to obtain the turnover volume of water in a wetland using nutrient modelling output (Bjornsson et al. 2003).

The optimal river exchange estimate was determined by WETMOD 2 based on the best percentage deviation ($D$) (see box). Given the availability of accurate daily river flow data as well as fortnightly nutrient data, it was possible to estimate the flow of nutrients carried by the lower River Murray. This provided accurate data for the estimation of the most significant external nutrient source, i.e. the river. Combined
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with successive calculations of wetland internal nutrient load by WETMOD 2, the wetland simulation results improved until the optimum exchange was attained. Once the external load was increased past the optimum the wetland simulations degraded, see Figure 22.

The lower the $D$ the closer the fit of modelled data to monitored data.

As discussed in section 3.3.1, $\text{PO}_4\text{-P}$ was in most cases used as the primary indicator of model $D$ as $\text{PO}_4\text{-P}$ is the most reliably modelled and monitored nutrient within the system (once $\text{PO}_4\text{-P}$ enters a wetland it is not diminished through a gaseous state). The flow exchange between a wetland and the River Murray was mostly estimated based on the model percentage deviation ($D$) calculation of $\text{PO}_4\text{-P}$, with $\text{NO}_3\text{-N}$ only used for Lock 6 wetland. Figure 22 presents an example of the selection of $D$ for Reedy Creek wetland. In this example the $\text{PO}_4\text{-P}$ shows the best fit at a river exchange of 3.5% of the daily river flow volume.

![Figure 22: Percentage Deviation based estimate of flow exchange: Reedy Creek wetland](image-url)
Wetland modelled results are presented and discussed in the sections below. Each wetland is assessed independently, and some comparisons are made.

**Category 1: Through flow wetlands with carp presence and no irrigation drainage (Paiwalla wetland)**

Paiwalla wetland is situated upstream of Sunnyside wetland (see section 3.2.1), with an area of reclaimed ‘swamp’ situated between them, which was used as dairy pasture prior to 1997 (refer to map in chapter 2). The runoff from this pasture was pumped into Sunnyside wetland and thereby transported nutrients from the irrigation drainage into Sunnyside wetland. In contrast there was no direct input of nutrient from the dairy pasture into Paiwalla wetland (Bartsch 1997). Paiwalla wetland was therefore...
chosen to represent through flow wetlands with possible carp presence and no irrigation.

The comparison between modelled and monitored concentrations of PO$_4$-P is seen in Figure 23A and Figure 23B for NO$_3$-N; macrophytes, zooplankton and phytoplankton are represented in Figure 24A, B and C respectively. Each graph of Figure 23 and Figure 24 includes results for the scenarios “no flow exchange” and “optimum flow exchange”. For monitored data in Figure 23 and Figure 24, error bars represent the standard error for measurements made on that date.

As seen in Table 3 Paiwalla modelling results for PO$_4$-P and phytoplankton improved due to the consideration of river exchange, phytoplankton result being significant. Figure 23A reflects this improvement. The NO$_3$-N $D$ shown in Table 3 does not show an improvement, however the graph in Figure 23B indicates a distinctive change in the model output due to river exchange. The NO$_3$-N variability, range and seasonality are realistically reflected by the river exchange scenario. It is therefore concluded that the model validation improved with regard to qualitative trends even though the quantitative accuracy is not optimal. There is a major improvement in the modelling results for phytoplankton following the introduction of river exchange. The modelled $D$ for phytoplankton (Table 3) is the best result of all output from modelled wetlands and scenarios; this modelling performance is also being displayed in Figure 24C where the modelled phytoplankton corresponds well with the trends of the monitored phytoplankton. There is some early macrophyte biomass growth in Paiwalla wetland however; there is a rapid decline due to increasing turbidity, see Figure 24A. Phytoplankton growth, as seen in Figure 24C, increases as expected following diminished macrophyte competition. As the solar radiation and wetland water temperature increase in spring, the growth of phytoplankton increases accordingly (Figure 24C). Zooplankton biomass increases in response to the growth of phytoplankton. This is due to the phytoplankton serving the zooplankton as a food source (Figure 24B) following the typical Lotka-Volterra predator-prey cycle as discussed in the introduction.

Through flow wetlands are highly variable due to the close link to the river and are therefore difficult to model with a simplistic model such as WETMOD. Although the modelling results for this category of wetlands were not as good as expected there was an improvement in the model output for Paiwalla wetland due to the introduction of
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river exchange. It shows the potential of simplistic models to assess the exchange volume of water and nutrients between riparian wetlands and the river.

Figure 23: Validation of simulation results for Paiwalla wetland of PO_4-P, and NO_3-N for both conditions with and without water exchange
Figure 24: Validation of simulation results for Paiwalla wetland of Macrophyte Biomass, Zooplankton and Phytoplankton for both conditions with and without water exchange.
**Category 2: Through flow wetlands with carp presence and irrigation drainage (Sunnyside wetland)**

Figure 25A portrays the PO$_4$-P and Figure 25B the NO$_3$-N monitored and modelled concentrations for Sunnyside wetland. Figure 26A, B and C depicts macrophyte, zooplankton and phytoplankton monitored and modelled concentrations respectively. The monitored concentrations of PO$_4$-P, NO$_3$-N and phytoplankton in the wetland, and of PO$_4$-P and NO$_3$-N concentrations in the irrigation drainage, are represented in Figure 25A and B and Figure 26C. For each monitored concentration, error bars represent the standard error for measurements. Each graph includes results of scenarios where no river flow exchange and no irrigation drainage were considered. Another trendline in each of the graphs includes river flow exchange estimated at a modelled best-fit $D$ (Table 3), according to monitored wetland nutrient concentration. This scenario was re-run with irrigation drainage included. To estimate the impact of irrigation drainage on the wetland simulation Sunnyside wetland was also simulated with only irrigation drainage influencing the scenario results and no river exchange. The response of the scenario where irrigation drainage inflow was the only outside nutrient source was minimal and effectively covers the simulation where no outside nutrient source was considered (Figure 25 and Figure 26).

Sunnyside wetland is an “exemplar” for the category 2 wetlands considered in the modelling project, which are wetlands having river water through flow and are directly affected by irrigation drainage. Simulation results demonstrated that an improvement in $D$ of only 0.01 was evident when a realistic volume of 500L of irrigation drainage per day was included in a scenario. In order to clarify the reason for this result, we must look at both assumptions made at the start of the modelling project as well as the monitoring design; this is discussed in section 4.1.1.

A better scenario of a wetland with irrigation drainage inflow in this wetland category is not possible due to the limited data available. However, the small response of the model to scenarios with drainage nutrient and the success of modelling Reedy Creek wetland with its irrigation drainage (described below in category 4 wetlands), indicate the possibility of a more successful modelling scenario when adequate data for this wetland category become available.
The modelling of $\text{PO}_4^-$-P (Figure 25A) does not pick up the early high wetland concentration monitored, neither with nor without the river exchange and irrigation drainage. However, with the introduction of river exchange there is a slight improvement in the trend modelled, as can be seen in the results between the months of May to June in Figure 25A. The improvement in the modelling trend of $\text{NO}_3^-$-N due to the introduction of river exchange can similarly be seen in Figure 25B. The $D$ for $\text{PO}_4^-$-P, $\text{NO}_3^-$-N and phytoplankton (Table 3) does improve with the introduction of river exchange, with a noteworthy improvement for $\text{NO}_3^-$-N and phytoplankton, however this improvement is not great. As mentioned, better data is required to successfully model this wetland.

There was a longer growth period of macrophytes in Sunnyside wetland than in Paiwalla wetland simulations (Figure 26A and Figure 24A respectively). Again, this can be attributed to the turbidity of the wetlands. The delayed increase in turbidity in Sunnyside wetland extended the growth period for the macrophytes. The growth in zooplankton and its high concentration (Figure 26B) is probably due to the shelter provided by macrophytes (Figure 26A), the first zooplankton growth phase followed by the increased food source phytoplankton in the second growth phase (Figure 26C).

The combination of simulated nutrient competition by macrophytes and grazing by zooplankton restrict the initial growth of the phytoplankton (Figure 26). The major growth phase of phytoplankton simulated occurs from May to July corresponding well with the monitored trend (Figure 26C).
Figure 25: Validation of simulation results for Sunnyside wetland of PO₄-P, and NO₃-N for both conditions with and without water exchange

For both Figure 25 and Figure 26 the grey line (modelled concentration (PO₄-P or NO₃-N) with 0.06% river exchange and no irrigation) falls behind the green line (modelled concentration (PO₄-P or NO₃-N) with 0.06% river exchange and 500L irrigation drainage inflow). The blue line (modelled concentration (PO₄-P or NO₃-N) with no river exchange and no irrigation) falls behind the pink line (modelled concentration (PO₄-P or NO₃-N) with no river exchange but with 500L irrigation drainage).
Figure 26: Validation of simulation results for Sunnyside wetland of Macrophyte Biomass, Zooplankton and Phytoplankton for both conditions with and without water exchange
Regional Scale Modelling of the lower River Murray wetlands

Category 3: Dead end wetlands with carp presence and no irrigation drainage (Lock 6 wetland)

Figure 27 and Figure 28 depict the modelled output of Lock 6 wetland for PO$_4$-P, NO$_3$-N, macrophytes, zooplankton and phytoplankton respectively. The error bars represent the standard error for the monitoring at that particular date based on three separate measurements.

Lock 6 wetland is a permanently inundated wetland situated adjacent to Lock 6 of the River Murray. It is a wetland classified as a “dead end” wetland. This wetland’s hypothetical management strategy was drying and compacting the sediment. Therefore, it was assumed for the modelling that following a re-flooding event the sediment re-suspension and wetland turbidity would be reduced (see section 3.4).

As there is no irrigation drainage flowing directly into Lock 6 wetland, only the river exchange volume was considered as an external influence upon this wetland. It was expected that all output parameters would have an improved response. It is possible that the high PO$_4$-P level modelled in the wetland was overestimated due to relatively high river concentrations. However, the trend was clearly modelled correctly when compared to monitored concentrations (Figure 27A) despite the $D$ indicating a worse fit (Table 3). This discrepancy is also reflected in the modelling result of the phytoplankton (Figure 28C). This shows that although the $D$ is a good method of finding the best-fit scenario during modelling, it is by no means a perfect method and model results should be analysed with an understanding of the expected trends. The modelling performance of NO$_3$-N was improved considerably by the introduction of river exchange, as seen in Table 3 and Figure 27B, and is the best modelling response of NO$_3$-N for all wetlands and scenarios.

Due to the high turbidity levels of Lock 6 wetland, the modelled macrophyte growth is inhibited showing that the original estimate of the potential macrophyte biomass used in the modelling scenario was probably overestimated (Figure 28A). It can be assumed that the high turbidity levels limited underwater light for macrophyte growth. However, due to the high nutrient levels within the wetland (Figure 27), and the lack of competition provided by the macrophytes, the phytoplankton were able to grow effectively (Figure 28C), reaching a peak biomass prior to the onset of winter. The lack of the spring growth phase can be attributed to the large volume of Lock 6
Regional Scale Modelling of the lower River Murray wetlands

wetland effectively buffering early rise in water temperatures. It must be remembered that the river chlorophyll-a used in calculating the river phytoplankton, which is consequently used in representing the exchange rate inflow into the wetland, was not available for this part of the river. As phytoplankton has a significant role in this model, its part in the wetland simulations could not be ignored. The zooplankton growth (Figure 28B) in Lock 6 wetland follows the phytoplankton growth as expected, and declines during the winter months.

Figure 27: Validation of simulation results for Lock 6 wetland of PO₄-P, and NO₃-N for both conditions with and without water exchange
Figure 28: Validation of simulation results for Lock 6 wetland of Macrophyte Biomass, Zooplankton and Phytoplankton for both conditions with and without water exchange.
Category 4: Dead end wetlands with carp presence and irrigation drainage (Reedy Creek wetland)

The Reedy creek wetland data set monitored by Wen (2002a) includes time-series for the water quality of the wetland, the River Murray and the irrigation drainage originating from the adjacent Basby farm. A period of 12 months with high internal wetland nutrient variability (1st Jun 2000 to 31st May 2001) was chosen from the data set, to represent the condition of Reedy Creek wetland. Figure 29A & B and Figure 30A, B & C contain the simulated results for PO$_4$-P, NO$_3$-N, macrophytes, zooplankton and phytoplankton respectively for Reedy Creek wetland. The monitored concentrations for PO$_4$-P, NO$_3$-N, and phytoplankton are displayed in Figure 29A, B and Figure 30C; the error bars represent the mean error for the entire monitoring period of 20th October 1999 to 16th September 2001.

A limitation of the drainage inflow time-series is that it was obtained from one source, that being a small drainage inflow from Basby farm. The catchment area of Reedy creek is 315 km$^2$, whereas Basby farm covers an area of 85ha (0.85 km$^2$) (Wen 2002a). The Reedy Creek catchment area results in significant natural flows and nutrient loadings to Reedy Creek wetland in response to precipitation. Unfortunately, no monitoring data existed of the nutrient inflow from Reedy Creek, as this was not required for the project responsible for the monitoring. Its contribution was therefore approximated by higher surface runoff and irrigation drainage into Reedy Creek wetland than was monitored at the one source; inflow from Reedy Creek catchment is known to grow to a substantial amount following rains in the region (Frears 2006). Accordingly it was assumed that the expected seasonal precipitation (described in section 2.3.1) would have reflected the relative seasonal flow pattern over the modelling timeframe. The monitored drainage source would have reflected the average concentration of nutrients per unit volume expected from surrounding farms contributing to the Reedy Creek. In order to determine the most appropriate flow, multiple scenarios were run each with an increasing multiplication of the irrigation volume entering the wetland. The best fit was chosen depending on the deviation of modelled values from the monitored values $D$ (Table 3). As with previous wetlands the best values $D$ for the river exchange was separately modelled.
As seen in Figure 29A & B and Figure 30C there was a significant improvement in the modelling results of both PO$_4$-P and NO$_3$-N, and a considerable improvement on the modelling of phytoplankton. The PO$_4$-P results in Reedy Creek wetland improved clearly through the introduction of the irrigation drainage inflow; however the $D$ (Table 3) shows the river exchange flow to have the greater impact. As can be seen in Figure 29A this result is skewed by a particularly good fit for a short period from March to the end of May. The combination of both river exchange flow and irrigation drainage not only produced the best $D$ for both PO$_4$-P and NO$_3$-N, but also showed a better fit when the trend is observed as seen in Figure 29A. The Reedy Creek PO$_4$-P modelling shows the most significant improvement of PO$_4$-P simulation when compared with the other modelled wetlands. NO$_3$-N is influenced by both the river flow exchange and the irrigation drainage inflow to produce a significant improvement in model fit $D$ (Table 3). The phytoplankton modelling of Reedy Creek wetland shows a considerable improvement in $D$ through the introduction of river exchange and drainage.

Some of the extreme events in PO$_4$-P and NO$_3$-N concentrations from October to December (Figure 29A) were not realistically simulated by the model, although the trend is clearly visible. A limitation of the generic nature of the model WETMOD2 may be that short lived and extreme events cannot be successfully simulated.

Reedy Creek wetland is in a turbid state with minor macrophyte growth (section 3.2.1). The macrophyte growth curve shown in Figure 30A is a result of the high turbidity, which limits underwater light for growth. The zooplankton, lacking the shelter assumed to be provided by macrophytes, are reliant on the phytoplankton as their food source. The zooplankton growth, seen in Figure 30B, closely follows the phytoplankton growth seen in Figure 30C. As seen in Figure 30C, a combination of both river exchange and irrigation drainage inflow was required for phytoplankton to resemble the monitored and therefore expected concentrations. This further strengthens the validation of the model, showing that one external influence such as the river exchange is not enough to drive the simulation for a wetland such as Reedy Creek wetland. But rather the combinations of external influences such as the river flow exchange and irrigation drainage are required to successfully and comprehensively simulate the Reedy Creek wetland.
Figure 29: Validation of simulation results for Reedy Creek wetland of PO₄-P, and NO₃-N for both conditions with and without water exchange.
Figure 30: Validation of simulation results for Reedy Creek wetland of Macrophyte Biomass, Zooplankton and Phytoplankton for both conditions with and without water exchange.
Category 5: Dead end wetlands managed through implementation of dry periods with carp restriction and no irrigation drainage (Pilby Creek wetland)

Figure 31 represents simulation results for PO$_4$-P and NO$_3$-N concentrations in Pilby Creek wetland and Figure 32 the simulation results for macrophyte, zooplankton, and phytoplankton biomass within the wetland. The error bars represent the standard error, of three separate measurements, of the monitored concentration for each monitoring date.

Pilby Creek wetland is a dead end wetland adjacent to Lock 6 wetland (Category 3). Pilby Creek wetland is managed by artificial drying and wetting cycles resulting in sediment compaction. Restriction on the presence of large bottom-feeding fish such as carp, which are believed to stir up wetland sediment, is also believed to have contributed to reduced turbidity. The case study for Pilby Creek wetland was included in the modelling project to test the model validity for a restored wetland.

Although Pilby Creek wetland is not directly connected to the river, as well as being a dead end wetland, an exchange of water and nutrient with the river was assumed. The justification for this assumption is the possibility of an exchange through Pilby creek, which flows through at one end of the wetland (see Figure 14). The possible nutrient load change during the exchange through an intermediary creek should be taken into consideration when assessing the modelling success of this wetland. The model results support the assumption of water exchange through Pilby creek, as the model scenario $D$ improves with the introduction of river flow exchange (Table 3). The $D$ shows a considerable improvement for the PO$_4$-P modelling (Table 3). The peak concentration of PO$_4$-P simulated by the river exchange scenario (Figure 31A) was due to both a high peak in river flow and high river PO$_4$-P concentration (see section 3.2.3). This nutrient peak did not reach the wetland during the monitoring period as indicated by the internal wetland nutrient monitoring (Figure 31A), which may be due to the lag time of nutrient flow to Pilby Creek wetland from the River Murray. The NO$_3$-N curve is lower than expected during late February until April. However, with the exception of an extreme event at the end of April the curve does show a similar trend to that of monitored concentrations (Figure 31B), which is not as apparent in the simulation without the river exchange. The improvement in NO$_3$-N simulation is also reflected by the $D$ value (Table 3).
Following a drying period of two months in 1997, of Pilby Creek wetland, that was long enough to compact the sediments the high macrophyte growth seen in Figure 32A was a result of low turbidity as expected in a managed wetland within a short time after re-flooding. The macrophyte biomass decreased over the winter months with low water temperatures but increased during spring. Monitoring ceased at the beginning of October.

The observed phytoplankton growth in Figure 32C showed a rapid growth phase prior to the macrophyte growth, directly following wetland re-flooding. In this instance, the phytoplankton took advantage of the lack of competition as well as the high nutrient availability. Once competition set in with the growth of macrophytes, there was a reduction in the phytoplankton biomass. The phytoplankton biomass growth was thereby restricted until the decreasing macrophyte biomass in winter when phytoplankton again took advantage of less nutrient competition and increased its biomass. The phytoplankton had a faster response time in growth than macrophytes at the onset of the warmer period of spring. As with Lock 6 wetland, the river phytoplankton was derived from river chlorophyll-a levels monitored further downstream.

The zooplankton growth can be linked to the provision of a nourishment source, the phytoplankton growth, and possibly to a lesser extent the assumed provision of a shelter from predators by macrophytes (Figure 32). The lowest number of zooplankton occurred when there was a combination of both low phytoplankton and low macrophyte biomass. The lack of phytoplankton as a food source explains the reduction in zooplankton observed despite the potential supply of shelter provided by the macrophytes. The secondary growth phase of zooplankton corresponded to the secondary growth phase of the phytoplankton. During the spring growth phase of phytoplankton the zooplankton follows suit, again possibly as a consequence of shelter provided by the increase in macrophyte growth. The modelled growth behaviour of the macrophytes, phytoplankton and zooplankton described follows expectations of a wetland in the Pilby Creek wetland category (category 5). It must however be remembered that no data was available to validate model output for zooplankton and macrophyte biomass.

It is interesting to note that the growth of phytoplankton in a category 5 wetland (Pilby creek) was less than in a category 3 wetland (Lock 6). This can be attributed to
the competition between the macrophytes and the phytoplankton in Pilby Creek wetland, which is virtually absent in Lock 6 wetland. However, Pilby Creek wetland shows a relatively greater zooplankton growth than Lock 6 wetland when compared to the phytoplankton availability in each of the wetlands. The cause of the relatively larger zooplankton growth in Pilby Creek wetland may be as a consequence of added shelter opportunity within Pilby Creek wetland assumed to be provided by the macrophytes. The only discrepancy in the modelling of Pilby Creek wetland is the very late spike in PO$_4$-P levels described earlier, attributed to river flow and river nutrient concentration.

Figure 31: Validation of simulation results for Pilby Creek wetland of PO$_4$-P, and NO$_3$-N for both conditions with and without water exchange
Figure 32: Validation of simulation results for Pilby Creek wetland of Macrophyte Biomass, Zooplankton and Phytoplankton for both conditions with and without water exchange.
4.1.1 Implication for irrigation affected wetland representation

Considering the generic nature of the model and its structural restrictions and how this interacts with potential quantitative modelling performance, the qualitative modelling performance, the time and data available for model development and most importantly the project goals, the model displays the potential of a developed tool with purpose designed monitoring scenarios. The following discussion aims to represent the performance of the model in a dispassionate approach, focusing on where it has succeeded in fulfilling its objective and is at a stage where it can be applied to answer wetland specific management questions and therefore fulfilling the project aims.

Category 4: Dead end wetlands with carp presence and irrigation drainage (Reedy Creek wetland)

The modelling results from Reedy Creek wetland are an example of a successful simulation of a wetland that is affected by irrigation drainage. Both the quality of the trend as well as the statistical comparison improved with the introduction of irrigation drainage. The methodology of estimating the inflow volumes from the Reedy Creek catchment can at this stage not be confirmed as no monitoring of the sub-catchment inflow was performed concurrent with the wetland-monitoring project. However, although the inflow volume used in the model may be debateable, the methodology of the model derived optimum level gives future modellers the option to adjust the scenarios as this data becomes available. Any consequent monitoring could potentially refute or confirm the range of estimated nutrient and volume inflow. It is therefore not regarded as a high priority at this stage to invest expense and time in the improvement of the Reedy Creek wetland modelling scenarios. The validation of the macrophyte and zooplankton modelling output may however increase the confidence in the model. Future monitoring could assist in this regard by providing adequate data for model validation.

Category 2: Through flow wetlands with carp presence and irrigation drainage (Sunnyside wetland)

Modelling scenarios of Sunnyside wetland improved with the introduction of river exchange. However, the inflow of monitored irrigation drainage did little to improve
the scenario performance. The monitoring of Sunnyside wetland was not designed with this project in mind. Bartsch (1997) designed her monitoring project with the sole intention of comparing the two wetlands Paiwalla and Sunnyside; and therefore study the impacts irrigation drainage has had on Sunnyside wetland. Minor effort was therefore made to assess internal wetland dynamics by that project. Due to Bartsch’s (1997) project aims, and particularly the need to assess the impact of irrigation drainage into Sunnyside wetlands, most of the monitoring sites were located at one end of the wetland and close to the drainage outlet. The monitoring of nutrients was made mainly in, what can be recognised in aerial photos as, a channel through the macrophyte growth leading from the irrigation drainage outlet to the river (see Figure 33).

Figure 33: Sunnyside monitoring area
One of the central assumptions made in this modelling project is that all wetlands are homogeneously mixed from one time step to the next. However, partly due to the sporadic point source inflow of nutrients (irrigation drainage) the concentrations of nutrients are highly variable within the wetland. Further, the nature of Sunnyside wetland, macrophyte growth within the wetland and the channel through the
macrophytes from the irrigation drainage source to the river (see Figure 33), hampers the mixing of water and nutrients within the wetland. Sunnyside wetland, due to its highly variable nature, can therefore not be considered as homogeneously mixed. Monitoring within and close to the channel will therefore represent the concentration of nutrients entering the wetland. If this concentration is then assumed to encompass the entire wetland, the true concentration and particularly the inflow of irrigation drainage will be over represented. The monitoring of irrigation flow did not include volume. It is therefore not possible to estimate the true impact the irrigation drainage has on the concentration of nutrient within the wetland.

Another issue impacting on the use of the model to estimate a realistic irrigation inflow scenario may be due to the drainage inflow being sporadic (infrequent and short-lived), despite our assumptions of daily pumping. However, the methodology available to estimate the best-fit scenario shows a slight change in model fit as the drainage inflow only affects a minimum number of monitored dates. That does not mean the model proves there to be an insignificant detrimental impact on the wetland due to drainage, but rather that the infrequent nature and the unknown exact drainage volume for each particular pumping date complicates the modelling estimate. The cumulative impact of the drainage inflow on the wetland would however still persist as suggested by the slight change in model fit.

4.1.2 Implication for wetland representation

Comparison of wetlands Paiwalla and Sunnyside

For both Paiwalla wetland and Sunnyside wetland, being similar wetlands and in close proximity, the modelling scenarios performed well enough to allow a comparison.

Sunnyside wetland scenario underestimated PO₄-P considerably. The D (Table 3) is due to the low variability of PO₄-P concentration in the wetland. Further, from early April to the end of the simulation period the PO₄-P trend was simulated very well (Figure 25A). The Paiwalla wetland scenario PO₄-P D (Table 3) showed a somewhat worse fit than the Sunnyside wetland scenario, although the concentrations within Paiwalla wetland scenario are for the most part closer to the monitored (Figure 23A). In the Paiwalla wetland scenario the model fails to mimic the PO₄-P trend as well as it does in the Sunnyside wetland simulation (Figure 23A and Figure 25A). The most
likely reason for under prediction in Sunnyside wetland was discussed in section 4.1.1. The Paiwalla wetland scenario PO₄-P prediction, although with a worse fit than anticipated, did improve considerably with the introduction of river exchange.

The Sunnyside wetland NO₃-N simulation performance was good with a low $D$ (Table 3) and with a trend, displayed in the time-series, showing a very good fit (Figure 25B). The Paiwalla wetland $D$ (Table 3) of NO₃-N simulation, although not poor, is an indication to the potential failings of $D$, if used alone, in assessing a comparative modelling output. This statement is made as, although this is not evidenced in the $D$, the trend or rather the time-series fit (Figure 23B) shows a great improvement with the introduction of river exchange.

The Paiwalla wetland $D$ (Table 3) for phytoplankton improves dramatically and can also be seen in the time-series display (Figure 24C) and provides a strong argument for the validity of WETMOD 2 phytoplankton simulation capacity. The Sunnyside wetland phytoplankton simulation also improves with the introduction of river exchange both as represented by the $D$ (Table 3) and by the visual trend assessment (Figure 26C).

The macrophyte biomass increase within the Paiwalla wetland scenarios is low, with a rapid decline following the initial growth phase (Figure 24A). The cause of the decline is related to the turbidity level within the wetland limiting underwater light penetration. This monitored wetland turbidity does not increase in the Sunnyside wetland scenario until two weeks later, therefore allowing for a longer macrophyte growth phase. The later increase in turbidity in Sunnyside wetland is assumed to be as a result of the higher macrophyte levels within Sunnyside wetland that act both to settle out the turbidity and to reduce sediment re-suspension. The lower exchange volume of Sunnyside wetland (Table 3) is also assumed to be as a result of the macrophyte growth, whereby the water flow through Sunnyside wetland in comparison to Paiwalla wetland is reduced. The significance of the difference in the macrophyte growth phase between the two wetlands is reflected in the phytoplankton time-series. Where, as a consequence of competition for nutrients and light the Sunnyside wetland scenario shows a very small summer phytoplankton growth phase compared with Paiwalla (Figure 26C vs. Figure 24C). Another consequence of the higher macrophyte biomass content within the Sunnyside wetland scenario is the habitat availability assumed to be provided to zooplankton represented by the higher
summer zooplankton biomass compared with the winter biomass (Figure 26B). In contrast in Paiwalla wetland, with low macrophyte biomass, the zooplankton growth (Figure 24B) mimics the phytoplankton growth (Figure 24C) more closely. The macrophyte and zooplankton model output assessments are limited however by the lack of validation data.

**Comparison of wetlands Lock 6 and Pilby**

Lock 6 wetland and Pilby Creek wetland are located geographically close. Prior to the management of Pilby Creek wetland they were both in a similar degraded state. Unfortunately no monitoring of Pilby Creek wetland was undertaken prior to management so no direct comparison can be made at this time of simulations of a particular wetland in a degraded and in a restored state.

For Lock 6 wetland, both the PO₄-P and phytoplankton D (Table 3) increased once river exchange was introduced. However, a visual assessment of the time-series trend (Figure 27A and Figure 28C) showed a marginal improvement in both cases. The improvement in the Lock 6 NO₃-N simulation performance was exceptionally good both visually (Figure 27B) and according to D (Table 3 reducing by a full 20%), supporting the claim that the simulation of Lock 6 was successful. This discrepancy in PO₄-P and phytoplankton results was not seen in the Pilby Creek wetland scenarios, where there was an improvement in both D (Table 3) and the visual assessment (Figure 31A and Figure 32C). Both wetlands showed the assumed expected macrophyte biomass growth trends (Figure 28A and Figure 32A). In the Lock 6 wetland scenario there was a rapid decline from initial macrophyte biomass and in Pilby Creek wetland there was a substantial macrophyte biomass increase post re-wetting followed by an expected winter reduction. The phytoplankton biomass growth, in both wetlands (Figure 28C and Figure 32C), responded appropriately to the level of competition expected in respect to the macrophyte biomass present (Figure 28A and Figure 32A). In the Lock 6 wetland scenario low macrophyte competition caused phytoplankton biomass to reach high levels, only matched by Reedy Creek wetland, which can be viewed as another wetland with high nutrients concentrations (Figure 29) and low macrophyte competition (Figure 30A). The phytoplankton biomass in the Pilby Creek wetland scenario matched the time-series trend expected, with a growth phase both prior to and directly following the macrophyte growth phase.
Zooplankton in the Pilby Creek wetland scenario responded to the shelter availability assumed to be afforded by macrophytes. However, the zooplankton in Pilby Creek wetland (Figure 32B), despite being relatively more abundant when compared to the phytoplankton availability in both wetlands (Figure 32 and Figure 28), were restricted by the low food source of phytoplankton (Figure 32C). Whereas in the Lock 6 wetland scenario there was a large zooplankton biomass increase (Figure 28B) due to the ample nutrient source the phytoplankton biomass (Figure 28C). The ample phytoplankton biomass therefore minimised the otherwise negative impact of the lack of habitat normally provided by macrophytes (Figure 28A).

The good scenario trend results provided by the model in the case of Lock 6 wetland and Pilby Creek wetland confirms the applicability of WETMOD 2 to wetlands in both extreme stable states (turbid and clear). The model can therefore be applied with confidence to category wetlands belonging to either Lock 6 wetland or Pilby Creek wetland (category 3 and 5). This confidence being both placed in the representation of the realistic trend of wetland nutrient concentration as well as in the impact respective external nutrient sources have upon the wetlands. However, as stated for Paiwalla and Sunnyside wetlands the macrophyte and zooplankton model output assessments are limited by the lack of validation data.

**Comparison of wetlands Sunnyside and Reedy Creek**

The main difference between the two wetlands is the data quality and quantity. Reedy Creek wetland has more comprehensive data so is more suitable for modelling purposes. The Reedy Creek wetland simulation succeeds where the Sunnyside simulation struggles. Results from Reedy Creek wetland simulations provide the strongest argument for the validity of WETMOD 2.

For the Reedy Creek wetland scenarios, as can be seen by the $D$ in Table 3 and the wetland time-series data in Figure 29 and Figure 30, there are obvious improvements in the model output both with the introduction of river exchange, as well as the introduction of irrigation drainage nutrient inflow. There were significant improvements in the overall modelling performance at Reedy Creek wetland for NO$_3$-N and PO$_4$-P modelling (Figure 29) as well as considerable improvement in phytoplankton modelling performance (Figure 30C). Visual assessment of Figure 29
and Figure 30 shows the model to simulate the Reedy Creek nutrient and phytoplankton time-series trend satisfactorily. The Reedy Creek wetland macrophyte simulated biomass is low due to the high turbidity and low Secchi depth, the zooplankton therefore mimicking only the growth of its food source the phytoplankton.

WETMOD 2 shows great success with the notable performance in simulating Reedy Creek wetland. The results of Reedy Creek wetland simulations support the argument that the model is capable of simulating wetlands with both river and irrigation drainage as external nutrient sources. Therefore, the reasoning that the poor data quality for Sunnyside wetland affects its simulation performance is justified based on the successful Reedy Creek wetland scenarios.
4.2 Validation based on non-calibrated wetland data

WETMOD 2 has a generic nature; through the use of wetland categories and its simplicity it is applicable to wetlands and timescales other than where it was developed. In order to verify the model applicability at different timescales and wetlands, the model must show itself to be accurate outside of the data range where it was developed. Therefore, to rigorously test the model, it should be fitted to one set of data, while checking for agreement with independent data (Goodall 1972; Tsang 1991; Wood 2001). Extra validation therefore, not only serves the validation of the model for the monitored wetlands, but also supports the argument of the model’s generic applicability. If the model is capable of accurately simulating a separate set of data than used in the calibration, the acceptance of the qualitative simulations for category wetlands where no time-series are available should be strengthened.

For the purpose of rigorous validation, some of the data for Reedy Creek wetland (category 4 wetland) were withheld during the model calibration stage. This extra data stems from the same source project and covers the seven months prior to the data used in the model calibration stage (the data used in the model calibration stage spanned one year, see Box).

The time period chosen, for Reedy Creek wetland data, in the model development stage was due to two significant factors;

1. It was a highly variable year therefore providing the model with complex data and dynamics.
2. It was from the winter period of low growth to the next winter period (so it encompassed an entire growth cycle)

Following the monitoring project that provided data for the modelling of Lock 6 and Pilby Creek wetlands, another project monitored the same wetlands. The data from this second monitoring study, performed by van der Wielen (nd), was kept separate from the data used in the model development. It is therefore possible to validate the developed WETMOD 2 on the data not used in 3 of the 5 category wetlands.

The method used by van der Wielen in assessing the NO$_3$-N concentration was a colorimetric method (Cadmium Reduction Method) (van der Wielen nd). Colorimetric methods require an optically clear sample as the turbidity of a sample can conflict with the colorimetric measurement (APHA et al. 1992). After discussions with van
der Wielen (nd), it was considered likely that the very turbid waters of the River Murray wetlands compromised the monitored NO$_3$-N values. As a consequence the NO$_3$-N measurements in the Pilby Creek and Lock 6 data set cannot be relied upon. In this case the modelled PO$_4$-P compared to the monitored PO$_4$-P gives the best estimation of model validity. The $D$ for the modelled results of the validation data is presented in Table 4 below. The individual results are discussed below.

Table 4: Non calibrated validation of inflow data for 3 wetland categories

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<th>Wetland Category</th>
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<th>Modelled NO$_3$-N $D$</th>
<th>Modelled Phytoplankton $D$</th>
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Category 3: Dead end wetlands with carp presence and no irrigation drainage (Lock 6 wetland)

The simulated time-series for the non calibrated data validation of WETMOD 2 for category 3 wetlands are presented in Figure 34 and Figure 35. The standard error, at each monitoring date, is represented for PO$_4$-P, NO$_3$-N and phytoplankton biomass.

The wetland scenario did not initially perform as well as was expected. The $D$ actually degraded with the introduction of exchange (Table 4). The time-series graph in Figure 34 however, does show an improvement in the modelling trend after the introduction of the river exchange. For this scenario the default exchange volume for Lock 6 wetland was kept at the same level as used during the model development stage.
Regional Scale Modelling of the lower River Murray wetlands

However, during simulations using monitored data from three different locations within Lock 6 (available in van der Wielen’s data), it was discovered that the impact of the river exchange diminishes as the distance of the monitoring location from the river channel increases. It was therefore assumed to be reasonable to examine a different exchange volume as the monitoring site locations within the wetland differed. A reduced exchange rate at 0.05% of the river daily flow volume showed an improved $D$ (Table 4) and a well fitting time-series as can be seen in Figure 34A.

The Lock 6 $D$ improvement for PO$_4$-P is noteworthy despite the model not being calibrated for this data. Therefore, the model was considered valid for the PO$_4$-P scenario within Lock 6 wetland. However, the NO$_3$-N and phytoplankton $D$ results were poor. As discussed previously the NO$_3$-N monitored data was to be considered with scepticism and cannot be relied upon. Looking at the result in Figure 34B one can however see a slight improvement in NO$_3$-N estimation during October 1998. Based on this scenario and due to the unreliable nature of the monitored NO$_3$-N the model can, for NO$_3$-N wetland concentration simulation, neither be considered valid nor invalid.

In Figure 35C the phytoplankton shows an improvement, despite the $D$ results, during the October 1998 to January 1999 modelled period. For this scenario the phytoplankton modelling results show a significant overestimation for the modelled period. However, due to the performance of the model with regard to phytoplankton, both during model development and in the following validation scenarios at other wetlands described below, the model should not yet be considered invalid. Future model development should focus on addressing the phytoplankton discrepancy, which may be as simple as the monitoring methodology, the conversion of chlorophyll-a to phytoplankton or addressing the sediment impact on wetland water nutrient load. In the mean time phytoplankton volume estimation from modelling scenarios should be reviewed carefully before management decisions are made based on the model results.
Figure 34: Validation of simulation results for Lock 6 wetland PO$_4$-P and NO$_3$-N, using non-calibrated wetland data.
Figure 35: Validation of simulation results for Lock 6 wetland Macrophyte Biomass, Zooplankton and Phytoplankton biomass, using non-calibrated wetland data.
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Category 4: Dead end wetlands with carp presence and irrigation drainage (Reedy Creek wetland)

Reedy Creek wetland data provided the best data for model development. The data from Reedy Creek wetland withheld during model development also provided the most comprehensive and reliable data for extensive model validation based on non-calibrated driving variables. Figure 36 and Figure 37 display the simulated output for the non-calibrated data validation of category 4 wetlands. The standard error at each monitoring data based on three separate measurements is included where appropriate.

Evaluating these scenario results for PO₄-P simulation there is a significant improvement in the D (Table 4) where both river exchange and irrigation drainage are considered. Although the model was not calibrated for this time-series the results show a satisfactory resemblance to the monitored PO₄-P as seen in Figure 36A.

The NO₃-N D (Table 4) during this time-series actually shows a better fit to the monitored data than the original calibrated data time-series. This can be attributed to the high variability in the development data series, which were partly chosen as a consequence of this variability. The time-series seasonality and fit can be seen in Figure 36B. The NO₃-N modelling result is the only NO₃-N data available with which to verify the model outside of the data used in model development. The notable improvement in the improvement of the D and the good fit shown in Figure 36B provide a strong case for the validity of WETMOD 2 with regard to NO₃-N simulation. The phytoplankton D shows a significant improvement (Table 4), although the seasonality is somewhat exaggerated as seen in Figure 37C.

The performance of WETMOD 2 for Reedy Creek wetland, with both data sets calibrated and non-calibrated, demonstrates the performance that can be obtained when adequate data is available. The model performance for Reedy Creek wetland is the strongest argument in the favour of model validity. Therefore, the shortcoming of the model in previous instances can to a large degree be attributed to data quality.

The Reedy Creek wetland results show that the availability of adequate quality data improves the performance of the model. However, it is a generic modelling tool where simple data sets can be used giving reasonable trends, thereby assisting potential management decisions. The lack of quality data should in this case not necessarily hinder scenario analysis however; the decision maker must understand that the quality
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of the modelling output is very dependent on the quality of the data used as driving variables.

Figure 36: Validation of simulation results for Reedy Creek wetland PO$_4$-P and NO$_3$-N, using non-calibrated wetland data
Figure 37: Validation of simulation results for Reedy Creek wetland Macrophyte Biomass, Zooplankton and Phytoplankton biomass, using non-calibrated wetland data
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Category 5: Dead end wetlands managed through implementation of dry periods with carp restriction and no irrigation drainage (Pilby Creek wetland)

The non-calibrated driving variable validation of WETMOD 2 based on category 5 wetlands is displayed in Figure 38 and Figure 39. The standard error of the monitored data set is included where available.

Pilby Creek wetland PO$_4$-P in this case has a noteworthy improvement in $D$. Nevertheless, the improvement is best judged from the time-series in Figure 38A where the concentrations in the early modelling period were very close to the monitored concentrations. The September to February performance is exaggerated but is at least showing a similar trend to the monitored data. The main discrepancy in the PO$_4$-P modelling is the lack of the late November early December peak.

Pilby Creek wetland validation data stems from the same source as the Lock 6 validation data. The NO$_3$-N monitoring results are therefore, as in the case of Lock 6 wetland, to be considered suspect and therefore no model validation will be made based on NO$_3$-N model output for this data.

The phytoplankton biomass growth is greater than expected (see Figure 39C) particularly the initial peak growth phase, which is due to the lack of macrophyte competition. However, the model scenario does retain a low phytoplankton biomass load as is expected of Pilby Creek wetland given the simulated macrophyte biomass.

The macrophyte biomass growth does show an increase; followed by a winter decrease (see Figure 39A). The zooplankton biomass pattern as can be seen in Figure 39B follows both its food source pattern, i.e. phytoplankton, and assumed shelter availability afforded by the macrophytes. The zooplankton does in this instance have a more complex growth pattern than the phytoplankton due to the high shelter availability provided by the macrophytes.

From the modelling results in this case as well as the two above, the model has shown itself capable of simulating wetlands for which it has been calibrated, but with non-calibrated data sets. Each of these wetlands is either in a different stable state, i.e. clear vs. turbid, or has added external influences (Reedy Creek wetland irrigation drainage inflow). This supports the argument that the model is generically applicable to similar wetlands. Where data for these similar wetlands is non existent, the accuracy WETMOD 2 trend development allows the use of “exemplar” data obtained
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from the calibration wetlands, and consequently the development of qualitative scenarios and hypothetical quantitative outcomes. The application of WETMOD 2 to category wetlands in such a manner is explored in chapter 6.

Figure 38: Validation of simulation results for Pilby Creek wetland PO$_4$-P and NO$_3$-N, using non-calibrated wetland data
Figure 39: Validation of simulation results for Pilby Creek wetland Macrophyte Biomass, Zooplankton and Phytoplankton biomass, using non-calibrated wetland data
4.3 Evaluating model performance

4.3.1 Generic nature and structural restrictions of model

When wetland scenario results are evaluated and compared, WETMOD 2 performs satisfactorily and as expected, even for wetlands with extreme conditions of turbid and clear. The quantitative results may not reflect the accuracy expected of a dedicated wetland model. However, as discussed in the introduction, the limiting model structure, the lack of data availability and the models generic nature does not allow for WETMOD 2 to be fitted to one wetland in particular. This allows the model to be applied to a larger range of wetlands, even where verification may not be possible, with confidence in the simulation results qualitative trend. Therefore, in developing WETMOD 2, a compromise on quantitative accuracy was made in order to be able to compare the relative conditions of wetlands, including impacts external influences may have on the wetlands and/or wetlands with minimal or no time-series data.

The data quality available for a given wetland has a direct impact on the accuracy of WETMOD 2 to simulate internal nutrient dynamics, as seen for Sunnyside and Reedy Creek wetlands. The potential to simulate management scenarios is directly linked with model performance. Consequently, due to the lack of data quantity and particularly quality for Sunnyside wetland management simulations for Sunnyside wetland and therefore category 2 wetlands were not feasible. However, for both Lock 6 wetland and Reedy Creek wetland, scenarios of potential management strategies were possible and are described and discussed in chapter 5. Using both Lock 6 and Reedy Creek wetland data “exemplar” category wetlands could therefore also be simulated, this is described and discussed in chapter 6.

In WETMOD 2 macrophyte growth is controlled to a large extent by light availability, where the growth of macrophytes increases with a decrease in turbidity and therefore increase in Secchi depth. This relates back to Secchi depth representation of underwater light availability. The equation in the model assumes that at increased Secchi depth there will be an increase in underwater light availability and therefore in the macrophyte growth. A limitation discovered during model validation pertains to the equation used. The equation shows a logarithmic growth curve with increasing Secchi depth, which in itself is not regarded as inaccurate. However, the limitation of
this equation is its lack of consideration of the maximal depth of the wetland, i.e. there is no correlation of the equation to the water depth and therefore the maximal light penetration possible. Therefore, a shallow wetland with a depth less than 0.6m is not simulated as having substantial macrophyte growth despite the underwater light being fully available to macrophyte growth, represented by the Secchi depth effectively penetrating to the wetland bottom. The wetlands, which were monitored and provide the wetland time-series driving variables, are all of a depth where this restriction is not of significant concern; and where appropriate the macrophyte growth is calibrated to expected trends. This limitation impacts on the application of the model to very shallow wetlands. As there currently is no model calibration data available or sufficient data available as driving variables for very shallow wetlands this limitation is currently not an issue. Future development of WETMOD should however take this limitation into account and replace the current Secchi depth equation with a more appropriate one.

### 4.3.2 Relevance of project objectives

The principal objective calls for the improvement of the resolution of spatial influences acting upon wetlands. That is, to develop or adopt a generic wetland process model to local external influences acting on a wetland. The purpose of the objective is to improve the understanding of the respective spatial influences acting upon a wetland, such as morphology and external nutrient sources, and how management can impact on the nutrient retention capacity of wetlands at each spatial location. The spatial differences considered in WETMOD 2 are any significant external sources acting upon wetlands, including:

- river nutrient load,
- the presence or absence of agricultural drainage (irrigation drainage) with its associated nutrient load contribution, and
- in isolated cases the impact of precipitation on irrigation drainage nutrient contribution. (South Australia is a very dry state with minimal precipitation. Most wetlands in the South Australian stretch of the River Murray do not have independent catchment areas. Precipitation is therefore in most cases not relevant).
As the principal focus of WETMOD 2 development was the spatial context of the wetlands, i.e. the individual external influences, it is important to discuss whether the model behaves logically based on the anticipated impact of external influences as well as the comparative differences of two wetlands. The validation of the model and the comparison of wetlands, discussed above, have shown the successful improvement of the model simulation output following the introduction of external influences. The simulation outputs therefore enable the study of the local and assumed external impact on wetland fulfilling this principal project objective.
4.4 Chapter summary and Implication for the first hypothesis

Based on the validation results presented above WETMOD 2 is considered capable of simulating wetland seasonal nutrient flux for individual wetlands that are affected by varying external influences. Further, WETMOD 2 is considered valid, based on the wetlands that were used in model development. To improve this confidence further model validation using separate wetland data was performed and is described in section 4.2. From this the model can be applied to category wetlands with reasonable confidence placed in the output trend.

The first hypothesis is that “a simplified generic wetland model can be used to realistically simulate multiple and different wetlands qualitatively”. To address this hypothesis the results of the different wetlands scenarios, developed as part of the model calibration and model validation, were reviewed as to their realistic representation of expected wetland nutrient and biomass growth trends. These wetlands are listed in Table 5.

<table>
<thead>
<tr>
<th>Category</th>
<th>Wetland</th>
<th>Simulated realistically</th>
</tr>
</thead>
<tbody>
<tr>
<td>Category 1 wetland</td>
<td>Paiwalla</td>
<td>YES</td>
</tr>
<tr>
<td>Category 2 wetland</td>
<td>Sunnyside</td>
<td>Limited</td>
</tr>
<tr>
<td>Category 3 wetland</td>
<td>Lock 6</td>
<td>YES</td>
</tr>
<tr>
<td>Category 4 wetland</td>
<td>Reedy Creek</td>
<td>YES</td>
</tr>
<tr>
<td>Category 5 wetland</td>
<td>Pilby Creek</td>
<td>YES</td>
</tr>
<tr>
<td>Category 3 wetland</td>
<td>(non calibrated data)</td>
<td>YES</td>
</tr>
<tr>
<td>Category 4 wetland</td>
<td>(non calibrated data)</td>
<td>YES</td>
</tr>
<tr>
<td>Category 5 wetland</td>
<td>(non calibrated data)</td>
<td>YES</td>
</tr>
</tbody>
</table>

This shows that the model is capable of simulating different wetlands, for which adequate data is available, realistically although not to the accuracy of individually tailored wetland models. This argument is strengthened by the results of the model validations based on non-calibrated wetland data.