Methods to Assess Environmental Flow and Groundwater Management Scenarios for Floodplain Tree Health in the Lower River Murray

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PhD Thesis

Methods to Assess Environmental Flow and Groundwater Management Scenarios for Floodplain Tree Health in the Lower River Murray

Submitted by

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ABSTRACT

Riparian environments have degraded world-wide as a consequence of human development and climatic change. The native floodplain tree communities of semi-arid river systems are under stress from reduced flooding frequencies as a consequence of water extractions, river regulation and climate change. In regions with saline aquifers, river regulation and land management have also caused soil salinisation, further impacting on floodplain tree health.

The lower River Murray in south-eastern Australia is a major ecological asset considered as an area of international significance. The dominant floodplain vegetation is suffering severe decline in health, with approximately 80% of floodplain trees reported as being in poor condition or dead. A reduction in water availability from reduced flooding and soil salinisation, has been identified as the primary cause. This has resulted from large irrigation extractions across the Murray-Darling Basin and elevated saline groundwater levels due to river regulation and land clearance.

Management of these ecosystems needs to address both surface and groundwater changes. Increasing flooding regimes from environmental flow management and lowering of groundwater in regions of shallow saline aquifers are the most common scenarios adopted world-wide. Traditionally the assessment of management options for floodplain habitats has focussed on changes in river flow with no consideration given to surface water and groundwater interactions. In addition groundwater has been treated as a single homogenous unit. Wide floodplains have high spatial variability of habitats due to historic meandering anabranch creek systems that cause changing elevations and soil types. This in turn creates a highly variable pattern of surface and groundwater interactions. This thesis investigates the major causes of floodplain tree decline and develops methods for predicting the spatial impacts on floodplain tree health from a range of management scenarios.

Surface and groundwater changes are often highly inter-connected but are usually considered separately at regional scales because of the complexity of management and modelling of surface and groundwater interactions over large areas. This thesis addresses the surface and groundwater changes at the regional scale of the lower River Murray. A floodplain inundation model for the River Murray (RiM-FIM) is developed to predict the extent of flooding at various magnitudes of flow and river regulation and a 'drought index' was used to indicate the risk to floodplain tree health of changing flow regimes. A floodplain impacts model (FIP) was applied spatially to predict groundwater discharge onto the floodplain and model vegetation risk.

At the floodplain scale, surface and groundwater need to be integrated to assess detailed management scenarios. This thesis develops a method for assessing soil water availability from surface and groundwater interactions using a spatial and temporal model of salt accumulation and recharge (WINDS). This model is then used to predict floodplain tree health.

The thesis contributes to the science of floodplain processes and develops a number of innovative modelling techniques for predicting the spatial variability of floodplain tree impacts, improving on traditional broad assessment methods. The tools are applicable to other saline semi-arid rivers and are useful for environmental flow and groundwater management decision making.

DECLARATION

I certify that this work contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text. In addition, I certify that no part of this work will, in the future, be used in a submission for any other degree or diploma in any university or other tertiary institution without the prior approval of the University of Adelaide and where applicable, any partner institution responsible for the joint-award of this degree.

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Ian Clifford Overton B.Sc. (Hons.)

Date:

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ACRONYMS

AHD	Australian Height Datum, standard measurement for heights in metres above sea level.
BigMOD	MDBA River Murray Flow Model used to predict River Murray flows for natural (pre-development), current and future conditions
CSIRO	Commonwealth Scientific and Industrial Research Organisation, which supported much of this research
FIP	Floodplain Impacts Model, a groundwater model to predict impacts on floodplain vegetation, groundwater seepage and salt loads, further developed further by this research
GIS	Geographic Information Systems
LiDAR	Light Detection and Ranging system for collecting elevation data
MDB	Murray-Darling Basin
MDBA	Australian Government Murray-Darling Basin Authority
MDBC	Australian Government Murray-Darling Basin Commission
MODFLOW	USGS Modular Three-Dimensional Groundwater Flow Model
RiM-FIM	River Murray Floodplain Inundation Model, a predictive model of flood extent developed by this research
WAVES	Water Vegetation and Salt Model, developed by the CSIRO as a model of vegetation growth incorporating surface and groundwater influences
WINDS	Weighted Index of Salinisation Model, developed by the CSIRO as a model of soil water availability to infer vegetation health, developed further by this research

1 INTRODUCTION

Aquatic ecosystems and freshwater biodiversity constitute a valuable natural resource, in ecological, scientific, cultural and economic terms and have considerable ecosystem service values. These ecosystems are characterised by high spatial and temporal diversity due to their complex hydrological and geomorphological drivers. Aquatic ecosystems have degraded in condition and extent across the world as a result of river regulation, water extraction and land use changes. Climate change and population predictions suggest there will be increasing pressures on water resources and riverine environments.

Many catchments are now seen to be over-allocated, in terms of their water resources, and management actions are being developed to re-address the balance between water for consumptive use and water for the environment. In arid and semi-arid regions the pressures of water use are increased and this can be further complicated by saline groundwater. Management options include releasing flows for the environment and managing groundwater levels.

The objective of this thesis was to develop methods to assess environmental flow and groundwater management scenarios for floodplain tree health in saline semi-arid rivers by considering the spatial variability of floodplain habitats and their surface and groundwater interactions. The thesis considers both regional and floodplain scales and the different approaches needed to assess management options targeted at these scales. The terms ‘floodplain’ and ‘riparian’ are used here synonymously; reference is made to all trees, adjacent to and away from water bodies, as ‘floodplain trees’. The term ‘environmental flow management scenarios’ includes all surface water management scenarios that release river flows, retain water to increase inundation or pump water to flood areas, for the purposes of environmental benefit. This chapter identifies the nature of the problem and outlines the objectives and the approach of the thesis.

1.1 NATURE OF THE PROBLEM

1.1.1 Floodplain Ecosystems

Fresh water makes up only 0.01% of the World's water and freshwater ecosystems occupy approximately 0.8% of the Earth's surface, yet they support at least 6% of all described species (Dudgeon *et al.*, 2006; Ward, 1998). Wetlands deliver a wide range of ecosystem services that contribute to human well-being, such as fish and fibre, water supply, water purification, climate regulation, flood regulation, coastal protection, recreational opportunities, and, increasingly, tourism.

In arid and semi-arid regions, aquatic ecosystems are characterised by a scarcity of available water and highly variable river flows (Smith *et al.*, 1998; Patton, 1998; Puckridge *et al.*, 1998; Walker *et al.*, 1995). As evaporation exceeds rainfall in these environments the aquatic ecosystems are highly dependent on the surface flows they receive. This is exemplified by the floodplain environments that rely on periodic flood events. The diversity and functioning of riparian plant communities on floodplains is mainly controlled by the flow regime (Poff *et al.*, 1997), which provides vital water in arid and semi-arid regions, as well as, generating physical disturbance affecting its temporal and spatial dynamics (Shafroth *et al.*, 2002).

Groundwater under the floodplain can also provide a vital water resource in arid and semi-arid environments, especially during low rainfall or drought periods. Surface water and groundwater are linked as surface water recharges groundwater and laterally through the creek banks during flooding. The high spatial variability of wide floodplain environments creates a complex pattern of floodplain habitats and consequently a mosaic of surface water and groundwater impacts on floodplain tree health. Wide floodplains have a history of traversing anabranch creek systems and meandering river channels that leave a legacy of complex soil types and elevations.

The Murray-Darling Basin (MDB) is a major ecological asset to Australia and contains 16 wetlands listed under the Ramsar Convention as areas of international significance (Ramsar, 1971). The MDB's floodplain wetlands are sites of extraordinary biological diversity with abundant and diverse populations of waterbirds, native fish, invertebrate species and aquatic and riparian vegetation (Kingsford, 2000).

In the lower River Murray region of the MDB there are many wide floodplains, including two listed as wetlands of international importance (Ramsar, 1971). The lower River Murray was used as the regional scale case study for this research. For the floodplain scale case study, the largest of these, the Chowilla floodplain, was used (Figure 2.2).

1.1.2 Water Resource Development

To develop arid and semi-arid areas, rivers have been regulated to secure water supply, produce power, control floods and provide for transportation routes (Nilsson *et al.*, 2005; Rosenberg *et al.*, 2005). In many cases this regulation has had profound hydrological and ecological side-effects on the rivers themselves and their adjacent floodplains (Poff *et al.*, 1997; Jolly, 1996; Rood *et al.*, 1995; Hughes, 1988).

The MDB covers an area one seventh of Australia but supports 70% of Australia's irrigated crops and pastures. The MDB is over one million square kilometres, with 107 large storages that can hold 25 million ML, an amount greater than the mean annual run-off of approximately 24 million ML.

The River Murray discharges the flow from the catchment into the sea in high flow periods, ceasing to flow during droughts. The River Murray provides a major water source for towns in the MDB, and more than half of the water supply for the city of Adelaide. During the drought of 1982-83, this reached 90%. The river has been regulated since 1920 by a series of ten weirs in the lower River Murray.

Throughout the world flows in many arid and semi-arid rivers have been declining for decades. The lower River Murray in South Australia is at the bottom end of the system and particularly experiences the decline in river flows as a consequence of extraction and regulation across the MDB (Walker and Thoms, 1993). The frequency and duration of floods in the lower River Murray have declined significantly since river regulation, with a reduction by a factor of three in the return period of medium sized floods (Ohlmeyer, 1991). During the drought of 2002-2010, the lower River Murray received no floods.

With increasing global temperatures, arid and semi-arid rivers are receiving reduced volumes of water commensurate with drought conditions (CSIRO, 2008). With escalating populations the water sources are being utilised increasingly through extraction of surface water and groundwater for domestic and production use. Regulation of rivers through weirs to increase the security of abstracting desired volumes of water, raises river levels and this in turn raises groundwater levels. The weirs in the lower River Murray raise the water level in the main channel up to three metres higher than average pre-development conditions and consequently raise groundwater levels under the floodplain. The regional groundwater is highly saline as a legacy of an inland sea. Land clearing across the MDB has also contributed to the rising groundwater as well as increased groundwater recharge from irrigation practices close to the floodplains. The combination of reduced flooding and shallower saline groundwater leads to soil salinisation and decline of the riparian vegetation (Jolly, 1996).

1.1.3 Declining Environmental Health

World-wide floodplain ecosystems are under pressure from river regulation, water resource development and climate change (Arthington *et al.*, 2010; Dudgeon *et al.*, 2006; Tockner and Stanford, 2002; Sparks, 1995; Hollis, 1990). The degradation and loss of wetlands is more rapid than in other ecosystems (Millennium Ecosystem Assessment, 2005).

In particular the native riparian vegetation communities of arid and semi-arid river systems are being threatened (Rood *et al.* 2005; Hauer and Lorang, 2004; Lemly *et al.*, 2000; Walker, 1985). Impacts of flow alteration include changes in both hydrology, causing changes in water availability, and in geomorphology, causing changes in habitat structure (Bunn and Arthington, 2002; Bendix and Hupp, 2000). In regions were the riparian environments rely on access to groundwater, falling levels from extraction of groundwater and reduced infiltration can cause even further vegetation decline or a change of ecosystem type (Horton *et al.*, 2001; Sophocleous, 2000a; Patten, 1998; Cleverly *et al.*, 1997; Stromberg *et al.*, 1996). When groundwater aquifers are saline, as is the case in much of Australia, Spain, Africa and parts of North America, shallow groundwater can lead to salinisation of the floodplain soils, increasing the stress to riparian vegetation (Lamontagne *et al.*, 2005; Jolly, 1996).

The MDB's floodplains are experiencing severe decline in condition and extent (MDBA, 2010; Kingsford, 2000; Walker, 1985). Floodplain tree health along the lower River Murray was mapped in 1990 with an estimated 82% of the floodplain trees in good health (Margules *et al.*, 1990). This decreased to an estimated 65% of the trees in good health in 2003 (DEH, 2006). A major drought period occurred in the MDB from 2002 to 2010, with estimates of only 20% of the trees in good health in 2008 (Cunningham *et al.*, 2008).

With the death of the floodplain trees, the ecosystem is likely to change as trees provide habitat, soil stabilisation, food and micro-climate variation. As trees become deprived of water their over-all health declines, which results in reduced seed production, reduced recruitment, reduced resilience to temperatures and droughts, and reduced resistance to parasites like mistletoe (Miller *et al.*, 2003). The poor condition of trees also provides reduced ecosystem services such as aesthetic, cultural and forestry uses.

1.1.4 Management Scenarios

Australia is not unique in the degradation of riparian areas. The case for management of riparian areas in arid and semi-arid regions can be built on evidence of ecosystem decline across the world (Hughes, 1988, South Africa; Walker *et al.*, 1995; Australia; Stromberg *et al.*, 1996, South-West United States; Salinas *et al.*, 2000, Spain; Glenn *et al.*, 2001, Mexico).

Management of the River Murray in south-eastern Australia had largely focussed on mitigating large floods to protect infrastructure, while maintaining storages for regular water supply to irrigators. Concerns over river health in the late 1990s focussed attention on environmental flow strategies that considered flow release and management to provide environmental benefits to the floodplain, wetlands and in-stream water quality. 'Environmental Flows', where water is deliberately released for environmental benefits, is accepted as being good riverine and environmental practice, and is being adopted world-wide (Dyson *et al.*, 2003). Strategies have included the cap on water extraction in 1995 (MDBA, 2010) and the Living Murray program to restore 500GL per annum of water to the environment in 2002 (MDBC, 2003).

Within the MDB the amount of water extracted for irrigation, industrial and twon water supply is generally considered to be too high with the remaining water for the environment too little to sustain a healthy aquatic environment. To address this, and other surface water and groundwater resource issues in Australia, the Australian Government developed the Intergovernmental Agreement on a National Water Initiative (NWI) in 2004. The NWI provides the legal impetus for addressing sustainability in water resource management. The Water Act of the Australian Government (Australian Government, 2007) shifted control of water resources in the MDB from individual states to the Commonwealth Government so that the MDB could be managed as a single resource. Part of the Water Act (Australian Government, 2007) established the Murray-Darling Basin Authority (MDBA) and undertook a large water buy-back program to obtain water for the environment from willing irrigators. The MDBA was tasked with developing a Basin Plan that set new sustainable diversion limits from the surface water and groundwater resources in the MDB in an equitable way. Part of the development of this Basin Plan is to determine the environmental water requirements. An environmental watering plan would then be developed to assist in the operation of delivering environmental water to the assets across the MDB.

The Basin Plan is focusing on 18 indicator key ecosystem assets and key ecosystem functions (MDBA, 2010). All of the indicator key ecosystem assets are wide floodplain areas, as these are large water users and significant habitats. The indicator key ecosystem asset in the lower River Murray is the Chowilla floodplain. Several ecosystem functions have also been used in determining the environmental water requirements, and therefore the sustainable diversion limits, one of the main ones is river and floodplain connectivity driven by overbank floods. Environmental flow and groundwater management strategies are being developed, and policies introduced, to control the impact of irrigation and river regulation on the floodplain.

The regulation of the flow of water into the lower River Murray can be manipulated to some degree by the management of the weirs and water storages, such as Lake Victoria and Menindee Lakes, and the raising and lowering of weirs. At a local scale the regulation of the flow of water through a floodplain can be manipulated by local flow control structures. How much water is required, for how long and how often, for the effective management of floodplain and in-stream environments, needs to be determined. This is particularly relevant during drought periods when artificial flooding is undertaken by pumping water into wetlands or over floodplains.

Naturally saline aquifers underlie many arid parts of the world and loss of arable land is a major concern in Australia. The lower River Murray floodplain is affected by two salinity problems. Firstly, higher salt loads enter the river as a result of groundwater inflow from adjacent regional aquifers. The naturally highly saline groundwater enters the river from a groundwater system which is hydraulically connected to the streams of the floodplain. River salt loads periodically exceed world health guidelines, creating problems for irrigation infrastructure and water quality issues for the environment.

The Salinity and Drainage Strategy for the MDB (MDBMC, 2000) aims to prevent the salinity of the River Murray from increasing. To do this, expected increases in salinity are offset by the development of salt interception schemes. These are usually in the form of direct groundwater pumping, which can lower the water table reducing groundwater inflow into the river. However, little is known about the potential environmental benefits to the floodplain as a consequence of such pumping.

The second major threat regarding salinity is the accumulation of salt in floodplain soils (Jolly, 1996). The regulation of river flows using locks and weirs and the establishment of evaporation basins on the floodplain has resulted in higher groundwater levels. Groundwater levels have also increased by irrigation drainage creating artificially high groundwater mounds. Higher groundwater levels cause an increase in the rate of water rising to the surface from evapotranspiration. As water is used by plants or evaporated it leaves behind the salt which then accumulates in the upper soil profile. Prior to river regulation these salts would have been leached from the soil by floods and rainfall.

It is anticipated that in order to restore good health to the riparian vegetation, a lowering of the water table to reduce the upward movement of salt through the soil, and a return of flooding regimes closer to their original state would be required. The assessment of ameliorative management strategies requires analysis of the current environment and prediction of the potential consequences of changes. Management plans must prepare for extreme conditions and climate change predictions suggest an even drier and more variable rainfall in the future.

Policy is complex at regional scales, so surface flow and groundwater management usually operates separately, sometimes in conflict. The Murray-Darling Basin Commission's (MDBC) Basin Salinity Management Plan (MDBMC, 2000) sets targets for the management of river salt loads. The MDBC's Living Murray (MDBC, 2003) sets targets for floodplain health. As salt is stored in the floodplain by reducing floodplain inundation, river salt loads are reduced, but so is floodplain health.

1.1.5 Assessing Management Scenarios

Management scenarios need to be assessed in relation to these complex environmental processes. There is a need for decision support tools that can predict potential outcomes from changed conditions and integrate surface water and groundwater at regional and local scales. Traditional methods of assessing impacts of changing river flows from management scenarios and climate change have focussed on assessment of changes in the river flow history (Kingsford, 2000; Lake and Marchant, 1990; Walker, 1985). The link between a river and its floodplain is critical for riverine and floodplain wetland health (Junk *et al.*, 1989). All components of river flow, including flood peaks, flood frequencies, durations, seasonality, overbank flows, freshes, low flows and cease to flow, are now considered critical components of the natural flow regime (Poff *et al.*, 2010; Arthington *et al.*, 2006).

Setting of environmental flow strategies has usually focussed on 'minimum flows' to maintain basic river processes (Dyson *et al.*, 2003). This approach does not consider all components of the river flow such high flows that trigger fish spawning or peak floods that provide dispersal and migration events. Understanding of the ecological requirements for flow regimes is usually developed by studying the preferred habitat of single species and combining what little is known with expert opinion to set environmental flow strategies (Acreman, 2005). A major modelling exercise (MDBC, 2003) using the Murray Flow Assessment Tool (MFAT) model (Young *et al.*, 2003), linked ecological responses along the River Murray to changing flow regimes. However, no quantitative or spatial tool for predicting the impact on the environment from flooding strategies is available. Although methods of comparing current river flows to natural regimes or flow regimes at reference sites are useful for in-channel assessments of flow requirements, they do not consider the complex patterns of floodplain habitat. Methods that consider the spatial variability of floodplains are required to achieve sufficient accuracy for assessing detailed floodplain management scenarios that alter creek levels, flood extents and groundwater depths.

To assess the impacts of changing groundwater conditions on floodplains, most researchers have applied a methodology of modelling the floodplain as a single homogenous region (Stanford, 1998) or using complex groundwater models that are not able to predict outcomes from temporal changes in flooding (Yan *et al.*, 2005). The coupling of surface water and groundwater models improves the ability to assess management scenarios (Bauer *et al.*, 2006). Threshold targets for groundwater depth, or limits on extraction, are used to define groundwater management rules, although managers are now questioning the concept of sustainable groundwater management as researchers better understand the dynamic nature of aquifers (Sophocleous, 2000b).

At the regional scale of the lower River Murray it is necessary to treat surface water and groundwater management scenarios separately due to the complexity of interactions at that scale. As current policy and management separate these, non integrated tools are appropriate. At floodplain scales the management of the floodplain is at much finer spatial resolutions and the interactions between surface water and groundwater are more easily identified. Management of these two water resources must be integrated, which therefore requires an understanding of these interactions, and ultimately a single integrated model of both processes to predict changes to floodplain tree health.

Vegetation provides the habitat for other species so it is appropriate to use vegetation as a surrogate for ecosystem condition. Riparian vegetation has an important role in aiding bank stability and providing habitat for fauna as well as improving the recreational and amenity value of the floodplain (Tabacchi *et al.*, 1998). In addition, floodplain vegetation plays an important role in many ecosystem services including providing nutrient exchange in riverine environments and water filtration in wetlands. In the lower River Murray the riparian vegetation community is dominated by the overstorey tree species which include the river red gum (*Eucalyptus camaldulensis*) and black box (*Eucalyptus largiflorens*). More is known about the trees than understorey species and tree health is easily assessed rapidly on the ground or by remote sensing. Tree health is also less complex to determine than animal population status, so this study used tree health as a surrogate for ecosystem condition. In this study tree health was assessed based on vigour of adult trees that can be measured visually in the field or from image analysis of remote sensing data. The condition of adult trees is a good indicator of the health of the tree population although impacts could manifest themselves on recruitment stages.

1.2 OBJECTIVES

The overall aim of the thesis is to:

Develop spatial methods to assess environmental flow and groundwater management scenarios for floodplain tree health in saline semi-arid rivers by considering surface water and groundwater interactions.

At the regional scale, it is feasible to consider the management scenarios affecting surface water and groundwater changes separately because the complexity of local surface and groundwater interactions would make regional scale modelling extremely complex. It is also adequate to consider floodplain tree health risk rather than individual tree health. At the floodplain scale individual tree health is important. The interaction between tree health, salinisation, surface water and groundwater is so complex that these factors need to be considered together. Therefore, the overall aim of the thesis can be broken into three main objectives:

- To develop a spatial methodology to assess regional scale surface water management strategies for floodplain tree health risk on saline semi-arid floodplains (Chapter 4);
- To develop a spatial methodology to assess regional scale groundwater management strategies for floodplain tree health risk on saline semi-arid floodplains (Chapter 4); and
- To develop a spatial methodology to assess floodplain scale surface water and groundwater management scenarios for floodplain tree health on saline semi-arid floodplains by including surface water and groundwater interactions (Chapter 5).

To address these objectives the lower River Murray in South Australia is used for assessing regional scale management scenarios and the Chowilla floodplain, at the northern most part of the lower River Murray, for floodplain scale management scenarios. The management tools developed will be applicable to other arid and semi-arid floodplain environments in Australia and around the world.

To achieve the three main objectives a number of sub-objectives were defined including:

- To characterise the major environmental conditions of the lower River Murray in South Australia and the Chowilla floodplain (Chapter 2);
- Determine the processes that affect floodplain tree health, how tree health has been modelled previously and how management scenarios have been assessed (Chapter 3);
- Identify the current tree health on the lower River Murray floodplain to validate predictive models (Chapter 4);
- Develop a regional scale method to identify the extent of floodplain inundation from surface water management scenarios, including increased flows and weir manipulation in the lower River Murray. Using this model of flood extent, develop a model of floodplain tree health risk to define the impacts of flow management scenarios (Chapter 4);
- Develop a regional scale method to identify the effect of groundwater movement in the floodplain and the impact of groundwater management scenarios in the lower River Murray. Using this model of groundwater movement, develop a predictive model of floodplain tree health risk to define the impacts of groundwater management scenarios (Chapter 4);
- Produce maps of surface water and groundwater impacts from management scenarios and potential groundwater recharge as determined from soil hydraulic properties for the Chowilla floodplain in order to predict soil salinisation rates, as well as the benefits of flooding on soil water availability (Chapter 5);

- Develop a predictive spatial model of floodplain tree health for a semi-arid floodplain from the interaction of surface water and groundwater conditions using the Chowilla floodplain as a case study (Chapter 5); and

- Test this model by determining the impact of groundwater and surface water management scenarios and climatic change on floodplain tree health on the Chowilla floodplain (Chapter 5).

1.3 THESIS OUTLINE

The outline of the thesis is presented in Figure 1.1 and described below. The research covers two spatial scales. Firstly the ***regional scale***, which describes an area of floodplain along a river length of approximately 100 kilometres. In the case of this thesis the regional scale is represented by the lower River Murray in South Australia. Secondly the ***floodplain scale***, which describes an area that can be managed locally and is the floodplain along a river length of approximately 10 kilometres. This thesis uses the Chowilla floodplain as an area of focus for the floodplain scale modelling.

This Chapter provides an introduction to the study, identifies the research problem and outlines the structure of the thesis. Chapter 2 describes the regional scale environment of the lower River Murray as the study region and the floodplain scale environment of the Chowilla floodplain. Chapter 3 is a literature review of the major factors impacting on floodplain tree health, examining the requirements for modelling, and previous methods adopted to assess environmental flow and groundwater management scenarios.

Chapter 4 presents the regional scale methods developed to assess environmental flow and groundwater management scenarios. This chapter describes the methods used to map the vegetation communities and health at the regional scale. The chapter presents the development of a Floodplain Inundation Model (FIM) which is then used to predict the risk to floodplain tree health from flow management scenarios. The chapter then develops a floodplain risk model from groundwater impacts in the lower River Murray, the FIP, which is then used to predict salinisation in the floodplain and consequently floodplain tree health.

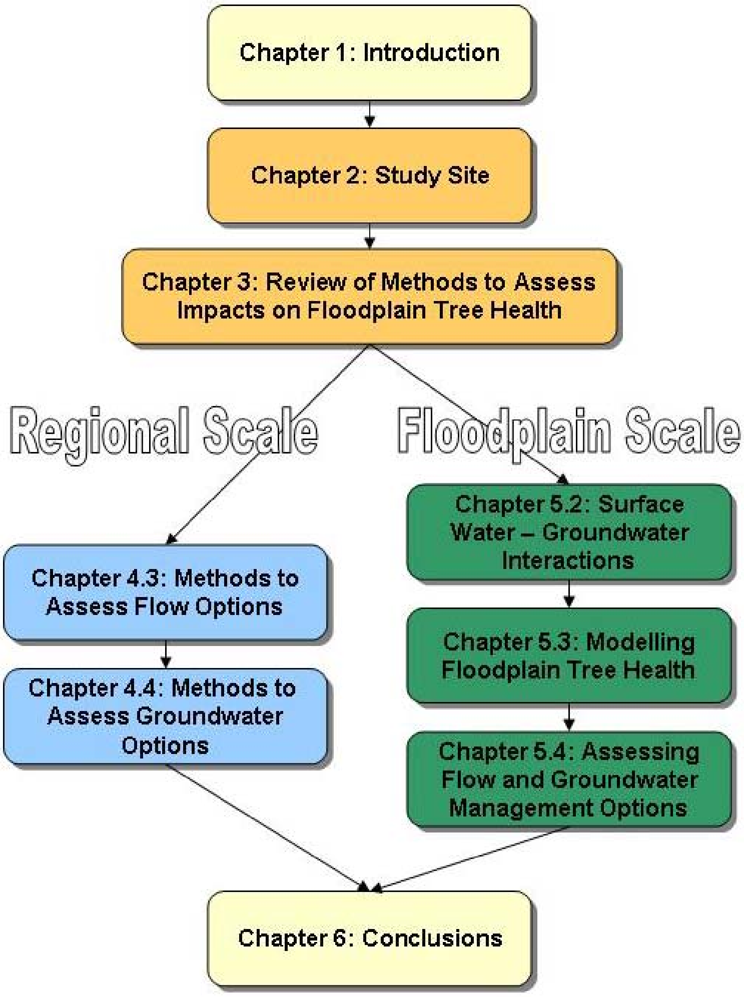

Figure 1.1 Thesis chapter structure showing the regional and floodplain scale approaches taken in assessing surface water and groundwater management scenarios for riparian vegetation health.

Chapter 5 develops methods for assessing surface water and groundwater management scenarios at the floodplain scale. The chapter first links the vegetation on the Chowilla floodplain with the pattern of floodplain inundation. The chapter develops an understanding of surface water and groundwater interactions through flood recharge and groundwater discharge to creeks, and incorporates a groundwater model to predict groundwater levels. It then describes the development of a process model for soil water availability (WINDS) from salinisation which is then used to predict changes in floodplain tree health from environmental flow and groundwater management scenarios on the Chowilla floodplain.

Chapter 6 is the conclusion and discusses the thesis outcomes for assessing management scenarios spatially at a regional and floodplain scale.

2 LOWER RIVER MURRAY ENVIRONMENT

2.1 INTRODUCTION

The MDB is approximately one million square kilometres and is Australia's largest river basin containing its two longest rivers, the Darling and the Murray (Figure 2.1). This chapter describes the environment of the lower River Murray which is the regional scale used in this study. It also describes the Chowilla floodplain which lies within the lower River Murray and represents the floodplain scale. The chapter describes the climate, surface hydrology, hydrogeology, geomorphology and the vegetation of the floodplain and identifies changes that have occurred over time as a consequence of river regulation and extraction.

The climate within the MDB ranges from sub-tropical to arid. The major landuses in the MDB include irrigated crops and pastures, rainfed agriculture, livestock grazing and many other primary industries producing wheat, wool, rice, cotton, fruit and vegetables for domestic and overseas markets. Approximately two million people live within the MDB in many rural townships. The capital city of Australia, Canberra, also lies within the MDB.

The average annual rainfall in the MDB is 530,000 GL/yr which results in approximately 22,000 GL/yr in river flow (4%) with the highest interannual variability of any major river in the world (Pigram, 2007). Of this, approximately 13,000 GL/yr is divertible stream flow. Currently approximately 11,500 GL/yr is diverted for irrigation, commercial and human water use with agricultural irrigation accounting for 95% of the water use (Pigram, 2007). The total storage capacity within the MDB exceeds 22,000 GL.

The lower River Murray is often defined as the River Murray from Mildura, downstream to the Murray Mouth and can include the lakes at the end of the system, Lake Alexandrina and Lake Albert. The lower River Murray is generally considered to be the portion of the river where the basin groundwater aquifer discharges into the river channel, creating a gaining river system along much of this region. In this area, the floodplain is generally one to five kilometres wide and is interspersed with a series of anabranches. This thesis concentrated on the River Murray floodplain from the South Australian border to Wellington at the top of the lower lakes.

To investigate both regional and local scale management scenarios it was necessary to examine the whole of the lower River Murray in South Australia as well as the more localised issues of managing individual floodplains. The Chowilla floodplain is the largest area of native riparian forest in this region and, with its diverse range of previous studies and a substantial collection of data, provided an excellent case study for investigations at a floodplain scale. It does not have any fringing irrigation, but the high regional groundwater inflow causes salinisation problems.

2.1.1 Regional Scale - The Lower River Murray

The lower River Murray is the region at the end of the MDB where the River Murray discharges to the sea (Figure 2.1). The study area was chosen to include the floodplain environments of the River Murray in South Australia from the New South Wales border, to Wellington at the top of the lower lakes.

The lower River Murray is in a semi-arid environment and irrigation development in this low rainfall area relies on regulation of the River Murray and large water extractions. Approximately half of the MDB's run-off is extracted for water use, with 95% of this being used for irrigation (Crabb, 1997). The riparian vegetation is reliant on flooding for survival and is highly adapted to the environment but the environmental conditions have changed leaving much of the tree vegetation in poor condition. The changes to groundwater depth and flooding frequency are significant. Flooding frequency has reduced by a factor of 3 and groundwater levels have risen to within metres of the floodplain surface.

Figure 2.1 Lower River Murray in South Australia showing its location in the Murray-Darling Basin in Australia.

The river in this area is highly regulated by six weirs (referred to as Lock 1 to 6) (Figure 2.2), has extensive floodplains and is a discharge area for the regional saline groundwater system of the MDB. Numerous publications have described aspects of the region including the resources (Mackay and Eastburn, 1990), wetlands (Pressey, 1986; Thompson, 1986; Jensen *et al.*, 1996), vegetation (Van der Sommen, 1987; Margules *et al.*, 1990), river flows (Ohlmeyer, 1991; Sykora *et al.*, 1999; MDBC, 2000) and salinity (Jolly, 2004). The region has also been summarised in MDBA (2010).

The area is a major irrigation region with large irrigation plots established since the foundation of Renmark in the 1920s. Irrigation practices in the past have led to an excess of water seeping into the groundwater, which combined with the removal of deep-rooted perennials, has contributed to the high groundwater levels. The area is a significant economic region for South Australia as well as a large recreational region for boating, fishing and tourism.

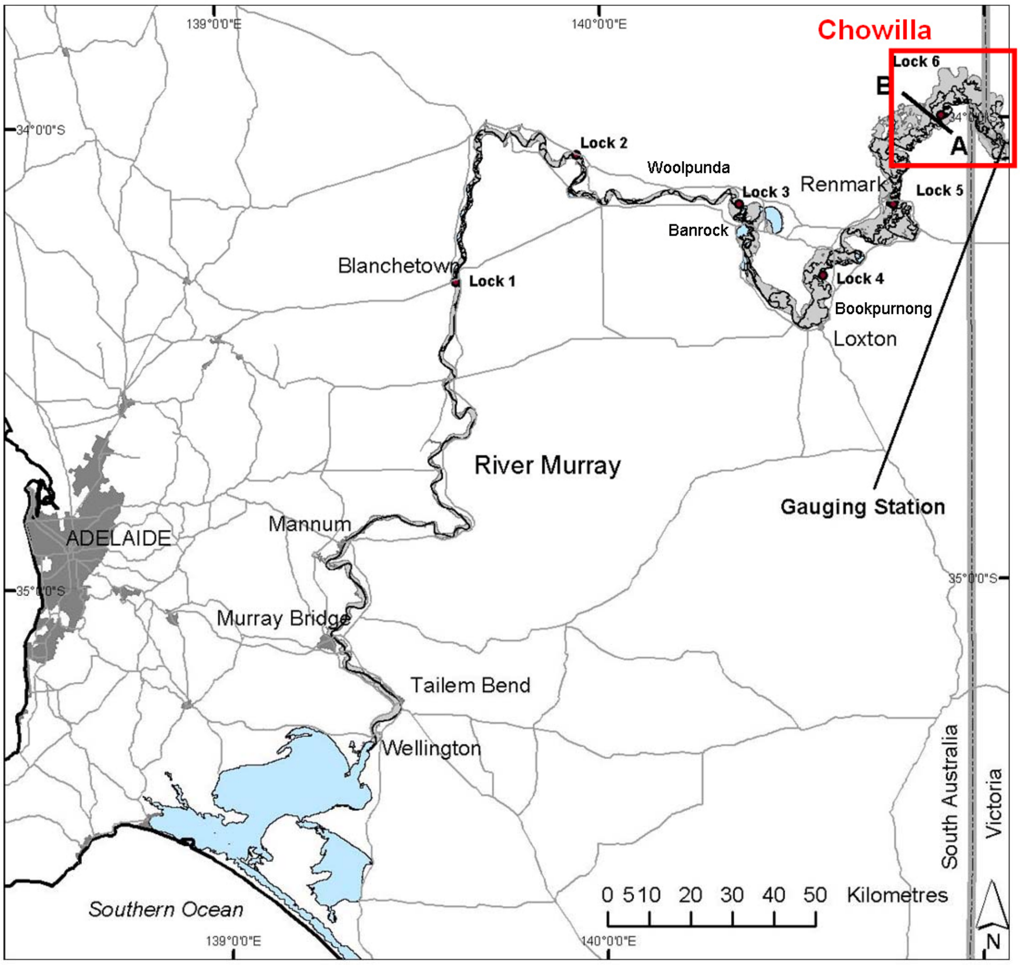

Figure 2.2 Lower River Murray in South Australia showing locks and localities and the location of the Chowilla floodplain in red.

Total extractions are approximately 11,500 GL/yr from an annual average of 22,000 GL of flow. The mouth of the river does not flow during dry periods in the MDB and the estuary, the Coorong, has changed from brackish to hyper-saline. Due to the combined impacts of river regulation, resource over-allocation and reduced rainfall, there has been severe decline of the environmental assets within the MDB, which include semi-permanent and ephemeral wetland vegetation, significant waterbird populations and native fish. Between 1997 and 2009, south-eastern Australia experienced persistent rainfall deficit leading to the worst drought on record, known as the Millennium Drought (SEACI, 2011). This extended dry period with high temperatures exasperated the decline in ecosystem condition.

2.1.2 Floodplain Scale – The Chowilla Floodplain

The Chowilla floodplain is approximately 50 kilometres east of Renmark, and is located on the borders of South Australia, New South Wales and Victoria (Figure 2.3). Chowilla has been the focus of numerous investigations since the proposal of a new storage dam in the 1960s, which would have left the entire Chowilla floodplain and those of Lindsay and Wallpolla under water. It is a large floodplain that was relatively pristine due to its remoteness, but has suffered severe decline in recent times.

The floodplain is the largest area of native riparian vegetation in this region. It requires large floods to inundate the whole floodplain and consequently is a good reference site for the health of the riverine system. The ecological significance of the floodplain is typified by its listing as a wetland of international importance under the Ramsar Convention (National Environmental Consultancy, 1988), as a Significant Ecological Asset under the Murray-Darling Basin Living Murray program (in combination with the Lindsay Wallpolla region in New South Wales and Victoria) and as one of the 18 indicator key ecosystem assets of the Guide to the Basin Plan (MDBA, 2010).

The floodplain is defined as the extent of the 1 in 13 year flood (peak flow of approximately 120,000 ML/day). The floodplain covers approximately 17,960 hectares, interspersed with a network of 100 kilometres of former river channels, anabranches and oxbow lakes. The dominant tree species are black box (*Eucalyptus largiflorens*) occupying the high areas of the floodplain and river red gum (*Eucalyptus camaldulensis) on* lower lying areas, near creeks and wetlands.

The floodplain is typical of those in the lower reaches of the River Murray in that it is underlain by saline groundwater at depths of two to four metres. This is in contrast with floodplains upstream of the lower River Murray region that are generally underlain by fresh groundwater (Jolly *et al.*, 1994a). The floodplain is the largest contributor of salt to the river after Woolpunda which occurs further down the river.

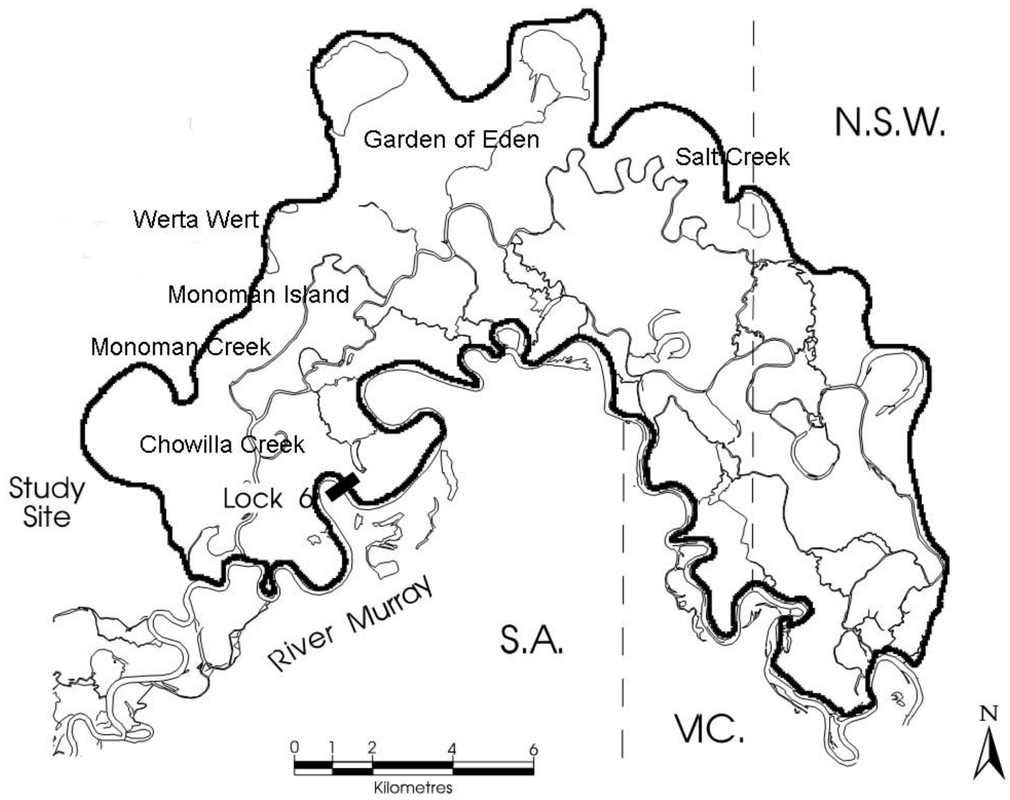

Figure 2.3 The Chowilla anabranch region defined as the limit of the 1 in 13 year flood. Lock 6 lies along the River Murray in Chowilla and there is a major anabranch system that passes around the lock.

Lock 6 lies on the River Murray in the Chowilla floodplain and creates a three metre height difference in river levels. Such a height difference means that the river is much higher than the regional groundwater level upstream of the lock and creates an area of freshwater that supports a dense healthy area of river red gum and black box. The position of the weir drives river flow through the anabranch system, which can be greater than the main channel flow.

The Chowilla floodplain has been grazed for many years and is still under a pastoral lease. In the areas where they are permitted to graze, sheep held on the property, damage the soil and remove new growth as it regenerates.

Numerous publications have described the Chowilla floodplain hydrology, hydrogeology, salinity, vegetation and ecology. Overton and Jolly (2003) provide an extensive bibliography on the Chowilla floodplain. The major publications provide a description of the floodplain and discuss the management issues (Social and Ecological Assessment, 1989; Hollingsworth *et al.*, 1990; Resource Conservation and Management Group, 1993; Sharley and Huggan, 1995; MDBC, 2003; DEH, 2006; DWLBC, 2006). Other publications have described the hydrology and hydrogeology (Waterhouse, 1989; Collingham, 1990a; Collingham, 1990b; Narayan *et al.*, 1993; Noyce and Nicolson, 1993; Charlesworth *et al.*, 1994; Jolly and Walker, 1995; Walker *et al.*, 1996; Yan *et al.*, 2005), and the vegetation and ecology have been well reported (O'Malley, 1990; Jolly *et al.*, 1992a; Eldridge *et al.*, 1993; Thorburn, 1993; Hodgson, 1993; Taylor, 1993; Mensforth *et al.*, 1994; Palmer and Roberts, 1996; Jolly and Walker, 1995; Zubrinich, 1996; Slavich, 1997; Akeroyd *et al.*, 1998; Jensen *et al.*, 1996; DSNR, 2003; Overton and Jolly, 2003; Smith and Kenny, 2005).

2.2 CLIMATE

The region has a semi-arid climate with mean rainfall of approximately 260 mm/year and potential evaporation of approximately 2,000 mm/year. Annual rainfall is extremely variable, ranging from less than 100 mm/year to over 500 mm/year. The distribution of mean monthly rainfall has a slight winter dominance (Jolly *et al.*, 1994a).

Plants have adapted to be drought tolerant to some degree as evaporation greatly exceeds rainfall. Although the amount of rainfall is small, it is still a critical factor in providing fresh water to plants and in leaching out accumulated salt from soils. Black box can exist on the high parts of the floodplain above the one in a hundred year flood by surviving on rainfall (Jolly, 1996). The Millennium Drought in combination with reduced flooding and rising salinity has caused widespread dieback of many river red gum trees (DEH, 2003).

The Murray-Darling Basin Sustainable Yields project (CSIRO, 2008) identified the surface and groundwater resources available in the MDB under future management and climate scenarios. This study reported that "*Water resource development has caused major changes in the flooding regimes that support nationally and internationally important floodplain wetland systems in the MDB*".

The floods of 2010 and 2011 provided some relief for ecosystems in the lower River Murray with a 1 in 10 year flood experienced over much of the floodplain in both years. With future climate changes, drought conditions in the south of the MDB will become increasingly common and surface water availability across the entire MDB is more likely to decline than increase (CSIRO, 2008). Global climate models predict a range of possible futures and the predictions for the south of the basin are that a substantial decline in water availability is possible, while in the north of the basin, significant increases are possible (CSIRO, 2008).

2.3 SURFACE HYDROLOGY

Surface flows from the MDB discharge through the River Murray and into the Southern Ocean. Of the 22,000 GL/yr of surface flow in the MDB, only a small portion of this makes it through the River Murray. Since regulation the flow in the River Murray has reduced as a result of extraction. This becomes pronounced during low rainfall periods such as the Millennium Drought. Figure 2.4 shows the river flow from 1892 to 2010.

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Figure 2.4. Annual inflow in the Murray-Darling Basin since 1892 (MDBA, 2011).

The floodplain environment is first and foremost a riparian system which relies on periodic flooding to maintain its proper ecosystem functioning. Floods provide freshwater, seeds, fish larvae and nutrients from up-stream. As a flood moves over the floodplain, it provides the trigger for flowering and spawning. Flooding also recycles the organic carbon on the floodplain, providing further nutrients. Flood waters bring more than just water and wetland pumping strategies, discussed later, are therefore not a long-term replacement for flood events. Because the floodplain topography is very flat small changes in river elevation can inundate extensive areas of floodplain.

Flood waters also provide a mechanism for leaching of accumulated salts in the soil profile. Without leaching, salt accumulates at much faster rates. Prior to regulation the River Murray used to periodically cease to flow. Since regulation flows have varied from approximately 3,000 ML/day to approximately 250,000 ML/day in 1956, which is the largest flood on record. There is some evidence based on tree ages that even larger floods have occurred on the River Murray. Snowball (2001) measured black box tree ages by radiocarbon dating and has suggested that a flood 2.11 metres higher than the 1956 occurred around 1750 and estimated it to be a 1 in 1,000 year event. Usually floods of approximately 150,000 ML/day are the maximum seen in the lower River Murray and represent an approximate 1 in 13 year flood event.

2.3.1 River Regulation and Extraction

Locks and weirs were originally installed for navigation as the River Murray dried out during drought periods. River regulation is now maintained for irrigation supply. Such regulation using control structures has raised water levels up-stream of each lock, creating a number of permanent wetlands. The effect of these permanent wetlands has been to drown river red gums (Figure 2.5) along significant stretches of the River Murray. Lock operation in the past has kept water levels as constant as possible, which has led to changes in the hydrological and ecological nature of the river channel and wetlands, by favouring introduced species such as willows and carp and reducing reeds and natural geomorphological processes.

As the river channel is in hydraulic connection with the underlying groundwater, permanently raising the river level with weirs has meant that groundwater has risen closer to the surface over much of the lower River Murray floodplain. This has been a major factor in increased salinisation of the floodplain upstream of the locks.

Figure 2.5 Many wetlands have become permanent such as the upper pool wetland at Lock 6 in Chowilla. Drowned river red gums are a common site upstream of locks.

Within the Chowilla floodplain region there is an anabranch creek which traverses the floodplain, bypassing the number 6 weir. The anabranch creek intercepts the regional groundwater and, as the creek is lower than the river channel in this region, it creates a reverse gradient from the main river to the anabranch creek. The main river then loses water into the bank and creates a 'flush zone' of fresher groundwater in the area between of the anabranch creek and the main channel.

2.3.2 Changes in River Flows and Floods

Since regulation and water extraction began in the 1920s, there has been a large decrease in the volume and the frequency of flood returns. The median annual flow from the MDB to the sea is only 27% of the flow that would have occurred prior to development. Figure 2.6 highlights the change in flood volume and frequency into South Australia as a result of regulation. The figure clearly shows the reduction in flows of all three flow bands.

There has also been a reduction in the return period of medium sized floods by a factor of three since regulation (Ohlmeyer, 1991). Figure 2.7 illustrates the return periods of floods in the lower River Murray, and indicates the impact on the flow bands where the majority of river red gum and black box species occur. The graph shows the reduction in return intervals of floods to areas occupied by black box from 1 in 2 years to 1 in 5 years and for river red gum stands 1 in 1.3 years to 1 in 3 years.

Figure 2.8 highlights the median flow each month into South Australia. Unlike areas up-stream, there has been little change to the seasonality of the peak flows although there is a small, approximately one month, shift with the peak of flow occurring in September at present compared to October under natural conditions. An audit in 1995 (MDBC, 1996) showed that diversions across the MDB were averaging 10,800 GL/yr. In response to the findings a permanent 'Cap' on water diversions was implemented in July 1997. The Cap is defined as: *'the volume of water that would have been diverted under 1993/94 levels of development. In an unregulated river this Cap may be expressed as an end-of-valley flow regime'.*

Water extractions have increased in recent years so that South Australia receives only its 'entitlement' under the Cap for most of the year (Figure 2.8). Flows to South Australia have been particularly low since 2003 during the drought period. Figure 2.9 indicates the flow into South Australia as measured at the gauging station located just up-stream in New South Wales (gauge number GS4261001). The graph highlights the flow bands where most of the river red gum and black box trees occur. The river red gums have received only one small flood in the 15 years prior to 2011 and that black box trees had not been flooded since 1994.

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Figure 2.6 Change in monthly flows at the South Australian border under natural and current conditions (MDBC, 2003). Years from 1890 to 2000 are presented down the page and months across the page in each column. Each month is given a colour based on the flow band for the peak flow conditions indicated in grey, light blue and dark blue for different size flows. The column on the left is for natural conditions pre-regulation and the column on the right is the same flow that would have occurred if no regulation was present.

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Figure 2.7 Peak flow recurrence intervals for natural and current conditions (after National Environmental Consultancy, 1988).

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Figure 2.8 Median monthly flows at the South Australian border under natural and current conditions, and the entitlement of SA (MDBC, 2003). The median Demonstration Flow is under a flow regime set by the Living Murray or an extra 500 GL/yr.

The change in the frequency of inundation of the floodplain is shown in Figure 2.10. This figure highlights that the Chowilla floodplain is one of the most highly impacted areas with the MDB. The average return interval for floods, overbank flows, has gone from 1 in 3 with natural conditions to 1 in 9 with current flows and is predicted to decline to 1 in 18 under a median 2030 climate as modelled by a range of global climate models (CSIRO, 2008).

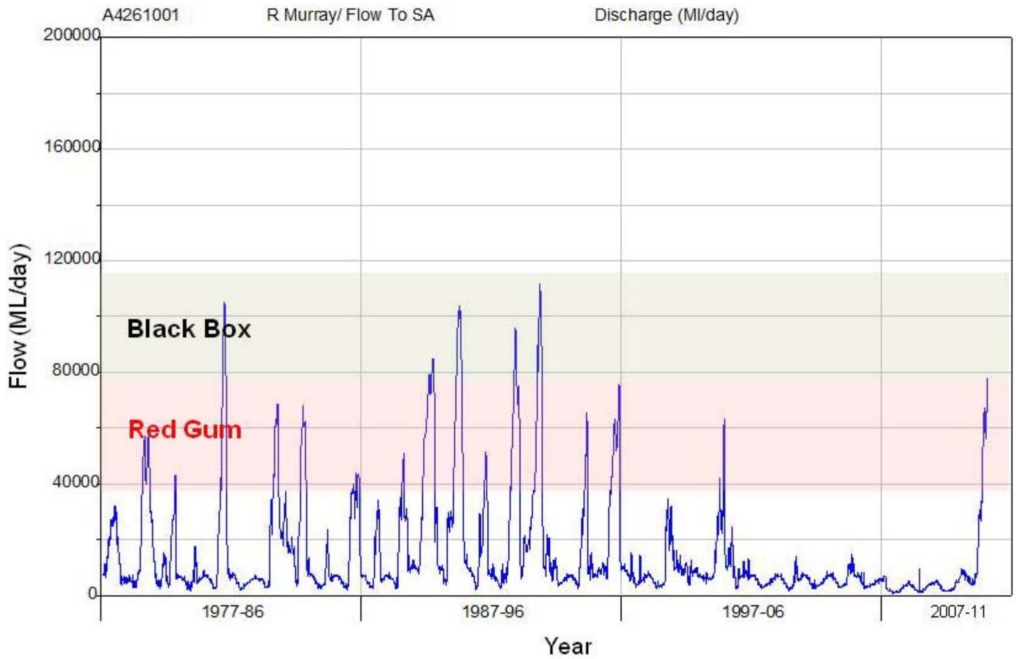

Figure 2.9 Flow to South Australia during 1977 to 2011 showing the flow bands for inundating the majority of river red gum and black box forests and woodlands.

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Figure 2.10. Changes in the average period between floods for a range of wetlands and floodplains in the Murray-Darling Basin (CSIRO, 2008).

2.3.3 Water Quality

Surface water quality has two components. The first is the salinity of the water which can be a concern when the water is extracted for human consumption and, when in higher concentrations, for irrigation. Since river regulation and the rise in saline groundwater aquifers there has been an increase in the salinity of river water. The concentration of salt in the river water is in one sense an equilibrium between the salt being stored in the floodplain soils. Large flood events release salt into the river during the flood recession but improve the situation for the floodplain soils. This thesis is not concerned with salt concentrations within the river.

Other water quality issues such as acid sulphate soils, black water events and blue-green algal blooms are important for ecosystem health but are not directly related to broadscale floodplain tree health and are therefore not considered in this study.

2.4 HYDROGEOLOGY AND GEOMORPHOLOGY

The lower River Murray is a sink for the regional groundwater system which discharges into the floodplain (Figure 2.11). The underlying aquifer is highly saline (approximately 60 dS/m) and comes close to the surface of the floodplain as it enters the River Murray. Much of the regional groundwater flow to the River Murray passes through the floodplain subsurface soils within the river valley (Barnett, 1989).

Groundwater flow for the floodplains east of Loxton in South Australia comes mainly from the saline Parilla Sands aquifer (approximately 60 dS/m), with some upward leakage from the fresher confined Murray Group Limestone aquifer (approximately 40 dS/m) (Barnett, 1991). The average groundwater inflow into the underlying Monoman Formation aquifer is approximately 5 ML/day, resulting in a salt load to the River Murray of approximately 120 tonnes a day (45,000 tonnes per year) (Chowilla Working Group, 1992).

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Figure 2.11 Map showing the regional groundwater flow directions and salinity in the Lower River Murray region (after Barnett, 1989).

The Chowilla floodplain contributes the highest inflow of saline groundwater into the River Murray in South Australia after the Woolpunda region which occurs west of Loxton. In this region the Murray Group aquifer is unconfined and would be generally fresher if not for the influence of upward leakage of the very saline groundwater of the Renmark Group.

The floodplain is covered by a layer of alluvial clay known as the Coonambidgal Clay (Hollingsworth *et al.*, 1990). The clay can be up to five metres thick close to existing or prior creek beds. The clay is absent on high floodplain areas and active creek beds. Jolly *et al.* (1994a) notes that the presence and thickness of the clay is an important controlling factor on the hydrology and vegetation of the floodplain as the dispersive clay limits infiltration. The Coonambidgal Clay overlies an unconsolidated alluvial sand deposit (Monoman Formation) which is approximately 30 metres deep, consisting of fine to coarse sand with varying amounts of clay and silt. The Monoman Formation is an unconfined aquifer which is thought to be in direct hydraulic contact with the regional unconfined Pliocene Sands aquifer and the streams of the floodplain (Jolly *et al.*, 1994a) (Figure 2.12).

Some areas of the floodplain have fresh groundwater. These areas act as recharge zones due to the presence of sandier soil profiles. The freshwater that percolates through the soil profile does not mix with the saline groundwater due to density differences between the two different concentrations. Fresh water sits above the saline groundwater and forms a ‘freshwater lens’. In areas adjacent to the river, where high river level pressure forces recharge of the adjoining aquifer through bank recharge, a ‘flushed zone’ is created. In both these areas (freshwater lens and flushed zone) the salinity of the soil water may be less than 10 dS/m, similar to main channel river water. In all other floodplain areas, salinity ranges from 35 dS/m to 85 dS/m (Noyce and Nicolson, 1993).

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Figure 2.12 Generalised hydrogeological cross section of the Chowilla floodplain (after Waterhouse, 1989). Vertical elevation is given in metres above the Australian Height Datum.

The local geomorphology ranges from gorge sections below Overland Corner (2-3km wide and 30-40m deep) to valleys 5-10km wide flanked by a broad floodplain (Walker and Thoms, 1993). The floodplain edge is defined by steep rises in most areas.

2.4.1 Land Management

Since European settlement, vast areas of the MDB have been cleared for agricultural development, where deep rooted trees are replaced with shallow rooted annual crops. The effect of this has been to increase groundwater recharge and flow of water to the river. This process has been accelerated by the introduction of irrigation along the River Murray. In the lower River Murray large areas (approximately 39,000 ha) have been irrigated since the 1940s with all water sourced from the Murray (Figure 2.13).

In some areas, the irrigation occurs right to the edge of the floodplain. Excess water applied to irrigated crops and citrus trees percolates down to the water table, increasing the discharge to the floodplain and river.

When this discharge is too great for the transmission through the floodplain soils, it flows out at the soil surface at the base of the cliffs and is referred to as seepage (Figure 2.14). The impacts of land management on the floodplain have been severe and now leave water tables within metres of the floodplain surface, causing the identified decline in tree species, as well as development of salt crusting in severe cases.

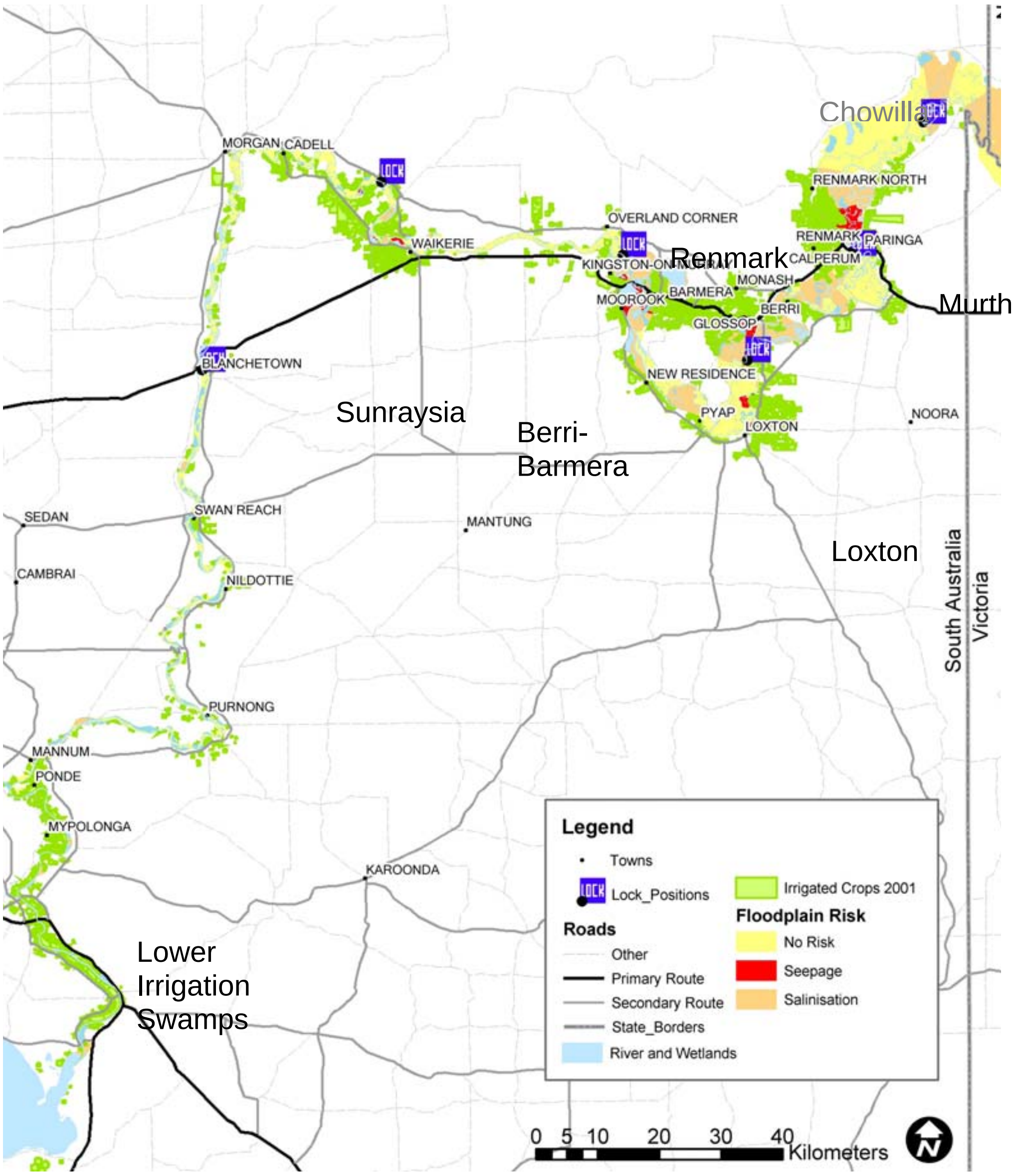

Figure 2.13 Irrigation in the lower River Murray in 2001 shown in green. The major irrigation areas are labelled. Floodplain risk modelled using the FIP model described in chapter 5. Note that Chowilla does not have any fringing irrigation.

Only a few floodplain areas are still grazed by livestock, however the effects of previous and current grazing are clearly visible in the absence of seedlings and compaction of the soil surface. The Chowilla floodplain has very few young black box trees as seedling establishment has decreased through reduced flooding, while grazing of new saplings has left only mature trees.

Figure 2.14 Seepage of saline water onto the floodplain at the base of the cliffs caused by irrigation.

The floodplain is under increasing attention from state and local government through the development of policy and land management strategies. The Basin Plan for the MDB (MDBA, 2010) is addressing the balance between a healthy environment and irrigation water use. At a regional scale, a key objective of the River Murray Water Allocation Plan (RMCWMB, 2002) is that the transfer, allocation and use of water must not have adverse impacts on the health, biodiversity status or habitat value of floodplains or wetlands of conservation significance. At a floodplain scale, a Ramsar site management plan (DEH, 2006) and a Chowilla Significant Ecological Asset Water Management Plan (DWLBC, 2006) are currently being developed. Targets for the protection of the floodplain have been developed under the MDBC Living Murray Initiative (MDBC, 2003) to preserve 70% of the river red gum communities present today and 20% of the black box communities. The Chowilla Significant Ecological Asset Water Management Plan has set targets of maintaining at least 70% of the area of river red gum and 45% of the area of black box in good health condition, and has identified areas to be managed, in part as a result of work undertaken for this thesis.

The South Australian River Murray Salinity Strategy (DWR, 2001) has proposed several salinity management scenarios to ameliorate the impact of salinity on floodplain and wetland health, in addition to existing measures to address in-stream salinity impacts. The salinity management scenarios include flow management, on-ground works and the establishment of floodplain and wetland protection zones.

Hence there is a need for a modelling tool to assist in the prediction of the impact of current and future irrigation developments and river management on floodplain salinisation.

2.4.2 Changes in Groundwater Depth and Salinisation

As a result of land clearing and raised river levels the depth of the saline groundwater aquifers in the MDB has risen. This is particularly the case in the lower River Murray and the Chowilla floodplain where groundwater levels are now within metres of the floodplain surface.

As a result of raised groundwater levels and reduced flooding frequency salt has been accumulating within the floodplain soils to the extent that in some areas the soil salt concentration is greater than sea water.

2.5 VEGETATION

The vegetation of the lower River Murray and the Chowilla floodplain comprises predominately native floodplain species that have occurred in that region since long before European settlement. Some of the floodplain trees have been dated to over 500 years old (Slavich, 1997). The vegetation of the lower River Murray was mapped from aerial photography and field observations by Margules *et al.* in 1990. This mapping has known spatial and attribute errors and the vegetation has been remapped as part of this study.

The distribution of the riparian vegetation is closely linked to the flooding frequency, governed by surface elevation (Margules *et al.*, 1990). Floodplain vegetation exists across distinct elevation gradients, with different species present at different elevations. This is an adaptation to the flooding regime. Bren and Gibbs (1986) have shown that the floodplain communities in the Barmah Forest occupied specific flooding frequency/duration niches, although there was considerable variability and overlap. The Chowilla floodplain has been mapped and a similar relationship between vegetation and flow regime was found (Van Der Sommen, 1987).

The most common tree species in the lower River Murray is black box (*Eucalyptus largiflorens*) (Figure 2.15a). This is the dominant species on the Chowilla floodplain, covering approximately 37% of the area (Noyce and Nicolson, 1993). In the lower lying black box areas, the trees are associated with tangled lignum (*Muehlenbeckia florulenta*), which covers approximately 10% of the floodplain, ruby saltbush (*Enchylaena tomentosa*), warrego summer grass (*Paspalidum jubiflorum*) and other ephemeral grasses and some sedges. Black box is more salt tolerant than river red gum (Thorburn, 1993).

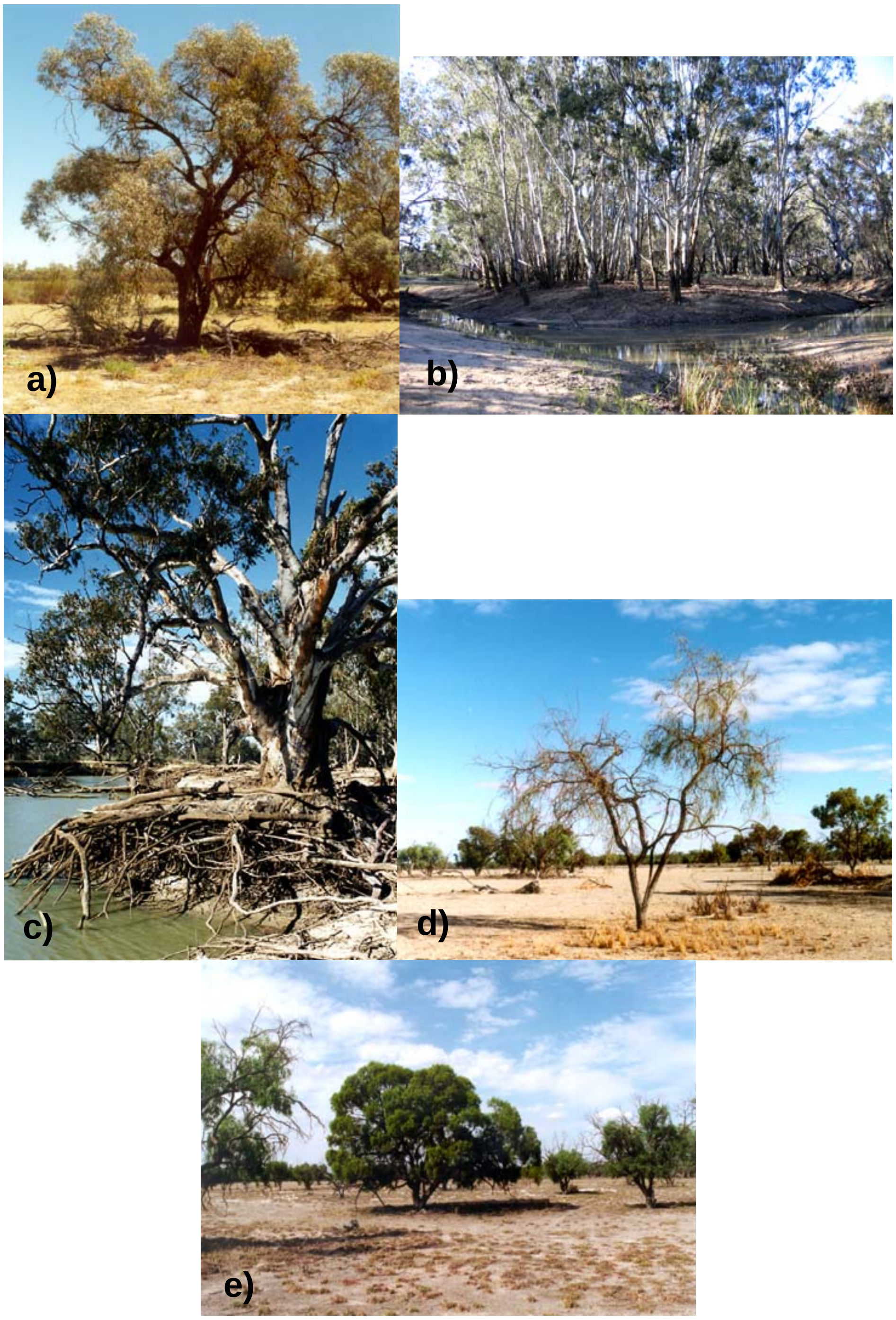

Figure 2.15 A) Black box is the dominant tree on the higher parts of the floodplain in the lower River Murray. B) Dense river red gum forest occurs on the sandy banks near creeks. C) Large river red gums line the edges of creeks and wetlands. D) River cooba is the third most dominant tree on the floodplain, this one displaying poor health. E) The green variant of black box can be seen easily amongst the less salt tolerant black box.

In the low-lying areas and along the river channels river red gum (*Eucalyptus camaldulensis var camaldulensis)* dominates. This iconic species occupies approximately 4% of the floodplain but is common along most creek lines and fringing wetlands. River red gum can be found in dense forests where water availability permits (Figure 2.15b) but also as large single trees that can live hundreds of years (Figure 2.15c).

River cooba (*Acacia stenophylla*) (Figure 2.15d) occurs throughout the floodplain in isolated areas and is found in association with river red gum and black box. It is the third most common tree species on the floodplain. It is associated with ephemeral grasses, and in those areas infrequently flooded, with perennial saltbushes (*Atriplex nummularia*, *A. rhagodioides* and *A. vesicaria*) (Hollingsworth *et al.*, 1990).

At higher elevations above the level of the black box, native Callitris pines (*Callitris preisii*) and chenopod shrublands exist. Mallee woodlands extend beyond the floodplain and is dominated by *Eucalyptus gracilis*, which is very drought and salt tolerant as its existence relies on rainfall alone. A genetic hybrid of the mallee eucalypt and black box known as a 'Green Variant' occurs in a few locations on the floodplain as isolated trees (Figure 2.15e). The hybrid is more drought and salt tolerant than the black box. This hybrid has the potential to be used in re-vegetation strategies if the seeds can be mass produced (Zubrinich, 1996).

Smith and Smith (1989) note that while the flooding regime may be the most important factor determining vegetation patterns on a floodplain scale, at the broader scale it is soil salinity that appears to have the dominant effect.

2.5.1 Changes in Tree Health

Much of the tree vegetation on the floodplain is in poor condition. The trees are deep rooted and large water users (Thorburn, 1993) and are good indicators of soil water availability. Stressed trees show a reduction in canopy cover as a result of increased water potentials required to extract water from the soil due to dryness (matric potential) and salinity (osmotic potential). They also exhibit epicormic growth and mistletoe infestations. Miller *et al.* (2003) showed that mistletoe is likely to attack a stressed tree rather than being the cause of the stress itself (Figure 2.16).

Figure 2.16 Black box in poor health exhibiting signs of stress through loss of canopy cover, epicormic growth and mistletoe attack.

A large tree die-back event occurred in the summers of 2002 and 2003 (DEH, 2006). Drought over the proceeding years had increased stress for trees that were already suffering from flooding reduction and increasing salt accumulation. Such an example of die-back occurred at the Monoman Island horseshoe (Figure 2.17) where fringing river red gum trees were in very poor condition.

At the commencement of this study no current vegetation map existed that depicted the location of the different tree species and their health. A more recent remote sensing estimate of tree health in 2008 found that approximately 20% of floodplain trees were in good health (Cunningham *et al.*, 2008). This compares to 35% in good health in 2003 (DEH, 2006), 60% in 1994 (Taylor (1993) and 82% in 1990 (Margules *et al.*, 1990).

Figure 2.17 Many areas of river red gums have died since the summer of 2002.

3 ASSESSING IMPACTS ON FLOODPLAIN TREE HEALTH

3.1 INTRODUCTION

This chapter reviews current knowledge on floodplain tree health and the processes of salt accumulation. It also reviews previous methods for mapping and predictive modelling floodplain tree health. The chapter identifies the key environmental factors determining vegetation health and provides a direction for the investigations and modelling undertaken in subsequent chapters.

The objective of this chapter is to understand the processes that affect floodplain tree health, how tree health has been modelled previously and how management scenarios have been assessed.

3.2 MAPPING FLOODPLAIN TREE HEALTH

The problem of vegetation decline in the River Murray floodplain in South Australia has been noted for many years. Margules *et al.* (1990) recorded that 18% of the vegetation in the lower River Murray was in poor health. This figure rose to 40% in 1994 (Taylor *et al.*, 1996), 65% in 2003 (DEH, 2006) and 80% in 2008 (Cunningham *et al.*, 2008).

Margules *et al.* (1990) produced the first complete vegetation survey of the region. The vegetation was mapped into 15 units at scales of 1:50,000 to 1:100,000 from interpretation of aerial photographs and existing vegetation maps. The resultant boundaries between units were digitised by the Victorian Department of Conservation and Environment. However, errors have been identified with misclassification of units in some areas and spatial inaccuracy of units in others (MDBC *pers. comm.*, 2005).

There are a number of vegetation health and groundwater monitoring sites on the Chowilla floodplain recorded by CSIRO from 1991 to 1996. Black box health, groundwater levels and salinity were recorded in the western part of the floodplain (McEwan *et al.*, 1995). This data was used in developing the relationships between vegetation health and salinity that were used in later modelling and assessment discussed below. However the mapping was not comprehensive enough to be used in the modelling for this study. Also no mapping was undertaken for the Chowilla floodplain.

Health assessment from field observation is subjective and subject to variation in interpretation from different observers and different observation dates. Eldridge (1991) developed a field tree assessment that was modified from Grimes (1987). The original method was developed for forestry applications and Eldridge used indicators that were suitable for open woodland trees on the floodplain. The four indicators used were canopy shape, density of canopy cover, number of dead branches and the presence of epicormic growth. Each factor was scored from 1 to 5 giving a total health score of 4 to 20 with 20 being perfect health and 4 being dead. This method has been used in this thesis for rating black box and river red gum health.

Taylor *et al.* (1996) recorded the health of black box communities only on the Chowilla floodplain using a 'Good' or 'Poor' classification based on crown density (Figure 3.1). Although this approach was useful, the distinction of health classes was insufficient to be used for further health assessment to validate a predictive model.

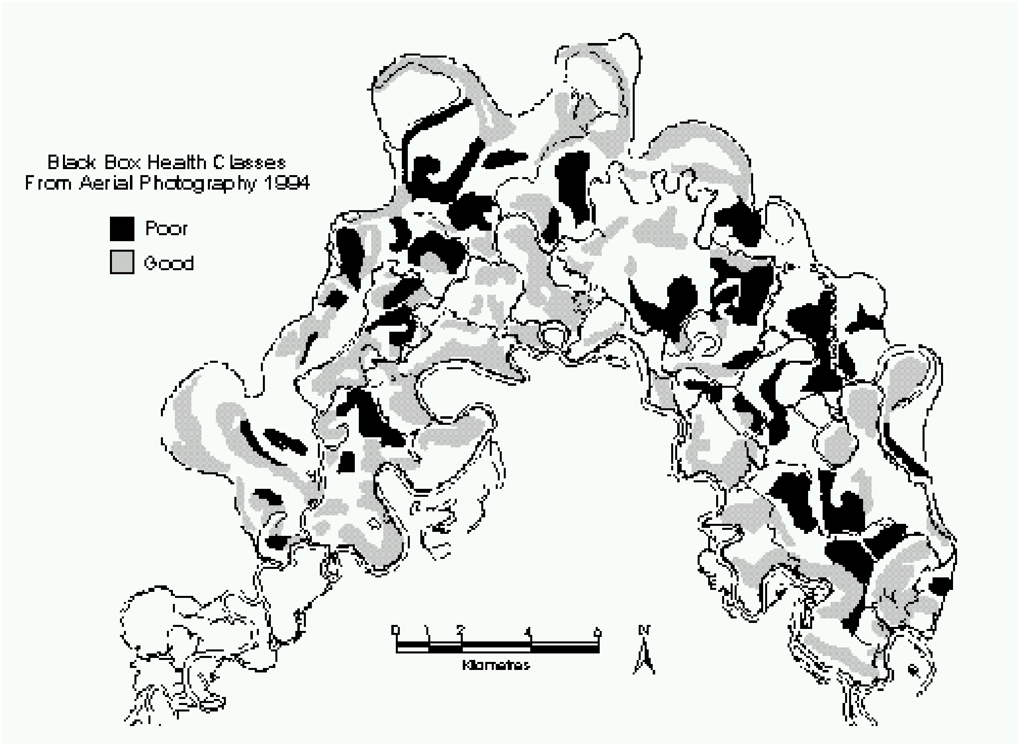

Figure 3.1 Black box health classes interpreted from 1:40,000 aerial photography (1994) showing areas of poor and good health (Taylor *et al.*, 1996).

Between 1998 and 2000 a number of studies were undertaken to map the health of floodplain vegetation in parts of the lower River Murray (Telfer *et al.*, 1998; Telfer and Overton, 1999a; Telfer and Overton, 1999b; Cooling and Overton, 1999; Telfer *et al.*, 2000). The areas mapped included the Loxton, Bookpurnong, Pyap to Overland Corner, Renmark, lower Pike, Ral Ral and Woolenook Bend floodplains (Figure 3.2).

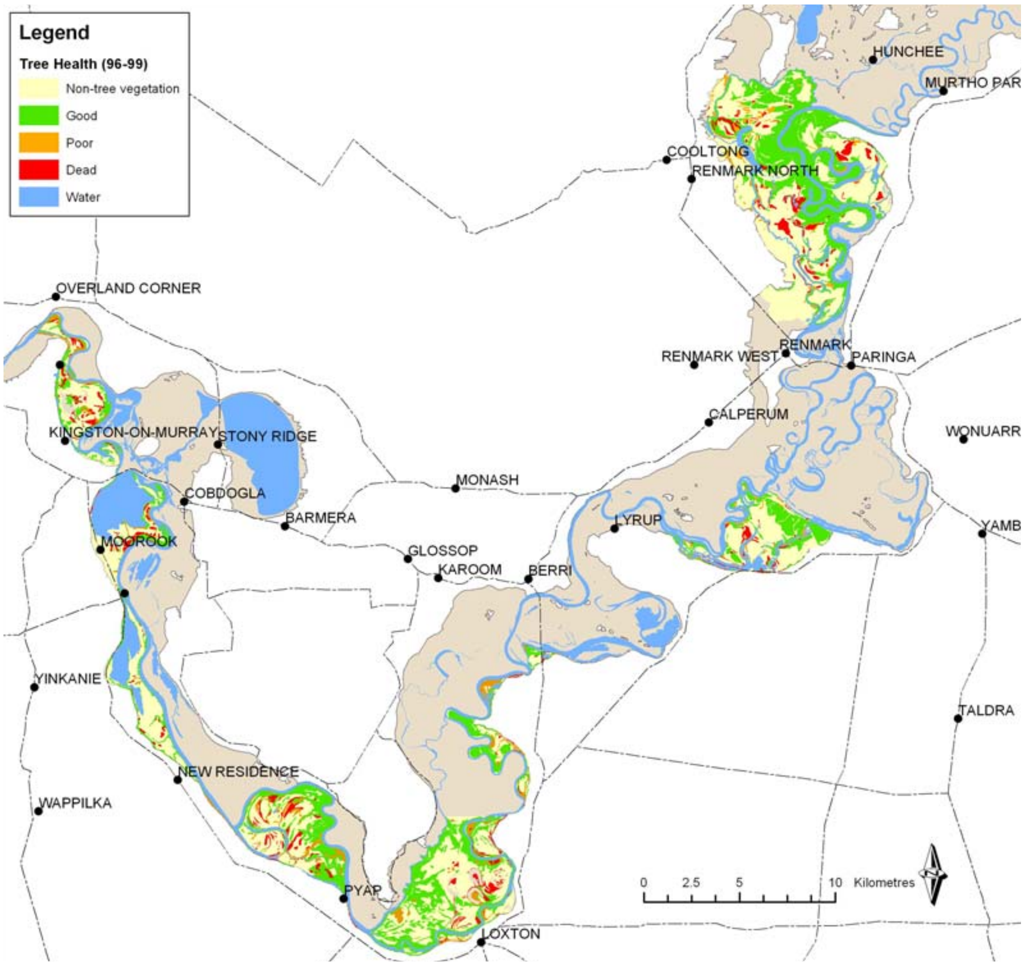

Figure 3.2 Vegetation health map for 1996-1999 (Telfer *et al.*, 1998; Telfer and Overton, 1999a; Cooling and Overton, 1999; Telfer *et al.*, 2000).

The methodology used field observations and aerial photography including infra-red and true colour at 1:10,000 scale to define vegetation boundaries and health. The process produced vegetation maps at a scale of 1:10,000. The vegetation was mapped for communities that have overstorey trees. Vegetation health was mapped for the tree species only and one health value of good, poor or dead was applied to each vegetation area. This approach improved on the level of tree health distinction but was time consuming and not considered suitable to extend to all areas of the floodplain for this study.

The previous authors also undertook a record of historic vegetation change by mapping earlier dates derived from historic photographs. The earliest photography available for the floodplain was an RAAF survey undertaken in 1945. As the vegetation communities change slowly it was possible to use current condition of vegetation type and health as a reference and track backwards though past photography to determine when the tree vegetation was last in good condition. The task was made easier as the vegetation type in 1945 was generally the same as it is today and the health of the tree species was all (apart from a few small areas) of good health. Figure 3.3 shows the health map for the lower River Murray areas in 1945.

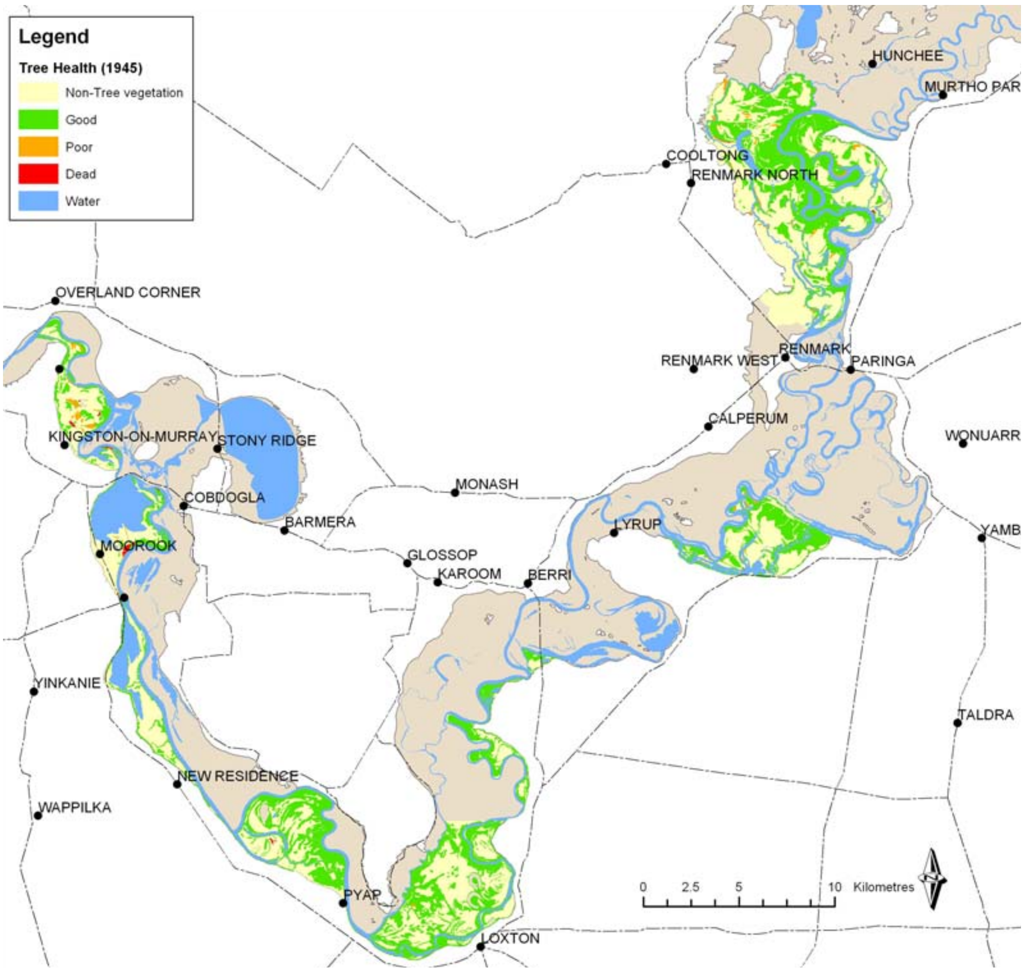

Figure 3.3 Vegetation health map for 1945 (Telfer *et al.*, 1998; Telfer and Overton, 1999a; Cooling and Overton, 1999; Telfer *et al.*, 2000).

This mapping was useful for validating vegetation health predictions for multiple dates.

In 2003 the South Australian Department of Environment and Heritage (Smith and Kenny, 2005) undertook a field floristic, structural and health survey of the vegetation (Figure 3.4). The floristic survey classified the floodplain into 63 classes based on full species lists at 205 quadrats, grouped with pattern analysis. The field health classification method used was similar to Eldridge (1991) but a simpler classification of six health ratings from 0 (dead) to healthy (5) based on the canopy cover, dead branches and epicormic growth, although it ignored canopy shape. This field classification was developed by Holland (2002) modified from Lay and Meissner (1985).

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Figure 3.4 Lower River Murray between Overland Corner and the South Australian border showing vegetation health as mapped in 2003. Mapping from aerial photography and field assessment by Smith and Kenny (2005).

The Smith and Kenny (2005) mapping covered only the South Australian portion of the Chowilla floodplain. A survey was undertaken in the New South Wales section (DSNR, 2003) using a similar but not identical method. Most of the vegetation was classified for health using the Lay and Meissner method (1985) of 5 health classes but then areas of vegetation were given the range of health classes for trees as one of 11 classes (Figure 3.5).

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Figure 3.5 Tree and perennial shrub vegetation health map for the New South Wales side of the Chowilla floodplain (DSNR, 2003).

The method undertaken by Smith and Kenny (2005) and DSNR (2003) improved the spatial accuracy of vegetation mapping and provided the most up to date vegetation health mapping. This study built on these two datasets to develop a vegetation health map for use in the predictive modelling explained in Chapter 5.

To date most of the vegetation type and health mapping has been undertaken using field observations and aerial photography. Multispectral remote sensing has been used successfully in other regions to map vegetation health using the different reflectance characteristics of good and poor vegetation (Campbell, 2002). Vegetation in a good health has higher concentrations of chlorophyll and absorbs more red light than vegetation in poor health. Healthy vegetation also has more turgid cells in its leaves which reflect more infra-red light than unhealthy wilting leaves (Gates, 1980). The difference in the absorption of red light and reflectance of infra-red light can be used to map vegetation health as this difference is unique to healthy vegetation. The ratio between red and infrared reflectance is commonly used as an index of vegetation vigour. The normalised difference vegetation index (NDVI) is a common index used for vegetation health mapping (Crippen, 1990). In many cases Multispectral satellite images such as Landsat Multi-Spectral Scanner (MSS), Landsat Thematic Mapper and SPOT have been used to discriminate between healthy and unhealthy vegetation (Xie *et al.*, 2008), for example in Swedish mountain ranges (Nordberg and Evertson, 2003). The advantage of remote sensing is the ability to map large areas of land quickly with spatial accuracy. Another advantage is that repeated mapping at later dates to monitor changes over time can be achieved more efficiently than by field based methods.

Satellite image analysis has been used with repeated images to monitor changes over time to land cover and vegetation health. There are shortcomings with using satellite imagery for vegetation type and health mapping, which include difficulties in distinguishing species, confusion over mixed species and vegetation types at the ground resolution of the sensor, difficulty in mapping tree health separate from understorey vigour, confusion over vegetation vigour versus density of plants, and calibration of multiple images for consistency over time.

SKM (2005) used the modified Grimes health classification method (Eldridge, 1991) to map the vegetation health on Lindsay and Wallpolla Islands on the lower River Murray east of Chowilla. They also used high resolution Quick Bird multispectral satellite imagery to define a canopy density rating. They found a good correlation between the satellite derived canopy density score and the field health score for two of the three study sites. This shows that vegetation health scores can increase with increasing tree density and that these two main factors have to be separated.

Hyperspectral imagery offers increased number of bands with narrower wavelengths of reflected light measured over satellite imagery (Varshney and Arora, 2004). This hyperspectral imagery allows fine discrimination of reflectance across the visible and infra-red range for improved analysis of vegetation structure and health. The high spatial resolution achieved with airborne instruments reduces the averaging of vegetation types and health values in each pixel. The imagery has been used in eucalypt forests in south-eastern Australia (Coops *et* al., 2006; Goodwin *et al.*, 2004).

High spatial imagery is necessary as spatial variability in vegetation health can be extremely fine, with individual trees having significantly different health than their neighbours (Figure 3.6).

Figure 3.6 The spatial variability of the groundwater and soils can, along with local depressions for water accumulation, lead to tree by tree spatial variability.

This study attempted to improve the field and aerial photography based vegetation mapping using satellite imagery but results were confounded by the issues raised above. Validation of models at the regional and floodplain scale were therefore undertaken using the field based and aerial photography derived mapping as explained in later chapters.

3.3 FACTORS AFFECTING FLOODPLAIN TREE HEALTH

Floodplain vegetation relies on a periodic water source from floods and occurs on characteristic floodplain soils which provide a distinct habitat different from the surrounding highlands. Because of the low elevation of the floodplain, it is also an environment where trees can access groundwater, if present, which can sustain them between floods. The research in this thesis focuses on a semi-arid region where potential evaporation exceeds rainfall and the trees are reliant on the periodic flooding.

The lack of water in the soil profile is attributed as the major cause of plant decline on the floodplain (Jolly *et al.*, 1993a). In many areas soil salinity and water content work together to reduce the amount of water available to plants. In other areas, such as higher floodplain elevations, the effect of drought can be the primary cause of poor health. Decrease in flooding leads to drought stress, as flooding provides a source of water in an otherwise dry environment. The alternative water source can be an increase in soil water from lateral recharge or the presence of groundwater of sufficiently low salinity.

Other factors affecting water availability include soil type, distance to a water supply such as a creek or wetland, rainfall and evaporation. These can be of critical importance to vegetation relying on such water sources.

Streeter (1993) found that Eucalypts growing in areas of high groundwater salinities were able to survive by removing salt from their roots and by lowering their transpiration rates, indicating that these trees responded slowly to flood events. Black box prefers soils with low suction properties, such as sandy loams, in order to reduce the energy needed for water extraction (Jolly and Walker, 1995). Jolly and Walker (1995) identified the water sources for black box trees on the Chowilla floodplain using stable isotope analysis. They identified that black box used a combination of rain water and flood water when available and could supplement these with groundwater when necessary.

Transpiration rates up to 1 mm/day have been recorded for black box, and up to 2 mm/day for river red gum. River red gum on the Chowilla floodplain occur on the margins of creeks where the flooding frequency is greater, where alternate water sources, direct from the creek or through bank recharge, are present and there are reduced soil salinities (Thorburn *et al.*, 1993). Thorburn *et al.* (1993) found that river red gum can tolerate waterlogging and some soil salinisation. The adaptive mechanism for this is to use groundwater in low salinities (Mensforth *et al.*, 1994). Mensforth *et al.* (1994) found that river red gums within 15 metres of a creek obtain creek water directly during summer and recharged soil water in winter following high creek levels. River red gums further than approximately 15 metres from the creek edge used rain water stored in the soil profile in winter and groundwater in summer in areas where groundwater salinities were less than 40 dS/m. This thesis considers these water sources in the modelling of vegetation health.

The availability of water through osmotic and matric potential within the soil is therefore taken as the key factor influencing vegetation health, with soil salinity and soil moisture as the major determinants. The major factors affecting water availability – flooding, soil structure, groundwater depth, and salinity are discussed below.

Important to the health of any vegetation community is production of viable seeds, recruitment of juveniles, establishment of reproductive mature plants and rates of natural senescence from old age. Jensen *et al.* (2008) found that in water-stressed trees seed release was up to nine times less than healthy trees and that germination requires water from floods or local rainfall. George (2004) found that the current flooding frequencies were insufficient to support wide-scale recruitment needed to improve the ratio of juveniles to adults in the population of red gum and black box trees. These overall population health factors, although likely to be affected by water availability, are not considered in this thesis, which focuses on the health of the mature plants only. Floodplain trees can live for substantial periods, with recorded ages up to 500 years

(Slavich, 1997). It is expected that by maintaining healthy adult trees the population of the species will be preserved.

Vegetation health can be affected by other factors including insect attack and disease. These factors are not considered in this thesis as they are considered to be of less importance than water availability. In addition it is not well understood how management scenarios impacting on groundwater and flows would affect these factors.

3.3.1 Flooding Frequency and Duration

An understanding of floodplain inundation is required to determine the supply of fresh water, as well as leaching of salt from the soil profile and recharging of the groundwater. Ground records of the extent of the 1956 flood were mapped by the South Australian Government to produce a 1956 flood boundary, which is often used to define the extent of the floodplain in South Australia. This flood boundary depicts the 1 in 100 year event (Ohlmeyer, 1991). Most of the floodplain is inundated approximately 1 in 13 years under natural conditions (Sharley and Huggan, 1995).

Flood timing is important for plant recruitment and is therefore required in population modelling. The focus in this study, however, is on adult tree maintenance where timing of floods is not critical. Also in the lower River Murray the timing of floods has not altered from pre-development conditions.

The frequency of flooding that any tree will receive is governed by:

- the surface elevation of the tree on the floodplain, which determines its height above river level;
- river flow, in particular the peak flow that is required to inundate the tree, referred to as the commence-to-fill flow;
- the duration of the peak flow, this affects the volume of water and the areal extent that a peak flow will inundate;

- the height of the river as determined by weirs and other control structures which can be manipulated to create the effect of higher or lower flows; and
- the related flow paths including blockages and flow channels which can change daily and be managed with control structures and channelling.

A River Murray floodplain atlas mapped the extent of the 1974/75 and 1956 floods onto orthophoto base maps at 1:50,000 scale (GHD *et al.*, 1986). This Atlas covered the river from Hume Dam to the South Australian border and therefore only covers a small portion of the study area.

Remote sensing is particularly useful for observing and monitoring flood extents as it provides basic data of wetted area economically and more efficiently than ground based methods (Whitehouse, 1989). Previous studies to determine the spatial extent of flooding have commonly involved analysis of optical satellite imagery (Usachev 1985; Walker *et al.* 1986; Townsend and Walsh, 1998, Shaikh *et al.*, 2001; Sheng *et al.*, 2001; Frazier *et al.*, 2003), radar remote sensing (Townsend and Walsh, 1998) or an integration of remote sensing and GIS (Brivio *et al.*, 2002). More detailed studies have used digital elevation models to create a floodplain surface over which inundation can be modelled (Townsend and Walsh, 1998; Cobby *et al.*, 2001; Sanders, 2007). Elevation methods are particularly useful for predictive studies of changing flow paths through the floodplain by allowing manipulation of flow barriers. However, surface modelling may not give the best representation of flood inundation extents, as there are numerous impediments and small channels across a predominantly flat floodplain. The modelling of inundation from surface elevation also requires detailed information on stage heights, backwater curves, flow impedances and roughness coefficients, as simple heights are insufficient in this dynamic environment.

Most studies that have assessed flood extents from remote sensing have used image classification procedures applied to multispectral imagery (Munyati, 2000; Abuzar and Ward, 2003). Radiation in the near and mid-infrared wavelengths is almost completely absorbed by water and hence images show sharp contrast between the high reflectance of soil and vegetation in dry areas and the very low reflectance of water (Whitehouse, 1989).

Water is detected by isolating the pixels within the image that have very low reflectance values in the mid infra-red with wavelengths between 1.55 - 1.75 micrometers (Frazier and Page, 2000). However, other features, especially shadow, can also have very low reflectance in this spectral region. Discriminating water from dark shadow is not possible using a single band image. This problem is compounded by the slightly higher reflectance of turbid or shallow water, leading to these features being misclassified as non-water. The definition of the threshold value for determining surface water in a single band image is a judgment made by the analyst based on the data characteristics. However, Frazier and Page (2000) found that a simple density slice on Band 5 (Landsat wavelengths of 1.55 to 1.75 micrometres) achieved an overall accuracy of 96.9%, when compared to aerial photographic interpretation and proved as successful in delineating water as a 6 band maximum likelihood classification. Abuzar and Ward (2003) used linear un-mixing to determine the proportion of water within each pixel from Landsat imagery. The second method was more intensive and required ground truthing sites. A density slice method was chosen for this study.

Flood maps for the Chowilla floodplain have been developed for analysis with vegetation (Noyce and Nicolson, 1993). The maps were produced from a density slice of the Landsat 5 TM near infra-red band 5, separating out areas where the ground had low infra-red reflectance indicating the presence of water or shadow. These maps were provided as raster files with a cell size of 30 m x 30 m, providing high spatial accuracy, estimated to be within 30 metres (one pixel width). This analysis resulted in the production of flood extent maps of 33,000, 42,000, 47,000, 62,000, 70,000, 77,000, 82,000, 98,000, and 101,000 ML/day. In addition the maps were useful in identifying the relationship between flooding and vegetation distribution.

Recent modelling of river flow and inundation has been undertaken for wetlands on the Darling River in New South Wales (Shaikh *et al.*, 2001), floodplains on Roanoke River in North Carolina (Townsend and Walsh 1998) and the River Murray in Australia (Frazier *et al.*, 2003; Abuzar and Ward, 2003). These studies have shown that satellite imagery is appropriate for mapping large expanses of flooding.

Floodplain inundation models have been developed for other rivers using hydrodynamic or hydraulic models built using 1, 11/2, 2 and 3D models with software packages such as Mike11 and Mike21 (Water Technology, 2005; Tuteja and Shaikh, 2009). This type of modelling is data intensive and requires significant field verification. These models also require computer processing times that make them unsuitable for real time management decisions.

There is no spatial model of floodplain inundation for the lower River Murray. This thesis explores the use of satellite image analysis of flood events to build a floodplain inundation model for the lower River Murray.

3.3.2 Soil Properties and Groundwater Recharge

Soil type determines the rate of salt accumulation, as well as the rate of recharge and leaching that the soil exhibits. Soils of the lower River Murray floodplain have been mapped previously as a single unit. Exceptions include the Chowilla floodplain which was mapped in detail for soil landscape geomorphology by Hollingsworth *et al.* (1990) which defined 24 land units. Each land unit is a unique soil association which was identified as being sandy or clayey to divide the floodplain into two soil textures. These two major soil types define the type of vegetation and the salt accumulation and leaching rates. The presence of the Coonambidgal clay layer, the sediment from which the clay soils have developed, is therefore a key factor in groundwater recharge.

An airborne electromagnetic survey was undertaken to determine the presence and thickness of this clay layer in the highlands (Munday *et al.*, 2008). A small portion of the survey covered the floodplain and was analysed successfully to determine the soil moisture, clay content and salinity of the floodplain soils on the Bookpurnong floodplain when compared to field records (Doble *et al.*, 2006). An airborne electromagnetic survey has been obtained over the Chowilla floodplain and initial results mapping soil types and recharge are presented in this thesis.

On a regional scale it is known that the floodplain intercepts the regional saline groundwater before it reaches the river (Barnett, 1989). The floodplain attenuation is estimated to be 30% of the inflow (Barnet *et al.*, 2002). The analytical cross-section model of groundwater movement through the floodplain (Holland *et al.*, 2009) has the potential to model this floodplain attenuation if applied to the whole of the lower River Murray floodplain. This thesis applies this model to map the spatial extent of floodplain attenuation for this region.

3.3.3 Groundwater Depth and Salinity

Lower River Murray

A network of piezometers across the lower River Murray monitors groundwater depth. There are large uncertainties in derived surfaces of groundwater depth and salinity from point data, given their spatial variability.

Regional scale groundwater models have been used to model the flux of water entering the river valley and the river channel using the 30% attenuation figure mentioned above. These models have been built with two dimensional finite element models of groundwater flow using software packages such as MODFLOW (McDonald and Harbaugh, 1996). These models are complex and require considerable data to develop and calibrate.

A GIS approach has been used to develop the SIMPACT model (Miles *et al.*, 2001) used to estimate the flux of groundwater entering the river valley from a range of irrigation inflows. The model can be used to examine changes in vegetation cover and irrigation drainage rates.

However, using these regional scale models, floodplain processes cannot be modelled at the resolution required to refine management strategies to the scale of the individual floodplain. The spatial resolution of such models and the lack of data required for calibrating them means that they are not capable of simulating individual irrigation developments or impacts on individual floodplains.

Floodplain groundwater level fluctuations have been simulated with a number of analytical and numerical models. Previous analytical, cross-sectional models have adequately simulated groundwater level fluctuation in response to changing river stage using the modelling package SUTRA (Narayan *et al.*, 1993; Jolly *et al.*, 1998). However, none of these models address salinisation of floodplain soils. The alternative is a highly complex, numerical spatial groundwater model, which can simulate both groundwater level fluctuations and groundwater fluxes. Armstrong *et al.* (1999) used a MODFLOW model comprising 84,390 cells in 6 layers to represent the Loxton Irrigation Area and a floodplain of approximately 5,000 ha. The model was able to simulate salt loads during floods, however, despite extensive field data and calibration, it was not able to adequately simulate groundwater interception and salt storage by the floodplain.

Generally, there is less data available for the remaining 95,000 ha of the floodplain in the lower River Murray than that for the Loxton Irrigation Area, but the area can be adequately characterised by regional scale hydrogeological data (e.g. Barnett *et al.*, 2002). The heterogeneity of the floodplain geomorphology cannot be modelled adequately with regional scale data, and so the underlying conceptual models need to be simplified.

Chowilla floodplain

Groundwater contours were estimated by the South Australian Engineering and Water Supply Department and were derived from a number of sources, including a network of 20-30 piezometers, river and anabranch level data and groundwater model interpretation. These contours were plotted as historic low levels and postulated post-flooding levels and are estimated to be accurate to within one metre. Groundwater depth has been derived by subtracting the groundwater contours from the surface topography (Noyce and Nicolson, 1993).

A groundwater salinity surface exists which consists of a shallow flushed zone close to the main channel of the river and upstream of Lock 6 (Collingham, 1990a; 1990b). This flushed zone is based on groundwater sampling and interpolation. The remaining part of the floodplain was mapped as saline. Taylor *et al.* (1996) found that the flushed zone contained groundwater salinities of 26 dS/m on average, but they were highly variable. Hodgson (1993) later added further areas of fresh groundwater (less than 40 dS/m) in recharge areas on the floodplain from analysis of piezometer readings and the construction of salinity contours. Taylor *et al.* (1996) showed a statistically significant association between the interpolated groundwater salinity surface and field values across the floodplain showing that the interpolation process was sufficiently accurate for modelling purposes.

A MODFLOW groundwater model was later developed for the Chowilla floodplain (Yan *et al.*, 2005). This model used stream heights and bore readings to calibrate the groundwater flow. Recharge rates associated with mapped soil units were linked with flood extent in the MODFLOW model.

The model was extended to predict groundwater salinity by using the MT3D solute transport model (Yan *et al.*, 2005). Modelling identified significant changes in evapotranspiration and discharge rates between pre-locking and current conditions. Pre-locking evapotranspiration rates were estimated to be 0.17 ML/day for the Chowilla floodplain as opposed to 3.03 ML/day currently. The large change in evapotranspiration rates is attributed to rising groundwater levels. Discharge of groundwater to the creeks was much higher in pre-locking times (2.36 ML/day) than it is today (0.70 ML/day). The permanent water in the outer creeks has decreased the amount of groundwater entering the creek system and has been attributed to the rise in groundwater levels.

Overton and Jolly (2003) note that the difference between the two groundwater salinity predictions for pre-locking and current conditions is most noticeable in the flush zone around the lock. In this area, the groundwater has gradually freshened as the level in the main channel is kept higher than the outer creeks by Lock 6. Freshening of the groundwater increases plant evapotranspiration (in this case river red gum forest) due to increased water availability. However the increase in evapotranspiration will eventually lead to salt accumulation, even in this predominantly fresh aquifer.

This thesis uses the MODFLOW groundwater model of Yan *et al.* (2005) to make predictions of vegetation health from current and future conditions in relation to differing management scenarios.

3.3.4 Soil Salinisation

Black box occur on the higher elevation areas of the floodplain and therefore they are infrequently flooded. This, together with low rainfall means that black box needs to be drought tolerant to survive. Black box therefore relies on groundwater for survival when flood water is not available, provided groundwater is not too saline. Where the salinity of the groundwater is too high for the black box to extract sufficient water, the trees must draw their water from the upper soil profile and become affected by salt accumulation in this capillary fringe zone. It has been found that the combined effect of salinity and reduced flooding is a major cause of dieback of black box (Jolly *et al.*, 1992b and 1993b; Eldridge *et al.*, 1993).

Soil salinisation and hence increased lack of available water has been attributed as a major cause of floodplain vegetation decline in the lower River Murray. A number of studies (Jolly *et al.*, 1994a; Jolly *et al.*, 1994b; Eldridge *et al.*,1993; Hollingsworth *et al.*, 1990; Margules *et al.*, 1990) showed that soil salinity is a major factor affecting the health of black box. The effect of accumulated salt in the soil is to increase the osmotic potential required for the floodplain vegetation to extract water from the soil. Once the salinity concentration is too high, the plants can no longer remove the water. This critical salinity varies for different species, with black box trees able to tolerate higher salinities than river red gums. Red gum tolerance is considered to be 20 dS/m (MDBC, 2003) or up to 35 dS/m at Chowilla (Overton and Jolly, 2003). The critical salinity for black box has been estimated to be 55 dS/m (Overton and Jolly, 2003). Salinity can also be toxic to the vegetation if in sufficiently high enough concentrations within the plant. Such toxicity is not considered in this thesis.

Jolly *et al.* (1992b) found that the black box were generally healthy in sites where groundwater salinity was less than 40 dS/m. Above this value, health was found to be variable. Depth to groundwater was also found to affect tree health. The study suggested that a critical depth to groundwater existed and this was dependent upon flood frequency. This was shown to be consistent with soil salinisation processes in that there is a critical depth to groundwater, at which there is no vertical movement of salt through the soil profile.

As flooding frequency increased, the critical depth to groundwater decreased. For areas that only flood at a peak flow rate of 100,000 ML/day or more, this critical depth is approximately four metres for black box (Jolly *et al.*, 1992b) and for areas that flood more frequently, it is between two metres and three metres. Most black box exist on the higher parts of the floodplain where depth to groundwater is rarely less than two metres. Black box in areas flooded by flows of less than 82,000 ML/day (1 in 10 years), in general, appear to be healthy. Taylor *et al.* (1996) observed that the black box on the dunes with sandy soils on the

Chowilla floodplain appear healthy, though not tall, in contrast to those on the clay which appear taller but less healthy. Jolly *et al.* (1993b) showed that the critical depth of groundwater is deeper for clay than for the coarser sands, hence trees situated on the sands would have more available fresh water in the soil profile. Threshold levels for river red gum are less understood. From observations of river red gum and black box health in areas of high salinity and low flooding frequency, it is suggested that river red gum has a lower salt tolerance and a requirement for more frequent flooding than the black box. (Holland *et al.*, 2010)

Salt accumulates on the floodplain due to capillary rise through soil profiles. This involves shallow groundwater being drawn up and either evaporated at the surface, or transpired by the trees. Salt is left behind in the upper soil layers where it slowly accumulates. Studies have indicated that there is very little leaching of this salt under the current flooding regime (Jolly *et al.*, 1994a; Akeroyd *et al.*, 1998). Before regulation, the upper part of the soil profile was leached free of salt by floods that came approximately every four years. Since regulation, however, the frequency of floods of sufficient size to carry out freshening has been greatly reduced. Secondly, locking has led to rises in groundwater, which have accelerated groundwater discharge rates and hence salt accumulation. For example, a soil profile that previously would have taken 20 years to reach a level of salinity that is detrimental to black box health for a groundwater depth of 4 metres would take 5-7 years for a depth of 3 metres (Jolly *et al.*, 1993a).

A number of factors affect the extent of soil salinisation, including the salinity of the groundwater, groundwater depth and soil texture. All these factors vary across the floodplain. The spatial variation in groundwater depth and flooding frequency are of particular interest as these affect the balance in water movement through the soil profile and are affected by flow management and river regulation.

Groundwater discharge rates are controlled by limiting soil physical conditions and the evapotranspiration rate of the vegetation. Soil-limited groundwater discharge rates are determined by the water table depth and the soil hydraulic properties. For steady state conditions the maximum steady upward evaporative discharge flux (q_m) that can be sustained from a water table at a depth (d) is dependent on the soil texture (A) and (n) given in Equation 2.1 (Gardner, 1958; Thorburn *et al.*, 1993).

$$q_m = A\, d^{-n} \qquad \text{(Equation 3.1)}$$

Groundwater discharge rates are lowest for sandy soils with deep water tables and highest for heavy clay soils with shallow water tables (Jolly, 2004). Salt accumulation rates are dependent on groundwater discharge rates and groundwater salinity.

This change in the balance between shallower groundwater, increased salt accumulation and less flooding has shifted from pre-regulation to current conditions (Figure 3.7). This has combined to create reduced water availability in the soil profile.

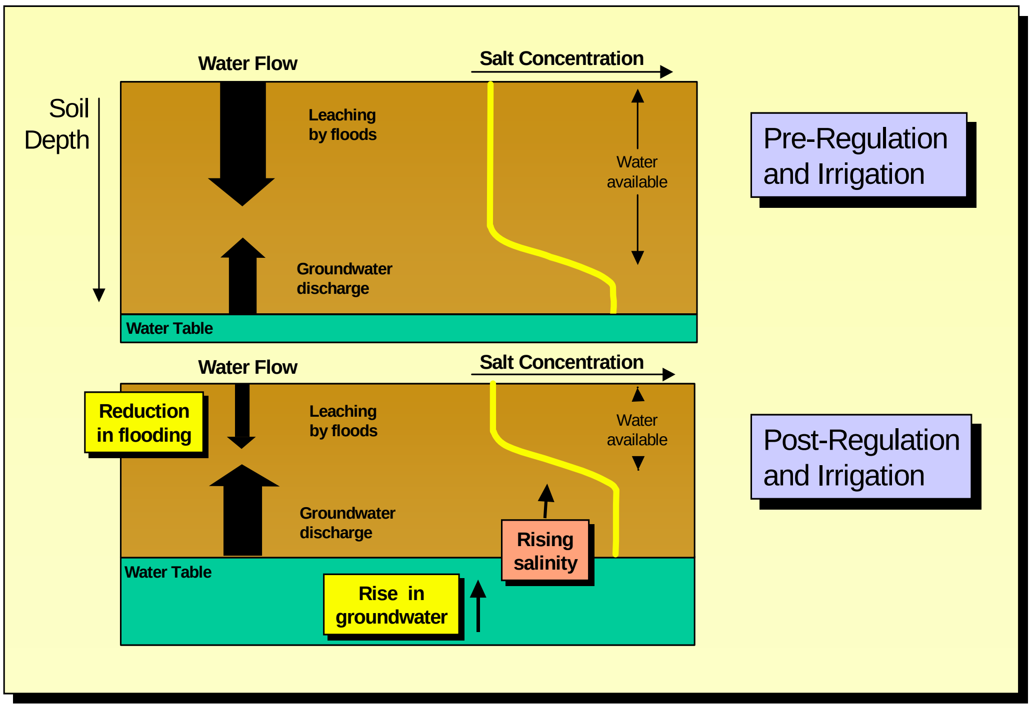

Figure 3.7 Conceptual diagram of the salt accumulation mechanisms in floodplain soils.

3.3.5 Freshwater Sources

Floodplain trees utilise the water within the unsaturated zone of the soil profile. This water is typically replenished through rainfall or flooding. Other water sources exist which become critical to the trees during drought periods. Red gums have been found to be able to use creek water directly when their roots are positioned into an adjacent creek (Mensforth *et al.*, 1994). For those trees without roots directly into the water the adjacent creek provides a water source when water moves into the banks through lateral recharge. Losing creeks, where water moves into the banks, occur when the adjacent groundwater level is lower than the creek level. The major example of this is the flush zone around the River Murray upstream of a weir. The flush zone at Chowilla was first mapped by Collingham (1990a). This is common occurrence on minor creeks as well with recharge occurring for approximately 50 metres. Holland *et al.* (2010) identified this process at Bookpurnong where the lateral recharge was created by pumping the adjacent groundwater.

Where the Coonambidgal clay is absent from the floodplain, flood waters can recharge the unsaturated zone and can remain on top of the saline aquifer due to density differences with the fresh recharged water. This process is called a freshwater lens and has been observed to support red gum trees in areas above their flooding frequencies (Seekamp, pers. comm., 2000). Other small areas of holes in the Coonambidgal clay can create pockets for recharge. Local areas may also receive more water by a concentration of rainfall run-off. This can be seen with the improved vegetation health of trees at the break of slope on the edge of the floodplain.

3.4 MODELLING FLOODPLAIN TREE HEALTH

3.4.1 Introduction

Identifying water requirements for sustainable ecosystem health is a challenging exercise. The 'natural flow regime' concept identifies the need for the natural components of the flow regime such as variability of magnitude, frequency, timing, duration, rate of change and predictability of flow events, to be reinstated. However, this can be a difficult task within the MDB due to the scarcity of water and the regulation controls over much of the river flows. It is also difficult to identify reference rivers within the MDB for ideal ecosystem conditions and processes. Management of the river has relied on the 'minimum' flows concepts to sustain important components of the ecosystem. This is an inadequate method of determining environmental flows for the long term sustainability of the system. Arthington (1998) called for a holistic approach to environmental flow assessment techniques. The drought conditions add another challenge to environmental flow assessment, with prioritisation given to those ecosystems under immediate threat, so resilience and response to watering need to be assessed.

Ecological responses of primary production, fish, birds and amphibians to water have been modelled at a range of temporal scales at many locations across the MDB. These models describe floodplain ecosystem function and changes of state resulting from altered hydrological regimes and colonisation by invasive species. Along with empirical data on ecological response to flows there is also a range of expert knowledge.

Conceptual models are often the first step in articulating an understanding of the ecosystem functions and processes and the major drivers, they are widely used in environmental studies and are often the basis of more advanced modelling techniques.

To apply these ecological responses for individual ecosystem components to an assessment of flow regimes for a whole ecosystem, a number of approaches are commonly used. The variety of different ecosystem response models can be categorised in numerous ways. One method could separate:

- Flow response models using single or multiple flow variables;
- Habitat suitability models that are usually spatially and temporally explicit;
- Process based models including physio-chemical drivers; and
- Population models for single species or food web models for complex species interactions;

These categories are discussed in further detail below.

3.4.2 Flow Response Models

Flow response models use a relationship between hydrology and ecology that has been determined by observation from change in one river or by comparison between two rivers.

Broad-scale flow response models identify components of a hydrograph that are needed for ecosystem function – an example of this is the Ecological Limits of Hydrologic Alteration (ELOHA) (Poff *et al.*, 2010). This incorporates the 'building block' method and the 'natural flow regime' concept (Figure 3.8). It is useful in in-channel environments where the hydrograph for the area in question can be tightly defined. It also relies on the ability of mangers to be able to control the desired aspects of the hydrograph that are identified. The natural flow paradigm has limited application in highly regulated rivers in resource limited environments. The River Murray has been altered to such a degree and for such a long time that consideration of 'natural' or a percentage of natural may no longer be useful. Flow rules are also limited in rivers that have very high variability. It is difficult to plan and make decisions over large time scales that can operate over decades.

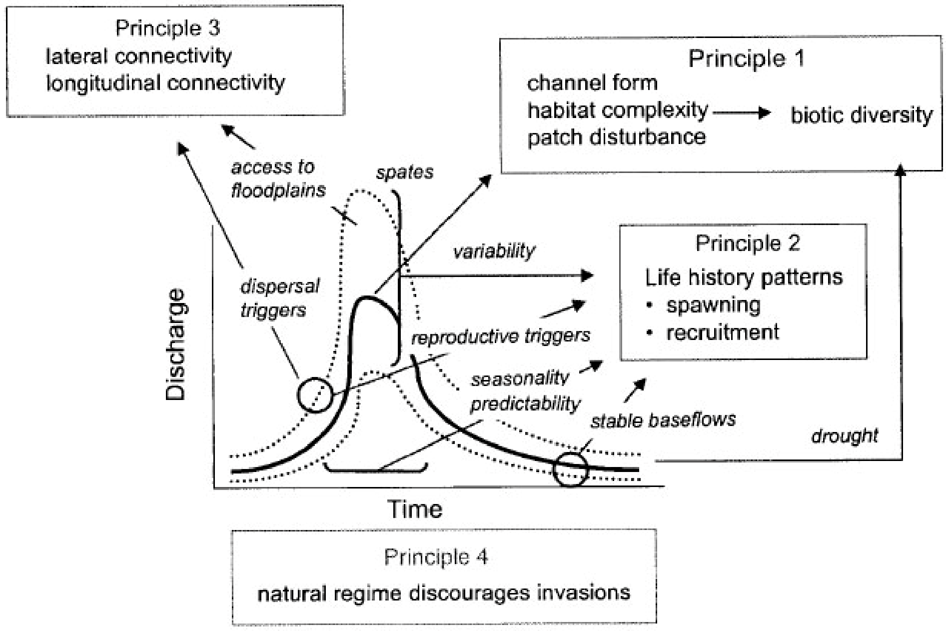

Figure 3.8 Conceptual model of the natural flow regime method (Bunn and Arthington, 2002).

Single response curves relate a component of the hydrology to a component of the ecology. For example it could relate the duration of inundation to the survival of an adult tree. These response curves are typically derived empirically and identify a suite of response relationships between flow variables on ecosystem components – an example of this is the Murray Flow Assessment Tool (MFAT) (Young *et al.*, 2003) which was used to identify the impacts of proposed environmental water allocations on the Icon sites in the River Murray (Figure 3.9). This approach allows incorporation of a range of ecosystem components when knowledge is available and provides the ability to assess the 'condition' of those components from a given flow regime. Response curves can be more useful in identifying small changes but may fail to identify ecosystem changes as a result of hydrological or ecological interaction.

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Figure 3.9 Empirical model of river red gum response to flooding relating health to inundation duration (Young *et al.*, 2003).

In the case of the MFAT model, it was not applied spatially so is limited in its application to the complex floodplain environment. More complex flow response models have been developed for the Northern MDB wetlands and flooding extent can be related to the flow requirements of key ecosystem components such as bird breeding, fish passage, aquatic vegetation biodiversity.

Other empirical models have been built on components of flow regimes such as connectivity of the wetland or floodplain area to the main river channel, such as the percentage of time connected, rather than direct components of the river flow. Ganf *et al.* (2010) developed a relationship between connectivity and probability of presence of aquatic macrophyte species in wetland of the River Murray (Figure 3.10).

Flow response models have limited use in modelling floodplain tree health due to the spatial complexity of water availability as a result of surface water flooding and groundwater variability. The interaction between flooding and groundwater is difficult to model in a flow response model.

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Figure 3.10 Probability of presence for several wetland species across a gradient in wetland connectivity recorded as the percentage of time the wetland is connected to the river channel (Ganf *et al.*, 2010)

3.4.3 Habitat Suitability Models

Habitat suitability models are usually rule-based and are used to identify a range of possible habitat outcomes given a set of rules. This approach is useful for considering alternative states of ecosystem condition given the response of the ecosystem to a range of variables. Rule based models are usually the next step in sophistication from conceptual models. Multiple rules can be used and habitat suitability models can be spatially and temporally explicit.

An example of a simple habitat suitability model in the lower River Murray is the River Murray Floodplain Inundation Model developed in this study. The model is a flow response model using two flow variables, frequency and duration, but spatially explicit. The model allows predictions of the extent and depth of a range of flow regimes and it has been used to assess the water requirements to sustain river red gum communities along the River Murray floodplain in Victoria (Overton and Doody, 2007). The results showed that 74% of the floodplain area was at high risk of flooding frequencies below those that would sustain healthy tree populations. Volumes of annual flows were calculated based on health indices for a range of flooding frequencies.

A number of studies have used GIS to identify the distribution of areas of environmental factors that pose a risk to vegetation. Miller *et al.* (1995) identified areas on a floodplain in Wyoming, USA, using aerial photography and flood history data and linked these areas with measures of landscape structure. Changes in vegetation patterns identified from Landsat Multi-Spectral Scanner imagery were also found to be correlated with changes in groundwater levels, temperature and rainfall in Arizona (Lee and Marsh, 1995). Both of these studies were able to predict the changes in habitat as a result of changes in hydrology within the confines of known habitat tolerances. Once the hydrology changes beyond known boundaries then the prediction of habitat cannot be validated.

A GIS-based methodology to assess the impact of river regulation, reduced flooding frequency and irrigation development on floodplains in the lower River Murray was developed by Gabrovsek *et al.* (2002). This analysis was based on three key risk factors of depth to groundwater from irrigation, depth to groundwater from locking and flooding frequency over the Lock 4 to Lock 3 reach. The results indicated that river red gums were less tolerant of all three salinisation factors compared to black box, and that declining river red gum health is mainly due to reduced flooding.

Gabrovsek *et al.* (2002) determined flooding frequency using the floodplain inundation model (RiM-FIM) developed in this thesis (Chapter 5) and areas above 70,000 ML/day were considered to be at risk from reduced flood frequency. Depth to groundwater from locking was determined from intersecting the River Murray height at pool level (entitlement flow) with the pseudo-elevation model developed from the floodplain inundation model (Chapter 5). Depth to groundwater from irrigation was more complex and was derived from the slope of the groundwater from the edge of the irrigation area to the edge of the floodplain and comparing this to the slope of the groundwater from the edge of the floodplain to the river. The floodplain was divided into 85 divisions for the Lock 4 to Lock 3 reach based on changes to groundwater contours in the highlands. The shape of the divisions on the floodplain was derived from the shortest path of the edge of the floodplain to the river.

The first attempt to model black box health on a floodplain scale considered the use of the critical threshold values of the main factors which affect black box health (Noyce and Nicolson, 1993). The results concluded that the spatial distribution of health patterns can be modelled successfully using a small number of variables within the GIS. Hodgson (1993) defined six GIS classes of black box health using the parameters of groundwater salinity, flooding extent and groundwater depth. This model was later used to explain broad scale spatial patterns in the health of black box (Taylor *et al.*, 1996) (Figure 3.11). Comparisons between aerial photographs and GIS classes were used to quantify the association. While the GIS class model was useful for vegetation mapping, it was less useful for predicting the effects of changes in management, as the classes are relatively broad and do not give any indication of the degree of health. This thesis further develops the use of GIS to model salt accumulation and vegetation health and model the effects of manipulating groundwater and flooding with this class model approach.

Empirical models may provide strong relationships between modelled and observed data. However, they are limited by their inability to model environmental changes outside the range of observed conditions. Process-based models allow predictions for an infinite range of conditions depending on the assumptions made in the model development.

Lester and Fairweather (2008) provide an example of a rule-based model which uses an adaptation of alternative stable state theory to model the potential future outcomes of the Coorong under different management and climate scenarios (Figure 3.12). Rather than placing emphasis on stability the model included transient states to model the non-equilibrium conditions that occur during the decline of an ecosystem. "*This, combined with a data-derived multivariate state-and-transition model, provides the framework for construction of the ecological response model*" (Lester and Fairweather, 2008).

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Figure 3.11 The Chowilla GIS showing good and poor black box health using the class model (based on Taylor *et al.*, (1996)).

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Figure 3.12 Rule based model showing alternative states (Lester and Fairweather, 2008).

3.4.4 Process-Based Models

Process-based models are those that model the response of particular ecosystem components to progressive changes in physical conditions such as climate, hydrology, geomorphology or biochemistry. These models incorporate, or are built on, understanding of the physical/ecological processes, versus empirical only relationships, and attempt to represent those processes and use them to model changes in state/condition. The advantage of process models is that they can be implemented spatially to model the whole environment in its complexity of habitats.

Empirical models are limited by their ability to show responses to small changes to the environmental factors. Process models are more complex but can provide greater control over the results and also provide a better understanding of the factors affecting salinisation.

When the major environmental factors affecting riparian vegetation health are known, a spatial model can be developed to examine the interaction of these factors. For example, Swetnam *et al.* (1998) linked a GIS model to a hydrological model of the Southlake Moor in Somerset, UK, along with plant-water regime requirements. This model was capable of determining the impact of river management on the health of associated riparian vegetation.

This study has adopted the development of process-based models where possible to predict tree responses to management options. To do this it builds on an analytical cross-section model for groundwater management at a regional scale and a vegetation growth model that are incorporated into a simplified spatial salinisation model for the floodplain scale.

Analytical Cross Section Model

Given the difficulties related to mapping groundwater depth in the floodplain, Holland *et al.* (2009) developed an analytical model that would predict the impact of groundwater flow from regional and irrigation induced discharge to the floodplain. The model can provide similar results to those of more complex numerical models. The advantage of the analytical model is that relationships between floodplain variables such as floodplain width, river height, hydraulic conductivity and floodplain interception can be developed. Not only were these relationships useful by themselves, but they could be implemented in a GIS framework to provide objective analyses of salinity risk at a sub-regional scale.

This thesis applies the analytical cross section model spatially to the lower River Murray floodplain, described in more detail in Chapter 4.

Vegetation Growth Models

A number of vegetation growth models have been developed in the past and many are still currently in use. Process-based models tend to be developed for specific purposes and focus on particular factors of interest in vegetation growth. Models that have been developed for Eucalypt forests include a number of carbon growth models: - 3PG (Landsberg and Waring, 1997), Cabala (Battaglia *et al.*, 2004), and WAVES (Slavich *et al.*, 1999a). In most cases these vegetation growth models are too complicated to be populated and run for multiple management decisions.

Salinisation Models

To obtain a more process-oriented model for tree health, consideration must be given to the processes of soil salinisation and their impact on plant health. While the salt front within the profile of a given soil may vary throughout the year in response to flooding, rainfall, water extraction by vegetation and groundwater fluctuations, over the long term a dynamic salt 'balance' generally exists, in which there is no net accumulation or leaching of salt (Jolly *et al.*, 1993a). This balance concept has been developed into a tree health model called the Weighted Index of Salinisation (WINDS) model by Slavich *et al.* (1999a). Salinisation models are described in more detail in Chapter 5.

3.4.5 Population and Food Web Models

Population models consider the life stages of an organism and model the population over time. Dependencies on recruitment rates drive these models over time. Ecosystem response models can also consider more than one species and the interactions between these such as predator prey or food webs can be modelled. In this study there were considered no critical species interactions with floodplain trees to warrant the development of a food web model. The development of a population model for floodplain trees has been attempted (Langridge *et al.*, 2006) to model tree population on the Lower Goulburn and Barmah Millewa Forests. Little empirical data on the life cycle dynamics and their salinity response and water requirements is available to develop and validate a population model of floodplain trees on the lower River Murray.

3.5 MANAGEMENT DECISION SUPPORT

3.5.1 Regional Management Scenarios

Environmental flow allocations in the early 1990s were "effectively ad hoc" (Walker *et al.*, 1995) because of the lack of scientific evidence linking ecological response to flow regimes. In 2000, environmental flow management was still based on 'best informed guesses' (Thoms *et al.*, 2000), rather than hard data. Thoms *et al.* (2000) identified a series of management recommendations for environmental flows along the River Murray based on an assessment of hydrological change from historic pre-regulation conditions and key hydrological components of the flow regime to support ecosystem processes.

Flow management in the MDB is currently governed by a complex system of water allocation rules that are regulated by State operations. A cap exists that provides for a minimum entitlement flow into the lower River Murray in South Australia. During the drought period from 2001 the river flow was less than this entitlement. The vested interests of the individual states in the control of the water within the MDB is seen as one of the major issues of water management. The transboundary issues of water management and the perceived over-allocation of the water for irrigation prompted the Australian Government to develop the Water Act (AG, 2007) which established the MDBA and instigated the Basin Plan.

The release of environmental water to improve the ecosystem is one management option in the lower River Murray. The Commonwealth Environmental Water Holder and the State Government of South Australia hold licenses for environmental water. The MDBC's Living Murray Initiative has set targets for maintaining the health of the floodplain vegetation and has instigated the 500 GL/yr initiative. This initiative plans to release an extra 500 GL per year into the river for environmental purposes. An amount of 500 GL was determined as a trade-off between what the ecosystem needed and socio-economic impacts. The Draft Basin Plan has identified that further water be released for environmental benefits.

The lower River Murray is highly regulated by its weirs. The major management objective in the past has been to keep upper pool levels as constant as possible (Brenton Erdmann *pers. comm.*, 2004). Using the weirs to manipulate water levels to create floods and to fluctuate levels in a more natural way are two management approaches available. Flows of greater than 35,000 ML/day are required to raise water above the channel banks and onto the floodplain. This can be achieved at lower flows using the weirs. Fluctuating river levels in the river channel without overbank flooding is still an important mechanism for recharging bank aquifers. Guidelines have been developed for the management of flows in the region.

An Environmental Flows Decision Support System (Young *et al.*, 2000) has been developed to examine a number of river environmental factors such as area of wetland and aquatic and terrestrial fauna. This model has been developed into the Murray Flow Assessment Tool (Young *et al.*, 2003). Neither model, however, considers the extent of varying flood extents, treating floodplains as one unit with a commence-to-fill flow and a rate of fill. These models do not consider salinity or the effect of changing the depth to groundwater.

The lower River Murray contributes the majority of the salt that is registered at Morgan, the point at which salinity targets are monitored. A number of salt interception schemes have been built and more are proposed to manage the salt inputs. Traditionally policy for surface water allocations and groundwater have been separate. The Basin Plan is being designed to incorporate both of these and link river salinity with environmental health. Hence there is a need for decision support to integrate surface and groundwater management options.

3.5.2 Floodplain Management Scenarios

Flow management for local floodplains involves river releases, local flow control structures and artificial irrigation. It is expected that the enhancement of flooding frequency will generally improve vegetation health. The management of environmental flows needs to consider that flood duration can have a major influence on the ecological benefit derived from the flood event. Little benefit may be gained unless thresholds are reached. Hence, it may be better to flood a smaller area for a longer time than to cover the greatest possible area. These thresholds will vary depending on species, soil type, soil salinity and groundwater depth and salinity. Vegetation health is not the only concern for flow management and trade-offs may be required against other issues relating to environment and infrastructure (e.g. high salt loads to the river).

The quantity, frequency, distribution and timing of environmental flows are questions not easily answered in managing stressed riparian vegetation with limited resources. Prioritisation of floodplain areas needs to be determined, to best utilise what water is available.

Moving available water around the floodplain is another management strategy. Local control structures can be used to retain water on the floodplain before releasing it downstream. An environmental regulator for the Chowilla floodplain has been designed and construction began in 2010. The cost/benefit case for the construction of the regulator used results from this thesis to show the environmental benefits of managing the regulator. Water quality needs to be considered when retaining or releasing water, including increased salinity, risk of cyanobacteria and black water events. Fish passage is also a major issue with new control structures.

Even flow control structures are unlikely to get water to high parts of the floodplain and in the frequencies required. Artificial irrigation of floodplain trees and pumping into wetlands are further management scenarios. Pumping water into wetlands can achieve inundation frequencies and durations close to those of natural conditions, but this strategy does not produce the nutrient cycling, seed and larvae distributions associated with real floods that are connected to the river and dissipate over the floodplain.

A salt interception scheme for the Chowilla floodplain has been designed (Howe *et al.*, 2007) using bores targeted at salt removal and extra bores to target environmental benefit. The location of these floodplain bores has been driven by work undertaken for this thesis.

3.6 CONCLUSIONS

The role of soil salinisation in vegetation health is clearly understood from previous research by Jolly *et al.* (1993a). Taylor *et al.* (1996), Hodgson (1993) and Gabrovsek *et al.* (2002) have used GIS layers to successfully map risk areas on the floodplain using class models. The processes of soil salinisation are also well understood and have been modelled in a point-based model (WAVES). Slavich (1995) developed the concept of a quasi-steady state Moving Salt Front model and developed this into a WINDS model. It was postulated that this approach could be used in the spatial grid model for predicting salinisation and vegetation health over time. No spatial model of soil salinisation has been developed for the floodplain that can be used in assisting management decisions.

On a regional scale, the vegetation map of the lower River Murray floodplain that was available before the Smith and Kenny (2005) survey contained spatial and attribute errors and was of an insufficient scale for use in modelling assessment. Much of the research for this thesis was undertaken before the DEH map was available.

Table 3.1 summarises the current ability to predict tree health outcomes from management scenarios. Although an understanding of the factors that impact on tree health are known, these have not been implemented into decision support tools.

Little is known about the extent of flooding in the lower River Murray. Very little is known about the soil types and recharge potential. An analytical cross-section model has been developed to examine groundwater movement under the floodplain but no groundwater salinity surface has been developed. Given the lack of spatial data, it is necessary to model the regional system using a simplified method.

Table 3.1 Existing tools for assessing regional and floodplain scale surface and groundwater management scenarios.

	Regional Scale	Floodplain Scale
Surface Water Flow Impacts	A river flow model existed that could turn flow into river height. No floodplain inundation model. Limited understanding of water movement onto the floodplain.	Hydrodynamic model for the Chowilla floodplain
Groundwater and Salinisation Impacts	No model of salinity impacts at a regional scale	Groundwater Depth MODFLOW Model, complex model that does not integrate surface outcomes
Floodplain Tree Health Impacts	Simplified MFAT model for tree outcome at a regional scale (not spatially explicit)	No predictive model for tree health at a floodplain scale

For the Chowilla floodplain, a considerable amount of information is available including vegetation mapping, some flood maps, good soil data, a groundwater depth model and a groundwater salinity model. It was possible to model the water availability on this floodplain in great detail for use in assessing management scenarios.

The objective of this chapter was to understand the processes that affect vegetation health from the major environmental factors of surface flows, groundwater levels and surface infrastructure, how vegetation health has been modelled previously and how management scenarios have been assessed. Literature on the occurrence, causes and management of salt affected and stressed floodplains was reviewed for the major factors affecting tree health and the major modelling methods to assess the benefits of management scenarios. The major factors have been determined to be flooding frequency, soil salinity and soil water availability. The major existing modelling techniques have been identified as a creek level flow model for the Chowilla floodplain, an analytical groundwater discharge model and a point scale process model for assessing tree health. The research presented in this thesis enhances these three models to develop predictive capability for floodplain inundation, groundwater impact and tree response to changing surface and groundwater conditions.

The chapter has concluded that a spatially explicit model is required to predict tree health on the regional and floodplain scale. This model or models are required to integrate surface and groundwater management options and should model the processes of soil salinisation as this is the major driver for vegetation health and is a complex interaction of both surface and groundwater conditions.

The following three chapters describe the methodology and results for each of these model inputs.

4 ASSESSING REGIONAL SCALE ENVIRONMENTAL FLOW AND GROUNDWATER MANAGEMENT SCENARIOS

4.1 INTRODUCTION

This chapter presents the development of spatial methods to assess regional scale environmental flow and groundwater management scenarios for floodplain tree health risk on saline semi-arid floodplains.

In Australia management policy at this regional scale separates environmental flow and groundwater management activities. This is due to the complexity of administration of management policy over large regions, and also because the interactions of surface water and groundwater are not well understood and are difficult to model across the whole region.

Aims of the research were to develop models to be used to predict tree health outcomes at the regional scale from both surface water management options that include releasing environmental flows and operating the existing weirs, and groundwater management options that include groundwater pumping and increasing irrigation efficiencies.

The chapter is structured in the following way:

1. it first identifies a tree health map that is used for the regional scale model validation (Section 4.2). The mapping was initially undertaken using remote sensing with satellite imagery, however during the research a new tree health map became available at the regional scale;

2. it then develops a regional scale model of surface water management options (Section 4.3). This development required the identification of flooding extent from a range of river flows and weir levels. It then incorporated these events into a predictive model of flood extent. This model is then used as a

habitat risk model for tree health in the lower River Murray; and then

3. it develops a model to predict outcomes from regional scale groundwater management options (Section 4.4). This model predicts the risk to floodplain tree health from changing groundwater inflows and levels. It is also able to predict changes in river salinity as a result of groundwater management options.

4.2 MAPPING FLOODPLAIN TREE HEALTH AT THE REGIONAL SCALE

Vegetation type and health maps existing at the commencement of the research were not sufficient to be used for model validation. To produce a current vegetation map that recorded tree health, remote sensing using satellite imagery was trialled, given its low cost methods for rapidly and repeatedly mapping large areas. A number of challenges were identified including scale differences in tree density and image pixel size, confusion between tree canopy health and vigour of all vegetation in the image pixel, and calibration of images across different dates. A high spatial and spectral resolution scanner (Compact Airborne Spectrographic Imager, CASI) was flown over a portion of the floodplain to investigate the use of this type of imagery for tree health mapping. The very high spatial resolution meant that the ground resolution was smaller than a single tree canopy and this led to a range of new problems. The multiple narrow bands brought more challenges of interpretation for tree health.

It was necessary to identify historic health changes since river regulation in the 1920s to allow temporal comparisons of predictive model outputs. Floodplain tree health mapping was available from 1945 for parts of the floodplain (Telfer *et al.*, 1998; Telfer and Overton, 1999a; Cooling and Overton, 1999; Telfer *et al.*, 2000). This mapping was used to develop the 'flood index' model of floodplain tree health in Section 4.3.

In 2003 the South Australian Department of Environment and Heritage undertook a field survey of the River Murray floodplain and produced a floristic and structural vegetation map that included tree health classification of good, poor and dead based on a modified Eldridge approach (modified from Grimes) (Smith and Kenny, 2005). This vegetation classification and mapping was undertaken at 1:10,000 scale and was therefore used for comparing model outputs with the flow management and groundwater management scenario models in this chapter.

4.3 ASSESSING ENVIRONMENTAL FLOW SCENARIOS

4.3.1 Introduction

The assessment of vegetation health responses to different management scenarios in the lower River Murray requires the ability to define and predict the spatial extent of flooding from changing flows and the operation of flow control structures in the river.

The objective of this section is to develop a method to assess the impact on floodplain trees from environmental flow management scenarios. The method needs to identify the extent of floodplain inundation from surface water management scenarios, at the regional scale, and needs to include the ability to model the impacts of increased flows and weir manipulation in the lower River Murray.

This chapter describes the development of a River Murray floodplain inundation model (RiM-FIM) using a GIS, remote sensing techniques and hydrological modelling for the 600 kilometre long and one to five kilometre wide portion of the River Murray in South Australia from the New South Wales border to Lake Alexandrina.

Components of the methodology, results and discussion of this section have been presented in the following publications:

Overton, I.C. (2005). 'Modelling Floodplain Inundation on a Regulated River, South Australia'. *River Research and Applications* 21: 991-1001.

Overton, I.C., McEwan, K., Gabrovsek, C. and Sherrah, J. (2006). 'The River Murray Floodplain Inundation Model – Hume Dam to Wellington (RiM-FIM'. CSIRO Water for a Healthy Country National Research Flagship, Technical Report, Adelaide.

Overton, I.C., Penton, D., Gallant, J. and Austin, J. (2007). 'Determining Environmental Flows for Vegetation Water Requirements on the River Murray Floodplain'. Proceedings of the International Conference on Environmental Flows, September 2007, Brisbane.

Overton, I.C. and Doody, T.M. (2008). 'Ecosystem Changes on the River Murray Floodplain over the Last 100 Years and Predictions of Climate Change'. Proceedings of the International Conference on HydroChange, October 2008, Kyoto.

Overton, I.C., Penton, D. and Doody, T.M. (2010). 'Ecosystem Response Modelling in the River Murray'. In: Saintilan, N. and Overton, I.C. (eds.) 'Ecosystem Response Modelling in the Murray-Darling Basin'. CSIRO Publishing, Canberra.

4.3.2 Flood Extent Mapping for the River Murray

The floodplain inundation model was developed using a spatial information system framework that allowed the integration of non-spatial river flow models with the extent of floodplain inundation (Overton, 2005). Spatial decision support is one of the main roles of GIS which provides an excellent framework for the integration of multi-criterion evaluation results (Taylor *et al.*, 1996; Jankowski *et al.*, 2001). Riverine ecosystems benefit from spatial analysis studies because they encompass three important temporally dynamic spatial dimensions (channel length, river height and floodplain width) and the physical and ecological processes taking place are complex. Much work has been done on the longitudinal (upstream and downstream) dimension, examining the ecological impacts of river regulation on native flora, fauna and the physical changes occurring in the littoral zone (Stanford, 1998). However, long term modification of flow rates, changes to the frequency of flooding events and alteration to the timing of flows have now been identified as causing degradation beyond the littoral zone into both the lateral and vertical dimensions (Young, 2001).

Previous models of river flow and inundation have been developed for wetlands on the Darling River in New South Wales (Shaikh *et al.*, 2001) and floodplains on Roanoke River in North Carolina (Townsend and Walsh, 1998) but no predictive model for the lower River Murray existed.

The previous information available to develop the floodplain inundation model included nine flood masks for the Chowilla floodplain for a range of flow magnitudes (Sharley and Huggan, 1995). A floodplain boundary and the extent of permanent water in the river channel and wetlands were also available, along with the extent of the 1956 flood (the largest on record) at approximately 258,000 ML/day, which covered what we now define as the floodplain boundary.

Floodplain inundation extents were developed from satellite imagery for a range of flows, interpolated to model flood behaviour and linked to a hydrological model of the river. Figure 4.1 shows a schematic flow diagram of the process involved which is described in detail below. Spatial decision support systems and the integration of GIS and process models is suggested as a tool for the future of information integration in the environmental sciences (Taylor, 1993; Hendriks and Dirk, 2000).

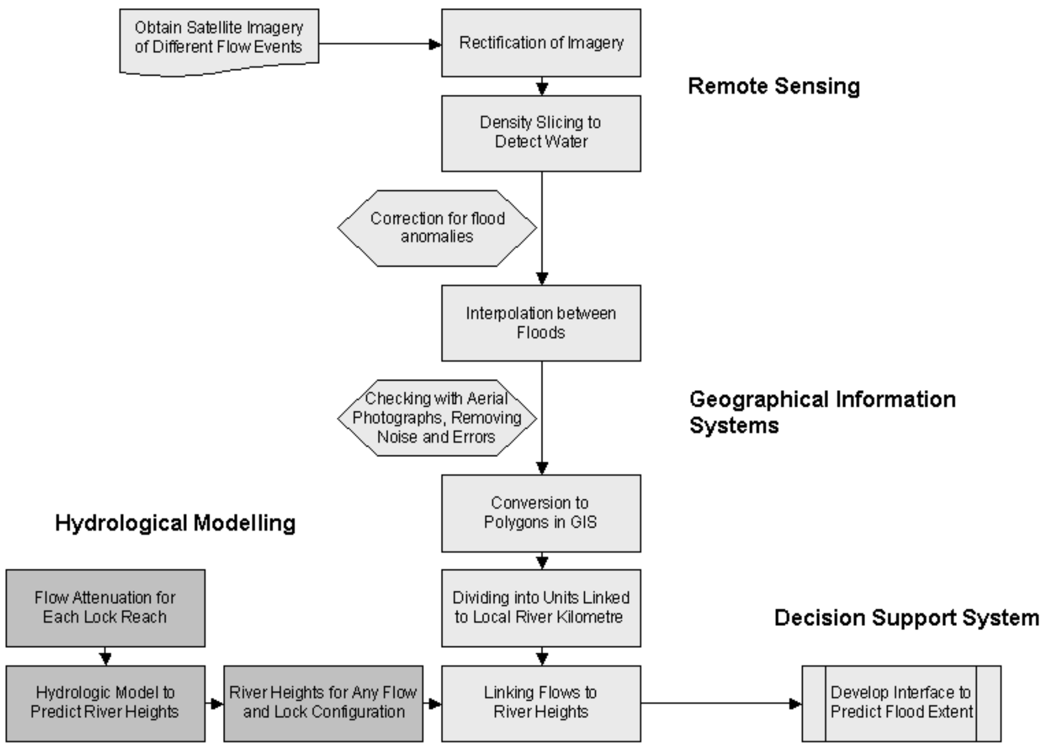


Figure 4.1 Flow diagram of the process of building the RiM-FIM.

The river exists as a series of pools under most conditions because of the weirs in this region. Floods can rise and fall in days or weeks and last for several weeks to months, meaning that the extent of flooding for a given flow has usually extended to its full potential extent before the flood peak passes. This type of regulated river is therefore modelled sufficiently from images of various events. Upstream in the unregulated reaches of the River Murray, stream flooding is very rapid and sporadic and two flood peaks of equal magnitude are unlikely to create the same extent of inundation (Frazier *et al.*, 2003).

Floodplains along the lower River Murray range from gorges below Overland Corner (Figure 1.1), 2-3km wide and 30-40m deep to valleys 5-10km wide (Walker and Thoms, 1993).

Satellite Image Processing

Satellite imagery (Landsat Thematic Mapper, TM) was obtained for this region at dates corresponding to a range of magnitudes of flood events. Correct interpretation of this data provides information on the spatial extents of flooding. Twenty one satellite images (Table 4.1) were chosen to correspond to a range of flow values as recorded at the gauging station near the New South Wales border (Gauging Station 426200). The flows ranged from 3,460 ML/day to 101,900 ML/day with dates ranging from October 1986 to November 1996. The images were chosen to capture periods of rising hydrographs (increasing flow in the river) to reduce the potential for remnant water from higher flows still present on the floodplain with flood edges being harder to delineate with a receding flood boundary. Rainfall events were not taken into consideration in choosing the image dates as surface water not connected to the river was filtered out at a later stage.

Surface water was detected using a density-slice (threshold cut-off) on the infrared band (band 7 - 2.08 - 2.35 micrometres)(Figure 4.2). Despite individual pixel mis-classification, the method of density slicing a single mid-infrared band to detect surface water has been used successfully in previous studies (Sheng *et al.*, 2001) and represents an economical method for determining flooding over large areas. A different threshold was required for each image as the images where not radiometrically calibrated.

Table 4.1: Image Path / Row and dates for the 21 satellite Landsat TM images used in the study. All images were from Row 84.

Flow (GL/day)	Path 97 / Row 84		Path 96 / Row 84		Path 95 / Row 84	
	Date	ML/day	Date	ML/day	Date	ML/day
<20	09/11/94	6,453	30/08/94	3,460	09/11/94	6,122
20-30	20/05/89	26,660	20/11/89	25,765	08/05/90	27,510
30-40	10/08/96	37,750	11/10/92	38,150	12/08/96	37,995
40-45	25/07/90	40,625	27/10/86	43,775		
45-50			03/08/90	46,695	21/09/93	47,415
50-60	20/09/96	57,025	02/09/95	55,050		
60-70	23/11/96	68,255	23/11/96	68,255	07/10/93	66,440
70-75			31/07/89	70,520		
75-80	16/08/89	78,085	16/08/89	77,420		
80-100	20/09/90	93,450	03/10/89	82,350	26/09/89	80,170
100-110	22/11/93	109,868	26/10/90	101,565	29/09/90	101,900

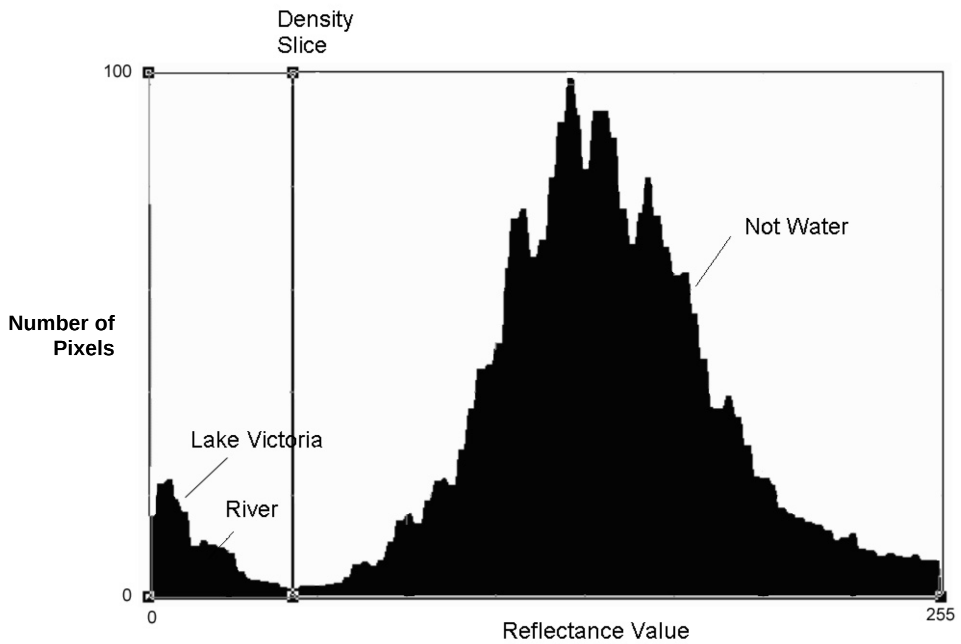


Figure 4.2 Histogram of band 7 of the image that relates to 102,000 ML/day flow (Path 95/Row 84). The central line indicates the threshold between water and non- water. Two low reflectance value peaks represent Lake Victoria (with deep water) and the River Murray (with shallow turbid water).

Registration and Spatial Accuracy of the Imagery

The successful comparison of images to identify flood extent with increasing flow relied on accurate image registration. The satellite images were registered to Australian Map Grid co-ordinates so that flood masks could be used within the GIS. Registration was performed on the first image in the sequence using both map co-ordinates and digital landuse data. All subsequent images were registered to this image.

The accuracy of the registration was assessed as being within 30 metres using a number of ground control points. Errors may occur in the analysis of sequential images. This problem is of special concern in this study, as individual locations were being monitored over time for inundation. Errors in the registration caused a shadowing effect on the boundaries of the flood extent in some cases.

Flood Map Editing and Coding of the Flood Masks

The flood extent images were used to predict the flood extent for a given flow once anomalies had been removed. Flood extent anomalies can occur from the identification of water that is not hydrologically connected to the river, such as remnants from previously larger flows, rainfall events, irrigation practices or mis-classified pixels. These areas were removed in order to determine the exact extent of flooding that would occur at a particular flow. This involved removing any water bodies identified outside the 1956 flood boundary, the largest flood event in recorded history. The process also involved removing areas of water that were identified at lower flows but not at higher flows. Imagery chosen only on rising hydrographs assisted in minimising such anomalies. Satellite imagery did not detect all water within the scene as some areas were covered by vegetation or have high turbidity or shallow depth. For this reason, the areas that lie within the river channel itself and areas classified as permanent wetlands (Pressey, 1986) were assigned a unique code.

4.3.3 Floodplain Inundation Modelling

Image Interpolation

Mapping from satellite images identified inundation resulting from a range of flows, but it was necessary to consider flows between these events to provide a more continuous predictive model. Interpolation between the discrete flow intervals was performed to produce finer intervals of flood extent.

The flood masks for each flow provided a boundary of equal magnitude flood extent. These boundaries were then interpolated to obtain the flow at all regions in the image. There are many potential ways to perform this kind of interpolation. The true situation is defined by the local topography of the area, which was unknown in this case. Therefore interpolation of the flow level is similar to the problem of interpolating the landscape height at each point. Kriging is the most common surface interpolation method (Burrough, 1986) but is influenced by areas outside the adjacent known boundaries, these being the two closest satellite image masks. It was decided to use an image morphological process called a 'marker-based watershed segmentation algorithm' (Bieniek and Moga, 2000) rather than traditional interpolation methods, to ensure that the information from each satellite image was preserved. The "watershed algorithm" is commonly used in mathematical morphological problems and is often used in relation to topographic analysis of digital elevation models (Vincent and Soille, 1991). The advantage of this method over other approaches to contour interpolation is that it can be applied to very noisy data with broken contours. This was the case, since the data was derived from the 30 metre satellite pixels.

Flood interpolation was achieved by dividing the combined satellite mask image into regions of constant minimum flow. Using this method provided certainty that interpolated values lay within the minimum and maximum flow bounds by interpolating each region independently. In each region, contours of equal flow are interpolated from the boundary points, at which the minimum flow is known. This contour interpolation was repeated iteratively, each time based on the contours that have already been estimated. Each contour was interpolated using a flooding simulation extending from the next higher and lower boundaries. Regions having the minimum flow value or representing land that did not flood at the largest flow were not included and used as the lower and upper limit of interpolation accordingly.

The difficulty in applying the watershed algorithm to flood interpolation was in choosing the source and sink points. The sink points are points adjacent to the region that have the next highest quantised flow level. The source points are those adjacent points that have the highest flow that is lower than the sink points for this region. The location of these sink and source points was chosen to model the behaviour of flood growth across the floodplain but also to represent the filling of wetlands from a single inflow channel. A set of colour aerial photographs for the whole region of a 70,000 ML/day flood was used to validate the growth behaviour of floods and minor changes were made to the final flooding grid.

The result of the interpolation was a raster grid of cell values that represented the flow in the river that will cause those areas to inundate (commence-to-fill). This grid was then filtered using a 9 by 9 majority filter to remove anomalies in the interpolated flood masks, such as higher or lower value pixels in the middle of lakes. This filter replaced the value of each pixel with the majority value of the eight nearest neighbours.

At this stage the derived surface was still in raster format (pixel based data). The raster data was converted to vector form (area or shape based data) to reduce the data volume and allow for easy retrieval, updating and generalisation of graphics and attributes. The conversion to vector created polygons of equal commence-to-fill values.

River flow and height model

To create a model that would predict floodplain inundation from a regulated river, the river height at every kilometre of the river was required. Recorded data of river height at certain distances from the mouth of the river, and flow at gauging stations during the flooding events, were obtained from the South Australian Water Authority, the organisation that manages the river in this region. The limited number of gauging stations meant that a modelled series of backwater curves was also used. These were water surface elevation curves between locks for a steady uniform flow computed by the Murray River Level Module (MURLEV), part of the River Murray Flow and Salt Transport (RMFST) computer model (Water Studies, 1992). Figure 4.3 shows the flow curves derived from these methods for a section of the river from the border to Lock 2. The flow rate of 3 GL/day equates to a river height referred to as weir pool level.

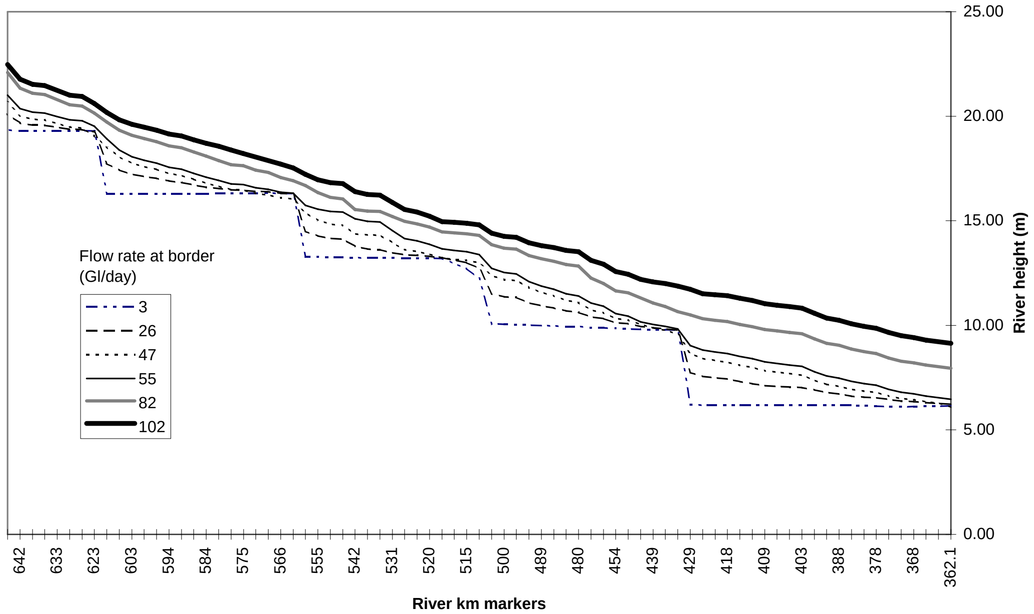


Figure 4.3 River levels for kilometres of the river from Lock 2 to the SA/NSW border for different flows.

The river is currently held higher than pre-regulation levels by the weirs which cause the river water levels to have a stepped backwater curve rather than a continuous flow downstream. Once flow levels have reached 60,000 ML/day the river is above the maximum height of the weirs and backwater curves become straight.

Flood Units

As river heights can be manipulated by the six weirs, largely independent of each other, the floodplain needed to be divided into regions that could be related to the local conditions in the river. Areas where the floodplain inundation responded to the same point in the river were identified using flood behaviour and river morphology as a guide. These regions were termed Flood Units.

Flood Units were assigned the closest kilometre marker from the hydrological model (trigger point) or the trigger that was assumed would most likely cause that area to flood. As a height difference of several metres can occur between the upper and lower pool levels of the weirs, behaviour of the flow path around the weir is complex. A weir manipulation trial at Lock 5 demonstrated that the area around the lock responded to changes in the river height below the lock. There was very little difference between the river height within two kilometre intervals in the modelled backwater curves, which implies that there may be an error margin of approximately two triggers up or downstream in the assignment of a Flood Unit to a trigger. Each mapping polygon for each particular reach contained attributes for Reach, Flood Unit, Flow and Trigger.

Once the flood extent data had been converted to polygons in the GIS it was clear that each Flood Unit included small areas of floodplain that were inundated at the same particular flow. These areas were termed unique Ecological Units as they are likely to have similar ecological conditions given that they occur at the same elevation and therefore flood regime.

Hydrological Inputs to the Model

The same flow rate at the South Australian border gauging station can produce different river elevations depending on the weir elevations, the antecedent conditions and the time of year. Local river height predominantly determines whether the river will break the banks and flood an adjacent Flood Unit. Antecedent conditions were not included in the model. During hot dry summer periods, a flow rate at the border will be further reduced downstream compared with wetter months, as water is evaporated and extracted from the river and less is replaced by rainfall. Curves were generated showing the attenuation of flow at each weir for each month of the year using the MURLEV (Figure 4.4). These were used in the model to take into account direct evapotranspiration from the surface of the river channel.

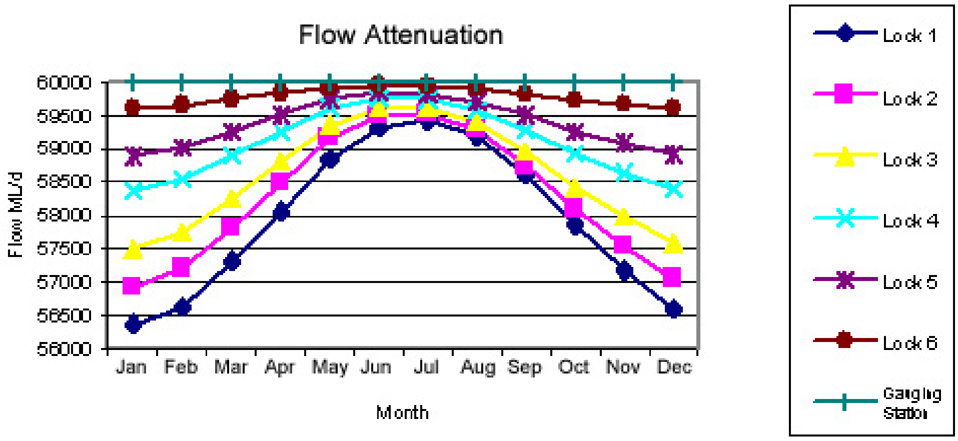
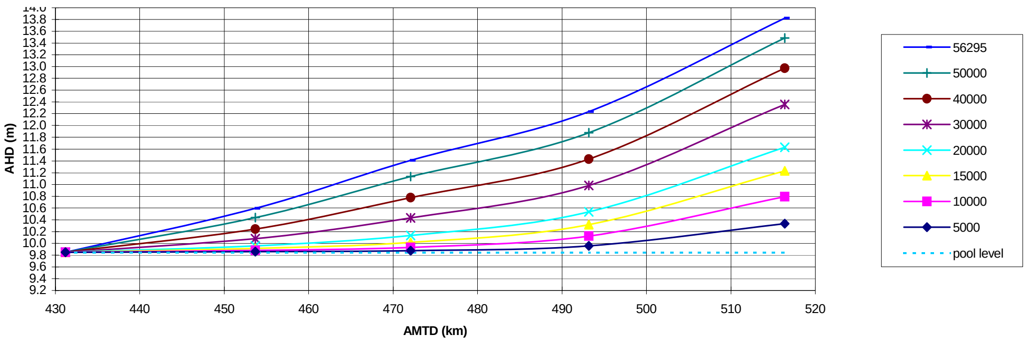


Figure 4.4 Example of seasonal attenuation curves.

The RMFST Model (Water Studies, 1992) was used to generate backwater curves for a range of weir manipulations and flow values. The model estimates the stage discharge matrices used to route the inflow hydrograph down a river. For a nominated water level and discharge at the lock, the backwater model defines the upstream water levels to the next lock. In this way relationships can be derived between the water levels at the downstream and upstream ends of a river segment and discharge. The river level and discharge relationships for each lock reach were established using a linear relationship. Figure 4.5 shows the backwater curve for the weir pool of Lock 3 at a level of 5 centimetres above normal pool level for a series of river flow magnitudes. The precision of the model is five centimetres in the river height predictions.

The floodplain inundation model is a steady state model that predicts the extent of flooding from a given flow on the first day of the flood. It does not consider the effect of antecedent conditions or the effect of flood duration. Further research on the wetting and drying behaviour of the floodplain and its wetlands needs to be incorporated into the model to be able to predict time sequences for management scenarios.



Distance from mouth

Figure 4.5 Example of a backwater profile for Lock 3 for different flow volumes with river level 5 cm above normal pool level.

Linking Images to Flows

Hydrological model output was provided as input data to the GIS. The GIS model then used look-up tables to query the required hydrological parameters that then link to the spatial layers in the GIS. This type of loose coupling allows existing hydrological models to be run without reprogramming within the GIS (Karami and Houston, 1996; Sui and Maggio, 1999). As the hydrological parameters do not change with the different management scenarios considered in this thesis, this method proved an effective and economical approach.

Developing an Spatial Interactive Model

The height of the river at triggers had to be related to the management areas, the Ecological Units derived from the original flow values and the interpolation process. With area inundated being extremely sensitive to slight alterations in river height, it was decided that a relationship between river height and area of flooding should be used in each Flood Unit. This relationship has the potential to be improved with further images capturing different flood magnitudes. The model will always be limited to the spatial resolution at which the images are acquired, which for Landsat Thematic Mapper represents 30 metres on the ground, regardless of the number of images used. If a change in the height of the river results in less than a 900 square metre change of the flood boundary, the area of inundation shown by the model will not change.

Other satellite systems may be useful in increasing the ground resolution and therefore identifying finer changes in floodplain inundation.

In addition to the resolution of the imagery, the method used to interpolate between flood boundaries also influences the spatial accuracy of the model when predicting extents between the input extents.

In the region downstream of Lock 3, overbank flows are still above 35,000 ML/day. River flows above this level inundate increasing area of floodplain until the flood extent reaches the narrower cliffs of the floodplain, which occurs at lower flows than in the section of wide floodplains above Lock 3. The relationships at each floodplain unit were used to code the flood masks with a river height which would cause the Ecological Unit to be inundated. The height of the river was assigned to the Ecological Unit for all Flood Units using the Flood Unit trigger as the point that would cause this Ecological Unit to flood.

A dialogue screen was then designed with input fields for values for the flow at the border, the month of the year and the height of each weir. These values are then used to perform a query which selects all the river height codings in the map layer where the local river height is less than that predicted for the given flow (attenuated for the time of the year). Figure 4.6 shows a screen from the model depicting two flood events modelled over the Chowilla floodplain area.

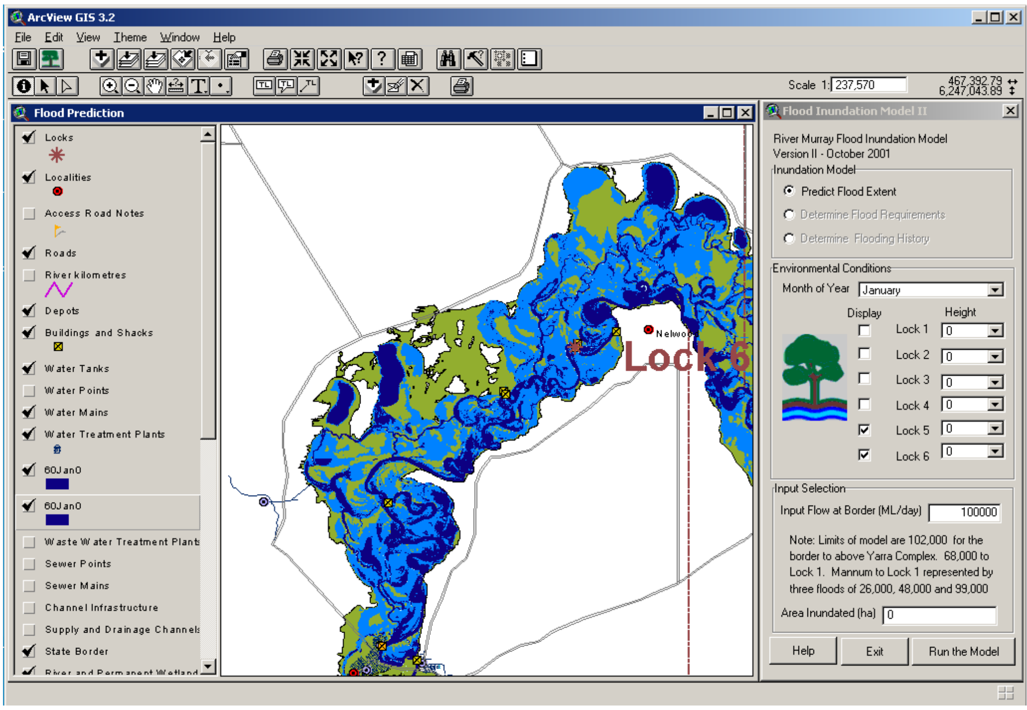


Figure 4.6 Example output of the GIS Floodplain Inundation Model showing an area around Lock 6 at Chowilla with two flood predictions of 60,000 ML/day (dark) and 100,000 ML/day (light).

Results of the RiM-FIM

The total area of inundation of the floodplain in South Australia, including the whole of the Chowilla floodplain, which is partly in New South Wales, down to Wellington in South Australia is approximately 118,000 hectares. Figure 4.7 shows the increasing area of inundation of the floodplain from 5,000 ML/day to 100,000 ML/day. The area of permanent water is approximately 24,000 hectares and comprises the river channel, wetlands and anabranch creeks.

Figure 4.8 shows an example of the RiM-FIM in the confined area below Lock 3.

The hydrological modelling allows the monthly simulation of a flow from the border under different weir configurations, with the GIS providing the spatial representation of the area inundated. The inputs to the floodplain inundation model are flow at the border, weir configurations of all 6 weirs and the month of the year. The outputs of the model are river heights at each trigger kilometre and area of inundation. The floodplain inundation model is in a GIS framework and has incorporated layers such as riparian vegetation, transport and water infrastructure and major wetlands.

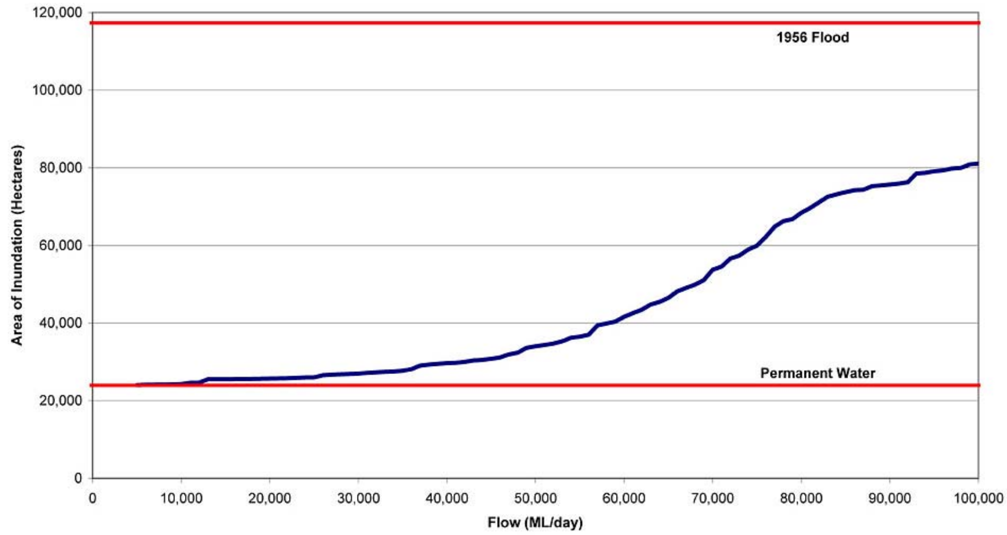


Figure 4.7 Area of flooding versus flow magnitude for the South Australian border (Chowilla floodplain) to Wellington. The graph shows the area of inundation compared to the area of permanent water and the extent of the 1956 flood (~258,000 ML/day)

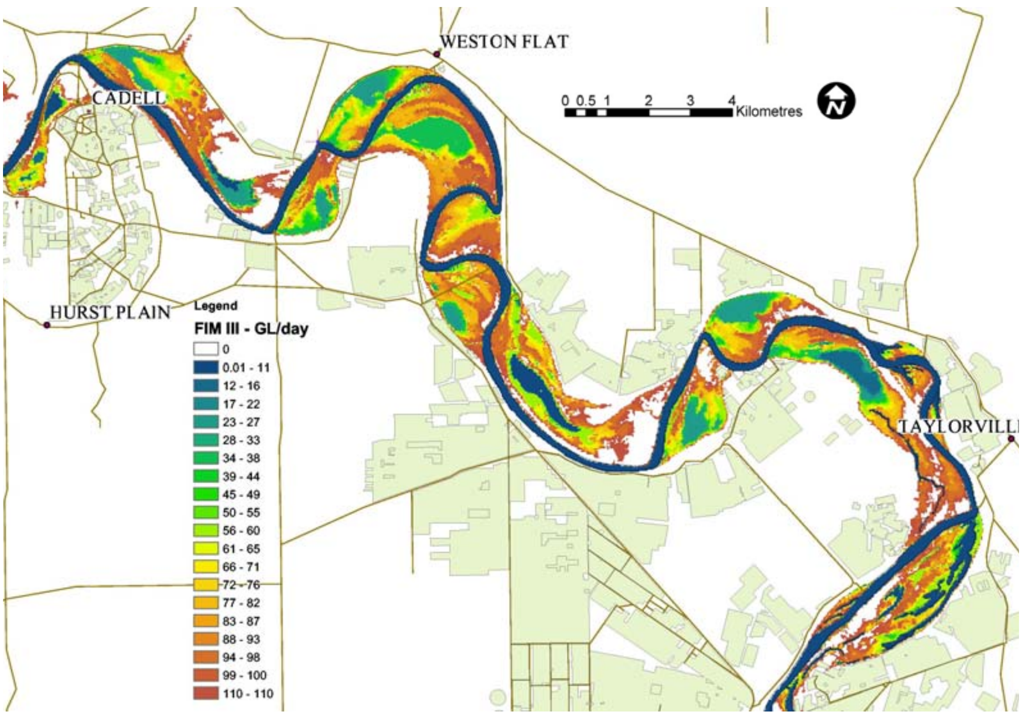


Figure 4.8 Example output of the RiM-FIM showing an area around the Sunraysia irrigation area upstream from Lock 2.

Validation and accuracy of the RiM-FIM

Aerial photography was obtained during a period in 2005 when the weir was raised and was interpreted for the area of inundation. It was estimated that the area of inundation predicted by the model was 15% less than the area interpreted from the photography.

A pseudo-elevation dataset was derived as part of the floodplain inundation modelling by using the river heights for commence-to-fill as surface elevation heights. This dataset has proved invaluable in a number of applications (FIP Model described in the next Chapter; MODFLOW modelling of Clarks floodplain near Loxton (Doble, 2004); Class Risk Model of the lower River Murray (Gabrovsek *et al.*, 2002); Floodplain Risk Methodology for the lower River Murray (Holland *et al.*, 2009) and was produced economically when compared to the capture of survey elevation data. The elevation surface produced from the floodplain inundation modelling work was independently tested against 126 bores measured for ground surface height in 2003 by CSIRO Division of Land and Water. The height of the surveyed ground surface was compared to the height from the RiM-FIM. A frequency distribution of the 858,776 cells from the elevation model compared to the floodplain inundation model heights for the whole of the lower River Murray is shown in Figure 4.9. The mean was -1.84 metres but the values below -9 represent areas where the model did not extend. This gives a true mean of approximately -25 cm.

Results have validated this approach for determining flood extent over such a large area and its merits when compared to detailed elevation and hydrodynamic modelling, or rating curves and hydrographs for analysis, by including lateral and spatial elements. The GIS model allows prediction of impacts on infrastructure, wetlands and floodplain vegetation, allowing quantitative analysis of flood extent to be used as an input into the management decision process.

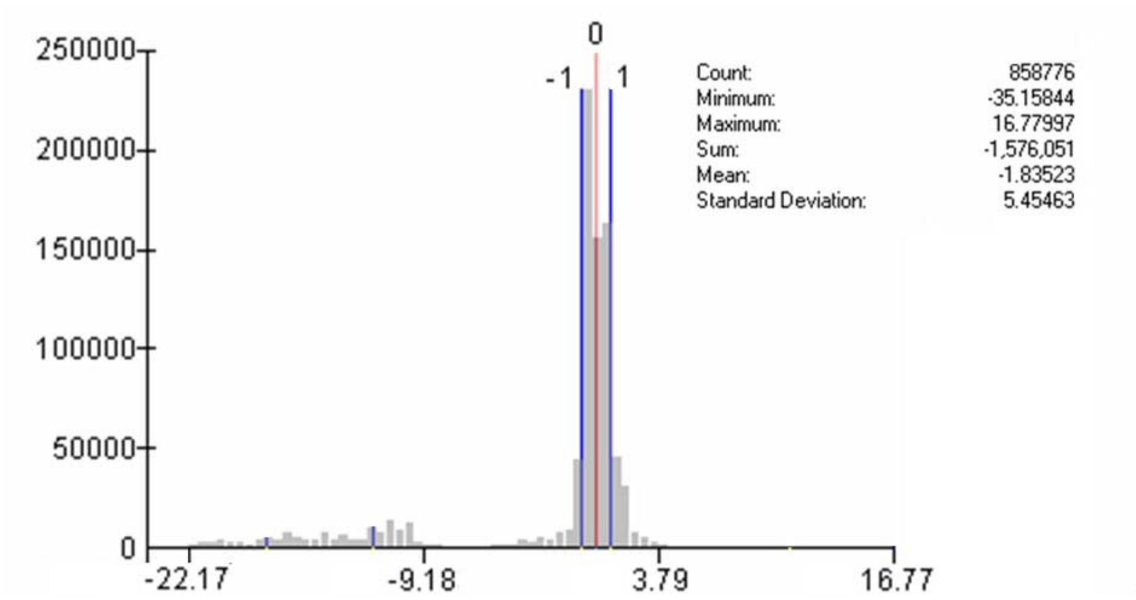


Figure 4.9 Frequency distribution of the cells from the elevation model compared to the floodplain inundation model heights for the whole of the lower River Murray.

The Floodplain Inundation Model produced has been used as the basis for an environmental flow strategy in South Australia (SKM and Mapping and Beyond, 2002) and a number of other studies on the hydrology and ecology of the floodplain. The Floodplain Inundation Model was used to predict the area of inundation expected to occur as a result of a raising of weir 5 near Renmark in 2000 (DWLBC, 2006).

Accuracy of the RiM-FIM is dependent on:

- Georegistration errors of the imagery which are considered to be less than one pixel. This influences the location of the flood extent, the growth in flooding and the interpolation;
- Errors associated with defining the extent of flooding, because flood edge may often be very shallow, or water under vegetation cover, there is uncertainty in defining the limit of flooding, using density slicing of the IR band;
- Images exactly coinciding with the flow magnitudes cited in table 5.1, or water may have risen or receded in the interval between image capture and the time/date and location of the gauging station;

- The interpolation process including the majority filtering;
- Determination of specific flow contours from the interpolated grid;
- Assignment of the trigger points; and
- The stochastic nature of processes with different flood extents for same magnitude of flooding – antecedent conditions, shape of hydrograph, rate of rise, duration of flood event

This can be described as the accuracy of the model, a level of certainty in terms of inundation, accuracy of the river height to cause inundation or accuracy of the river flow to produce that height.

Limitations of the RiM-FIM

The limitations of the RiM-FIM include the spatial accuracy of the model limited by the detection of water bodies from the satellite image density slice method and interpolation between discrete flood boundaries. This may be improved by using multi-band analysis as an increased number of bands may be useful to delineating flooding more accurately (Overton *et al.*, 2010). Increasing the number of images used may also improve the accuracy of the model by providing more flood levels to interpolate. A number of images of the same flow but different dates could also be examined to distinguish natural variability of flood extents from equivalent flows and could be used to produce an 'average' flood extent. The resolution of the model at a 30 x 30 metre grid means that flood boundaries that move less than approximately 15 metres will not be detected. Using higher resolution imagery would improve the ability to detect smaller changes in flood extent. The interpolation between image events is also restricted by this grid size and resampling the grid to a higher resolution would increase the interpolation method's ability to model the finer boundaries of 1,000 ML/day flood extents, however still limited by reliability of the interpolation method.

The RiM-FIM is reliant on predicting the extents of floods based on previous flood events. Changes in the surface flow patterns caused by water retention structures or changes in topography due to ground works or vegetation surface flow barriers, will limit the predictive capability of the RiM-FIM. In addition, flooding in the River Murray is complex and no two flood events will act in the same way and cover the same extent (GHD *et al.*, 1986). The steady-state modelling approach used in the RiM-FIM is limited by its ability to model antecedent conditions and variability in the hydrograph of particular flood events.

Modelling the effects of new flow control structures requires the development of a more dynamic model for the floodplain that can determine areas of inundation from the introduction of flow barriers. This will be explored in the next section.

4.3.4 Modelling Floodplain Tree Health from Flow Management Scenarios

Calculating a Drought Index

An index of current decline in flooding can be calculated by comparing the natural flood return period with the current flood return calculated from the last 50 years of flow record (Overton and Doody, 2008b). The MDB daily river model, BigMOD, provided flow data (ML/day) spanning the years of 1891 to 2000 (in the first instance) under current and natural flow regimes, which was used to extract a maximum flow rate per year, under both scenarios. Where available, actual current flow data was used from 2000 to mid 2007. The current data was used to calculate the number of years since a flood last occurred within that region, covering a range of flow rates (5,000 – 250,000 ML/day) using data from 1956 to the present. Flow data from the major 1956 flood was used as the baseline for flooding over the past 51 years.

BIGMOD modelled natural data was used to determine the natural flood return period for the range of flow rates. The natural flood return data was compared to the years since last flood information for the same zone, producing a ratio of actual flow/natural flow for each zone for the range of flow rates. This ratio is referred to as a 'drought index'. The drought index implies a level of risk to the floodplain and its vegetation communities. If an area of floodplain, that under a natural flow regime is flooded on a 4 yearly basis, but under current flow conditions has not been flooded for the last 20 years, then it has a drought index of 5, meaning that area has now not been flooded for 5 times its natural flood return period. Low risk to floodplain tree decline was identified as being a drought index less than 3 times the natural flood return rate and high risk as 3 times or greater than the natural flood return period from comparison between drought index maps and actual vegetation health.

Drought index maps were created for the upper River Murray using this same method for the Victorian Environmental Assessment Council using a Geographical Information System and RiM-FIM zone layers (Overton and Doody, 2007). A river red gum study site layer supplied by VEAC was clipped to each RiM-FIM zone layer. Drought index maps highlighting areas of low and high risk within the floodplain were produced for each region for the years of 1970 and 2007. The year of 1970 was targeted as it highlighted potential conditions under reduced extractions given current infrastructure, where the drought index remained low, indicating healthy vegetation for some zones. The further downstream each zone, the earlier the year where the floodplain was predominantly at low risk. This is due to decreased rainfall and floodplain salinisation that requires greater flooding and therefore requires the influence of the large 1956 flood. The area of the floodplain for each drought index value was calculated for both the 1970 and 2007 drought index maps, to determine what proportion of the river red gum study site fell into the low and high risk category for each zone in both of the years.

The Chowilla floodplain had a drought index of 1 or 2 over the whole floodplain in 1964 (Figure 4.10) and almost all vegetation occurred in areas of index greater than 3 in 2003 (Figure 4.11).

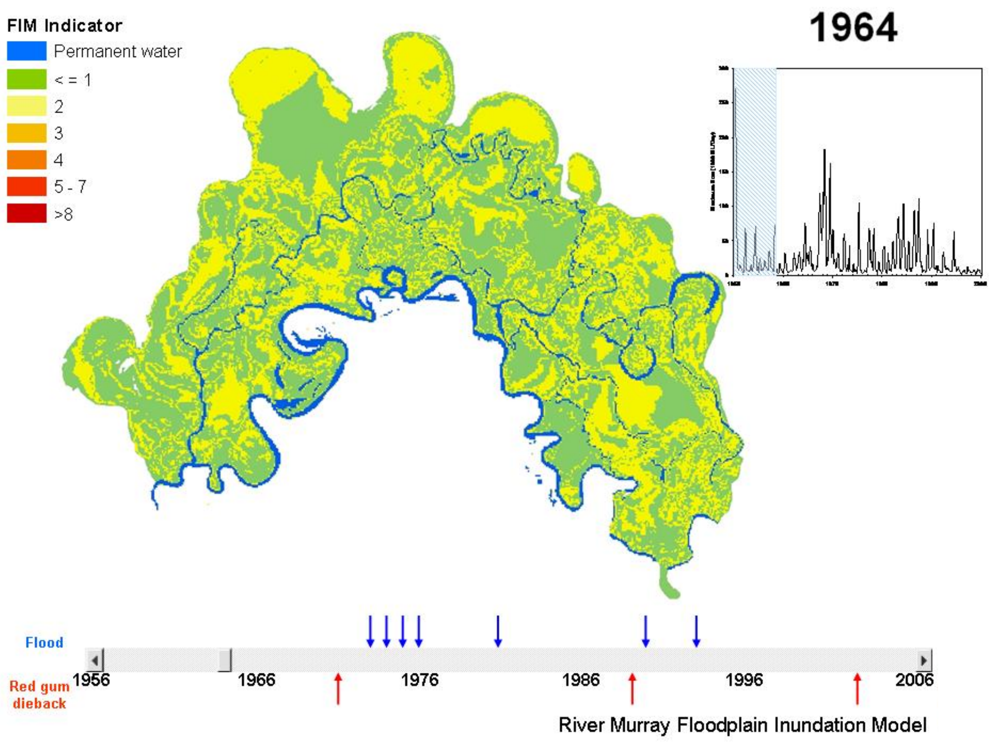


Figure 4.10 The Chowilla floodplain showing the drought index for 1964.

The drought index map developed for Chowilla floodplain in 2003 (Figure 4.11) was compared with the vegetation health map of the same year (Figure 5.3). This analysis was undertaken to test the validity of using the drought index to infer floodplain tree health and the critical value of 3 for the drought index to separate low from high risk of floodplain tree health decline.

Comparison of the two maps indicated there was evidence for using the drought index to distinguish vegetation health for areas away from permanent creeks and wetlands (Table 4.2). Most of the vegetation in the drought index 1 class was healthy (66%), whereas most of the vegetation with a flood index of 2 or more was unhealthy (65-82%). In areas close to permanent creeks and wetlands (Table 4.3) there was no clear trend in vegetation health for any of the drought index classes.

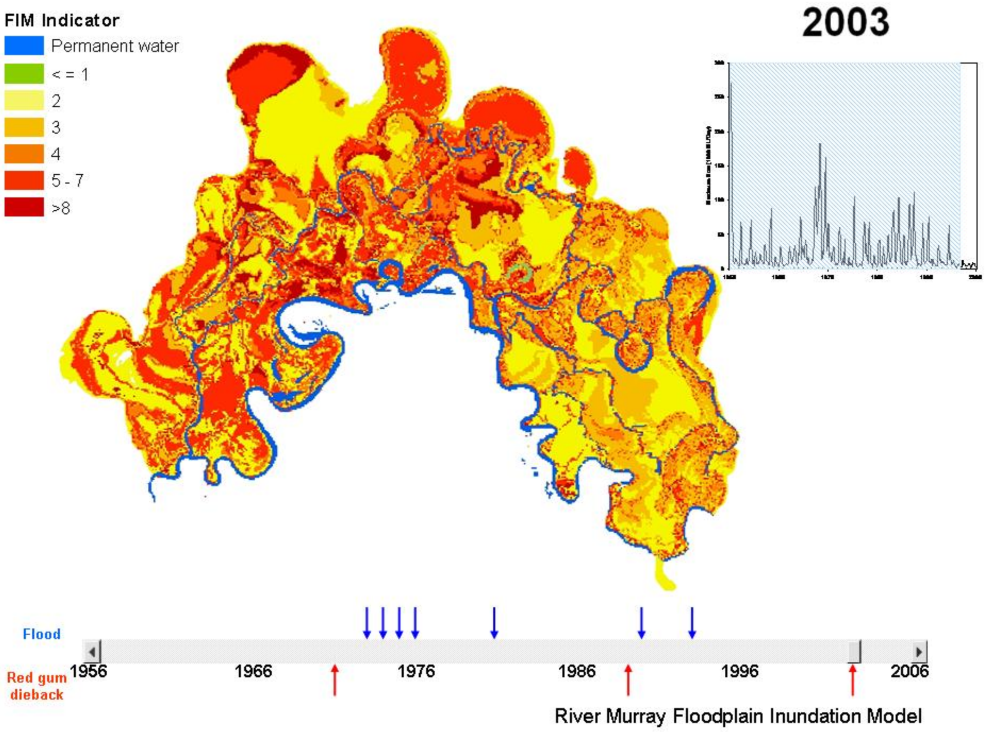


Figure 4.11 The Chowilla floodplain showing the drought indicator for 2003.

Table 4.2 Relationship between vegetation health, drought index and area when vegetation is greater than 50 m from the river at Chowilla.

Drought Index	Dead %	Healthy %	Unhealthy %	Total %	Total Area (m²)
1	20	66	14	100	1,664,539
2	1	19	80	100	19,137,815
3	3	16	82	100	12,498,424
4	4	19	77	100	9,828,433
5	2	33	65	100	3,335,194
6	8	24	68	100	6,735,882
7	0	58	42	100	187,309
9	1	41	58	100	2,525,702
10	5	30	65	100	209609

This data can be represented another way by analysing where the healthy, unhealthy and dead vegetation occurs. Table 4.4 displays the distribution of each tree health class away from creeks and wetlands and Table 4.5 for areas close to creeks. Away from creeks, most of the healthy trees occur in drought index 2 areas, most of the unhealthy in drought index 2 and 3 areas and dead vegetation in drought index 3-6 areas. Again, within the vicinity of permanent water the healthy trees are spread over a range of drought index classes along with unhealthy and dead vegetation.

Table 4.3 Relationship between vegetation health, drought index and area when vegetation is less than 50 m from the river at Chowilla.

Drought Index	Dead %	Healthy %	Unhealthy %	Total %	Total Area (m^2)
1	1	**74**	25	100	1,463,315
2	1	58	42	100	5,686,708
3	2	**62**	36	100	7,794,155
4	3	57	40	100	7,178,315
5	1	**72**	27	100	3,103,441
6	3	**69**	28	100	5,513,313
7	0	51	49	100	143,457
9	2	57	41	100	2,066,254
10	5	**84**	12	100	506,966

Table 4.4 Relationship between vegetation health and drought index when vegetation is greater than 50 m from the river at Chowilla.

Drought Index	Dead %	Healthy %	Unhealthy %
1	6	3	3
2	13	**31**	**36**
3	**19**	17	**24**
4	**22**	16	18
5	4	9	5
6	**33**	14	11
7	0	1	0
9	2	9	3
10	1	1	0
	100	**100**	**100**

Table 4.5. Relationship between vegetation health and drought index when vegetation is less than 50m from the river at Chowilla.

Drought Index	Dead %	Healthy %	Unhealthy %
1	4	5	4
2	6	**16**	**20**
3	**22**	**23**	**23**
4	**31**	**20**	**24**
5	5	11	7
6	**21**	**18**	13
7	0	0	1
9	7	6	7
10	3	2	0
	100	**100**	**100**

This analysis suggests that the drought index is useful for identifying floodplain tree health for areas away from permanent water. Trees in areas flooded within their natural return period are generally healthy, areas with a drought index of 2 are healthy to unhealthy and areas of drought index of 3 are unhealthy to dead. A drought index of 2 is half the water of a natural flood regime. Cullen (2001) states that '*Any river where more than 1/3 of the median flow is extracted is likely to be seriously damaged*'. This equates to a drought index of 1.5. It is reasonable to suggest that extracting 2/3 of the water (a drought index of 3) may lead to the death of the vegetation. Jones (2002) indicates that the probability of a '*healthy working river*' is high when there is 2/3 or more of natural flow, moderate when there is greater than 1/2 (drought index of 2) and low when there is less than half (drought index of 3 or more).

Further evidence for the 1970 flood volumes having potential for representing best case can be seen from the vegetation mapping of the Bookpurnong and Gurra floodplains (Figure 4.12). This mapping, described in Section 4.3, illustrates the condition of the vegetation in 1972. Apart from some small areas of decline, mainly due to irrigation inflows, the floodplain was in good health.

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Figure 4.12 Vegetation health on the Bookpurnong and Gurra floodplains, South Australia showing the decline in vegetation health in 1972 (Telfer and Overton, 1999b).

4.3.5 Discussion of Flow Management Scenarios

The RiM-FIM was developed using GIS, remote sensing and hydrological modelling. Flood extents were determined from Landsat satellite imagery for a range of flows, interpolated to model flood growth patterns and linked to a hydrological model of the river.

The resulting model was evaluated using aerial photography of a flood and weir raising event and shown to underestimate the flooded area by approximately 15%. The model has been used successfully as the basis for the development of a flow management strategy for the River Murray in South Australia (SKM and Mapping and Beyond, 2002).

The integration of the hydrological model with the GIS has enabled simulation for both scientific research and policy management. The visualisation, quantitative analysis and spatial correlation of environmental and infrastructure data of the GIS has improved the usefulness of the hydrological modelling. The GIS framework could be utilised to determine flow patterns and losses across the floodplain, incorporating wetlands as sources and sinks to create a more dynamic model of flooding (Costelloe *et al.*, 2003). The model has also provided a useful surface elevation model for further floodplain modelling work.

The approach used here to develop the floodplain inundation model for the River Murray could be used to develop other floodplain inundation models in large lowland rivers. Information needs include:

- Satellite imagery for a range of flows required to be modelled;
- Knowledge of river flows at the time of the images;
- A simple hydraulic model to convert river flow into river heights if river height manipulation through weirs or other regulatory structures are required; and
- Some local knowledge of trigger points to determine the local river point that influences inundation.

The South Australian model linked river heights to flows and allowed modelling of local river manipulation through control structures. This portion of the model required backwater curves for the range of river flows and control structures to be considered.

The RiM-FIM predicts the extents of floods based on previous flood events and is therefore based on existing or previous flow by control structures and surface flow barriers. Given the complex nature of flooding and changes to the terrain, the steady-state modelling approach used in the RiM-FIM is limited. Further research on the behaviour of floods on the floodplain and in wetlands is required before it can be used as a temporal model. A new pseudo-hydrodynamic model for the floodplain was developed to model inundation under changing conditions, such as operation of a new weir structure.

Changes in flow regulation, water extraction and recent drought have created very low water availability conditions on the floodplain. Changes in flood return periods have led to changes in the current extent of floodplain trees being no longer at their optimal elevation on the floodplain in relation to their preferred flow regime. Gabrovsek *et al.* (2000) noted that a 104,000 ML/day flood event has a natural return period similar to the current return period of a 70,000 ML/day of 1 in 4.8 years. Similar results can be seen from Table 4.6 (Sharley and Huggan, 1995), where the whole of the floodplain at Chowilla was once inundated by the 140,000 ML/day flood extent which is reported as occurring 1 in 7 years and which now has the equivalent frequency of a 70,000 ML/day event. This frequency can be considered as that required by the 'active floodplain'. This 'active floodplain' concept has been used throughout this thesis in determining potential management scenarios.

The RiM-FIM can be used to determine the extent of inundation as a result of increased flow and weir manipulation. One method of considering the impact of flow regime changes is the 'active floodplain' concept described in this chapter. Areas above 70,000 ML/day are above the influence of weirs and are dependent on natural floods of this size occurring in times when major storages along the River Murray are already full. It is more likely that flows below 70,000 ML/day can be manipulated using weirs and releases from storages. Therefore, approximately 30% of the floodplain can be actively managed for environmental flows in the lower River Murray without the need for new control structures.

Distribution of black box and river red gum trees on the Chowilla floodplain in relation to flow magnitude is given in Figure 4.13. Most of the black box and some of the higher elevation river red gum are in the 'old active floodplain' area and are likely to be suffering from water stress and possibly salinisation. The 'active floodplain' boundary is moving slowly to lower flows as water extractions increase and drought continues. Observations of widespread river red gum dieback in the lower River Murray in 2005 (MDBC, 2005) could be an indicator that the peak of river red gum trees at lower flows than 72,000 ML/day has now been shifted into the non-active floodplain area.

Table 4.6 Return periods of a range of flows to South Australia at Chowilla (modified from Sharley and Huggan, 1995)

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The ‘active floodplain’ concept is postulated to describe the area of floodplain that remains active as a flood dependent ecosystem. In areas of less flooding frequency the vegetation would adapt and change to a rain dependent system. This is evident with large areas of the Chowilla floodplain becoming dominated by saltbush and bluebush in areas that used to have flood dependent grasses. The active floodplain concept suggests that the trees above a 1 in 7 year return period (72,000 ML/day under current conditions) are under threat from water stress and possible salinisation. As the drought of 2001 continued it was evident that the line of the active floodplain was shifting further to lower flow magnitudes and exposing the majority of red gums as well as black box.

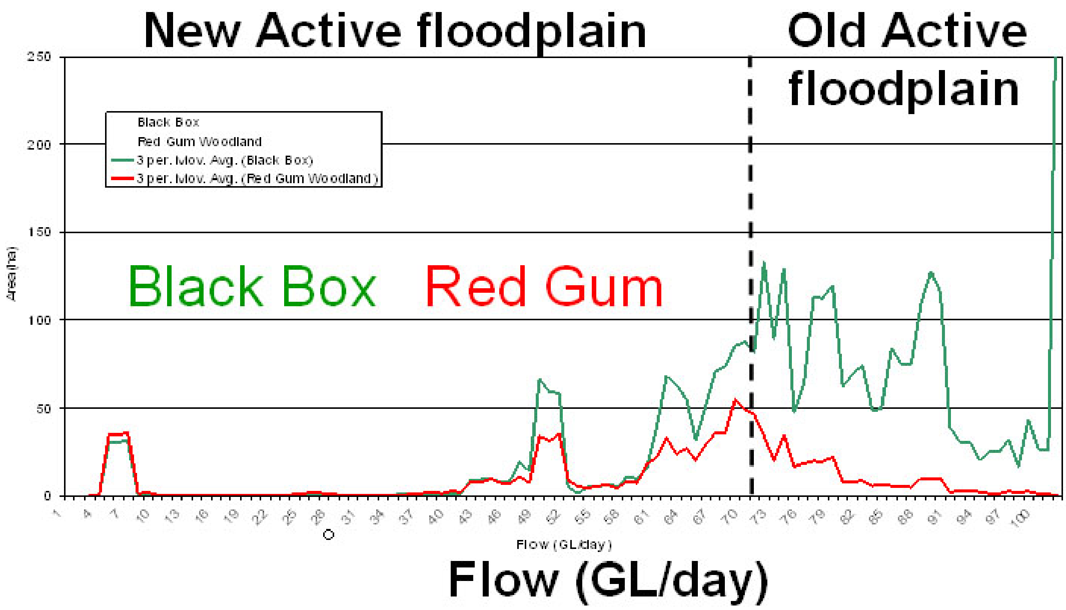

Figure 4.13 Distribution of black box and river red gum on the Chowilla floodplain in relation to flow at the Border.

An assessment of the changed flooding regimes on the River Murray floodplain was undertaken using the RiM-FIM. A flood index was developed annually for the whole floodplain which scores the number of 'natural return periods' since the last flood. This flood index has been linked to vegetation decline (high risk) when the index is 3 or greater (three times the natural return period since last flood). The results show that up to 74% of the floodplain is at high risk from reduced flooding frequency. The project has identified that much of the area that has the largest reduction in flooding frequencies is the riverine and floodplain forest areas, along with floodplain woodlands and lignum areas. Flood index values below 3 (low risk) were present over most of the floodplain in the late 1960s and early 1970s. This suggests that the volume of flow in the river during the previous 10-20 years was sufficient to maintain vegetation health. This is supported by observed healthy floodplain vegetation in that time period. Preliminary results of the volumes of water required to restore/maintain vegetation health can be estimated from the volumes during that time period. Volumes of approximately 1,700GL per year for the floodplains of the lower River Murray are required to maintain flood index values below 3.

The monthly flow in the lower River Murray that occurs as overbank flows can be graphed over time (Figure 4.14). This figure shows that the monthly flow has occurred in a peak and trough pattern on an approximate 20 year cycle. However the peaks have been decreasing in magnitude over the last 60 years. The latest trough corresponding to the drought that started in 2001 is much lower than in previous years. Indicating the severity of the drought on overbank flooding.

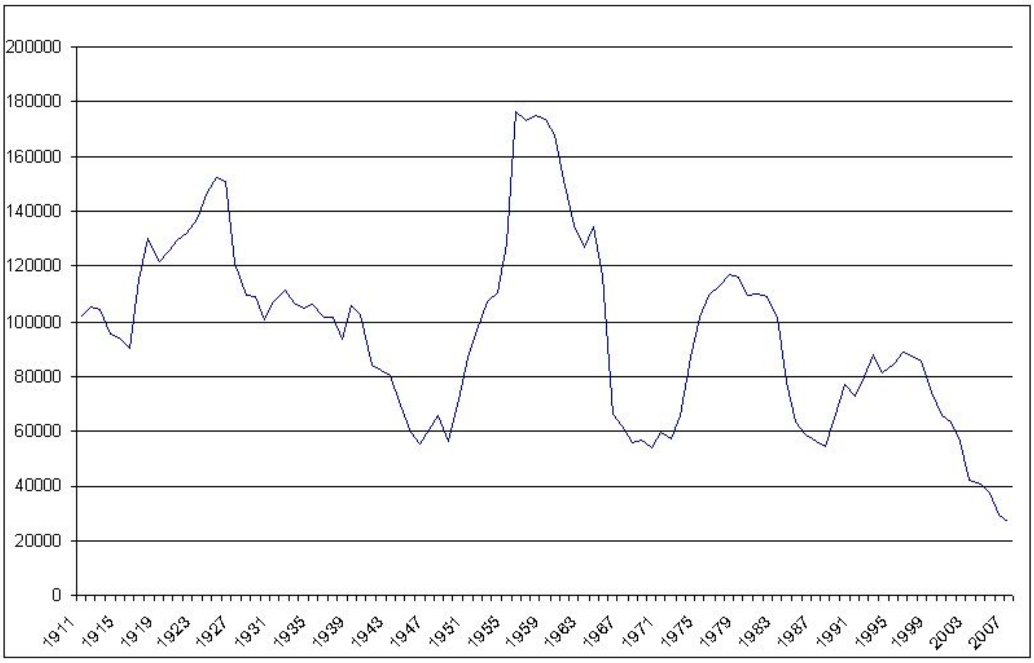

Figure 4.14 Volume (ML/day) of monthly flow as a moving window that occurs over 35,000 ML/day (overbank flow) from 1911 to 2008.

Changes in flow regime can be brought about by weir manipulation, increasing river height and by extra releases upstream to produce greater river channel flows. Although both of these approaches increase the river height, they are quite different in terms of the flow direction, velocity and water quality. A flood brings with it carbon, nutrients, seeds and larvae as it flows over upstream floodplains. Regulated inundation from retained river flow is unlikely to contain these to the same degree. It is therefore important to not replace all floods with artificial ones.

4.4 ASSESSING GROUNDWATER MANAGEMENT SCENARIOS

4.4.1 Introduction

A further objective of this chapter is to develop a predictive model of vegetation risk from the impacts of groundwater management scenarios for the lower River Murray.

Groundwater depth and groundwater salinity play major roles in vegetation health in this highly salinised, shallow groundwater environment. Shallow and highly saline groundwater with clay soils creates an environment at high risk of salinisation, low water availability and poor tree health. Deep groundwater and a freshwater lens with sandy soils means slower salt accumulation rates, greater leaching potential and a supply of freshwater during drought periods.

This Section presents the methodology and results of a regional groundwater model for assessing the impact on the floodplain environment from groundwater management scenarios. Groundwater depth is the most critical of these factors and is of primary concern when considering regional processes in the lower River Murray region where the groundwater salinity is fairly consistent. Given the difficulties in modelling groundwater depth across the whole of the lower River Murray floodplain the approach taken is to model the risk of salinisation by distributing groundwater inflow to seepage, evapotranspiration and salt loads to the river.

Components of the methodology, results and discussion of this section have been presented in the following publications:

Holland, K.L., Jolly, I.D., Overton, I.C. and Walker, G.R. (2009). 'Analytical Model of Salinity Risk from Groundwater Discharge in Semi-Arid, Lowland Floodplains'. *Hydrological Processes* 23: 3428-3439.

Overton, I.C., Jolly, I.D., Holland, K. and Walker, G.R. (2003). 'The Floodplain Impacts Model (FIP): A Tool for Assisting the Assessment of the Impacts of Groundwater Inflows to the Floodplains of the lower River Murray'. Report for the River Murray Catchment Water Management Board and the National Action Plan for Salinity and Water Quality by the CSIRO Division of Land and Water, Canberra.

4.4.2 Modelling Groundwater Management Scenarios

At the regional scale, management scenario modelling was based on the analytical cross-section model as this model is achievable with the data available. To develop this model required the establishment of the following spatial layers:

- Vegetation community map simplified from the Smith and Kenny (2005) survey;
- Vegetation health map to calibrate the model;
- Flood extent maps for a range of flows;
- Soil hydraulic values, aquifer thickness and other properties required by the analytical cross-section model; and
- Groundwater inflows to the lower River Murray.

The FIP model presented here develops a spatial model based on the analytical cross-sectional model of Holland *et al.* (2005) and applies this model to the River Murray region in South Australia using a GIS layer of the floodplain, divided into multiple divisions. Groundwater salinity and groundwater depth are spatially highly variable at the local floodplain scale.

Analytical Cross Section Model

Holland *et al.* (2009) conceptualised the River Murray floodplain in a simple cross sectional model, comprising a surface layer of Coonambidgal Clay overlying a layer of Monoman Sands (Figure 2.23). The highland area is represented by the unconfined Upper Loxton Sands (Pliocene) aquifer. Both the highland and floodplain are underlain by the lower Loxton Sands, a relatively impermeable clayey sand formation. In the region downstream of Overland Corner, the water table beneath the highland is generally contained within the Murray Group limestone aquifer that underlies the Pliocene Sands.

The arrows in Figure 4.15 represent groundwater flow directions and potential groundwater discharge sites. Groundwater flow in the Upper Loxton Sands (or Murray Group limestone in the region below Overland Corner) is a combination of irrigation recharge and regional groundwater flow. Groundwater from the highlands is discharged as either seepage at the break of slope if the groundwater level is above the surface, evapotranspiration through the floodplain when the water table is within the evapotranspiration extinction depth (vegetation rooting depth), or as base flow to the river.

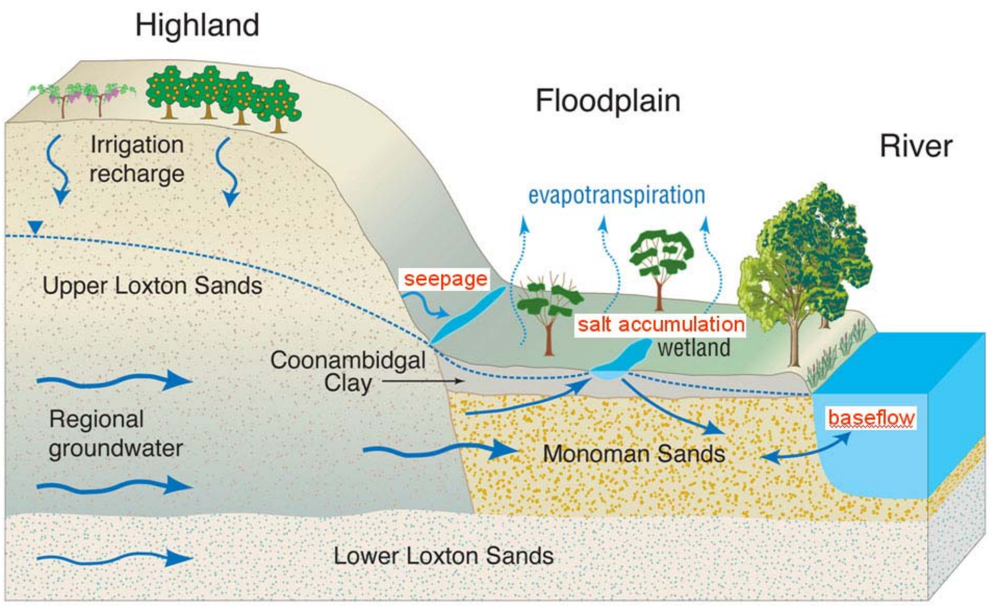

Figure 4.15 Conceptual model of groundwater inputs to the floodplain and potential groundwater discharge pathways within the floodplain (Holland *et al.*, 2009).

The following assumptions are made:

- groundwater flow within the floodplain is one-dimensional, under steady state conditions with no recharge, and the floodplain aquifer is homogenous and isotropic; and
- groundwater flow under the floodplain follows Darcy's Law (4.1), where Q is the flux of groundwater through a unit width of floodplain [L^2 T^{-1}]; K is the hydraulic conductivity of the aquifer [L T^{-1}]; b is the aquifer thickness [L]; and dh/dx is the groundwater

hydraulic gradient, where h is the height of the groundwater above river level [L] and x is the distance from the edge of the river valley [L] to a maximum distance L at the river;

$$Q = -Kb\frac{dh}{dx}; \tag{4.1}$$

- loss of groundwater through evapotranspiration, ET [L T^{-1}] can be approximated using the simple linear function described by (4.2) and (3.4); where a is the maximum rate of groundwater lost by evapotranspiration at the floodplain surface on an areal basis [L T^{-1}]. This maximum groundwater evapotranspiration rate declines to zero at z_{ext} (evaporation extinction depth [L]), which corresponds to the vegetation rooting depth, and h_f is the height of the floodplain above river level [L].

$$ET = \frac{a\left[h-\left(h_f - z_{ext}\right)\right]}{z_{ext}} \qquad h > h_f - z_{ext}\,, \tag{4.2}$$

$$ET = 0 \qquad\qquad h \le h_f - z_{ext}\text{; and} \tag{4.3}$$

- a sharp cliff and a flat floodplain characterise the shape of the river valley.

The full mathematical description of the model is given in Holland *et al.* (2009). The apportioning of groundwater flowing into the River Murray valley into seepage at the break of slope, evapotranspiration from the floodplain or base flow to the river can be categorised into four main scenarios and these are shown in Figure 4.16. The model automatically determines which scenario is occurring and solves the appropriate mathematical equations accordingly. Note that the model does not simulate situations where there are significant inflows of groundwater to the floodplain aquifer from leakage from underlying confined regional aquifers.

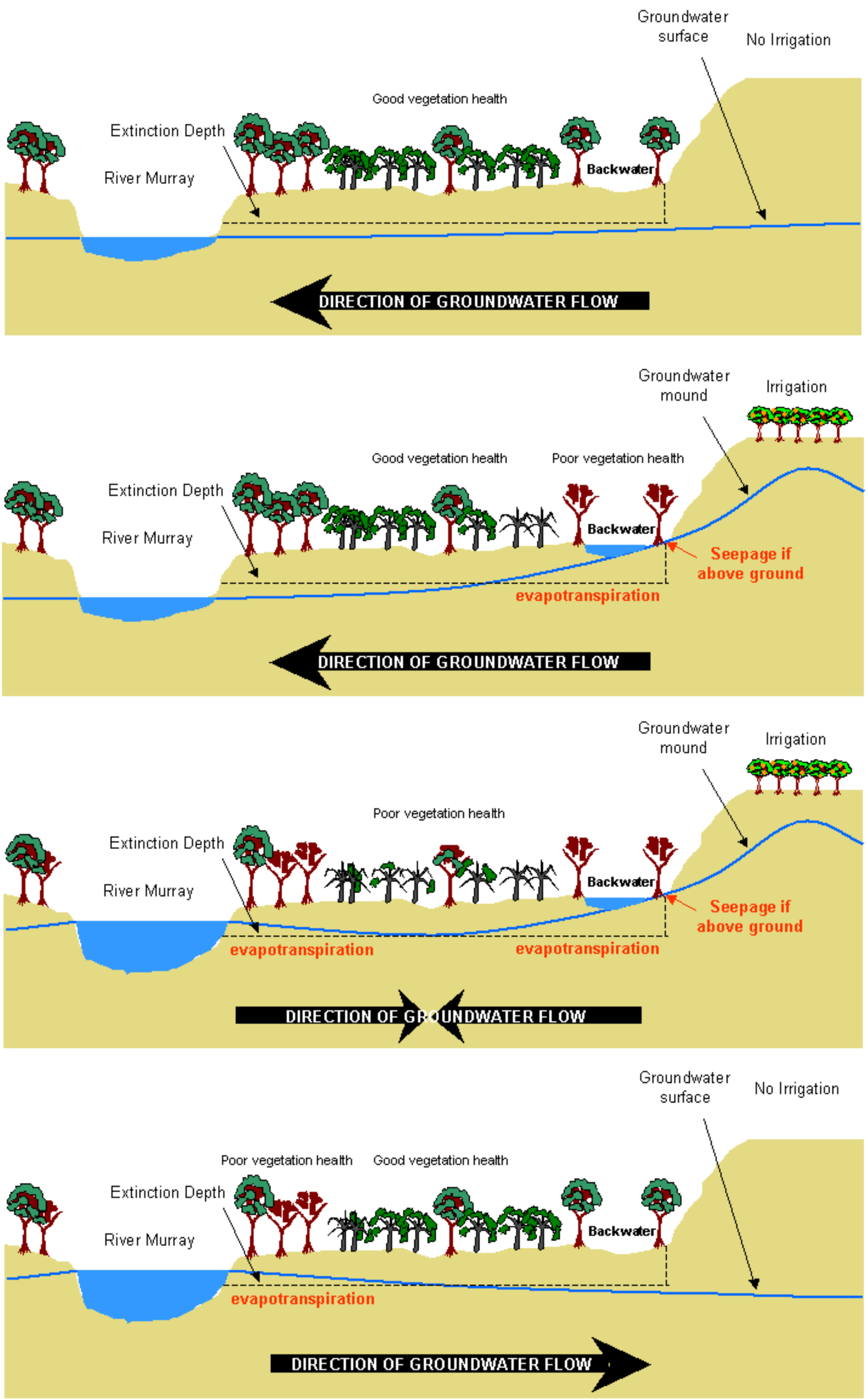

Figure 4.16 Schematic cross sections showing the different scenarios of groundwater input from the highlands and/or high river levels. (A) Downstream of a lock with no irrigation mound. (B) Downstream of a lock with an irrigation mound. (C) Upstream of a lock with an irrigation mound. (D) Upstream of a lock with no irrigation mound (Holland *et al.*, 2009).

Floodplain Impacts Model (FIP)

Generating a groundwater depth map for the whole of the floodplain requires a very precise model. However, the data required to generate an accurate groundwater depth map for the floodplain was not available. To address this, an analytical cross section model for estimating groundwater depth for the River Murray floodplain had been previously created by Holland *et al.* (2009). A spatial model based on the analytical cross section model can be developed so that a regional spatial prediction can be developed for current and future management scenarios. To apply the model spatially required using multiple cross sections and applying them to the whole of the floodplain. The method divided the floodplain into a number of small narrow divisions, to which the cross-sectional model could be applied.

The floodplain divisions included areas where there is no floodplain (i.e. where the river abuts the cliff). These divisions are necessary to calculate salt loads to the river and have been given a floodplain width of ten metres to allow the model equations to operate successfully. As discussed above, the divisions divide the floodplain into approximately 250 metre wide regions that represent the approximate groundwater flow lines. Detailed MODFLOW modelling of Clark's floodplain by Doble *et al.* (2006) has shown that at the whole floodplain scale, as long as the slice divisions follow the groundwater flow lines, the simpler Holland *et al.* (2009) model provides accurate predictions, when compared to the MODFLOW model, of the total volumes of seepage, floodplain evapotranspiration and base flow to the river. Floodplains with significant anabranch systems (such as Chowilla and Pike River) are not modelled correctly as their deep backwaters intercept a large amount of groundwater inflows (perhaps as much as 60% according to MDBC (1995) and REM (2002)).

Development of Floodplain Divisions

The analytical cross-sectional model simulates a cross section of the floodplain. Gabrovsek *et al.* (2002) developed divisions on the floodplain between Lock 4 and Lock 3 using variable width divisions and straight edges representing the shortest path between the highland and the floodplain. However, these shortest paths were considered to be insufficiently representative of the groundwater flow through the floodplain.

The development of divisions for the FIP model was carried out by using a combination of automatic and manual editing. The division boundaries were developed by dividing the floodplain boundary edge into 250 metre divisions. This was achieved by automatically creating a node along the floodplain boundary at 250 metre intervals. At each of these nodes, the edge of the division was derived using an elevation model of the surface of the groundwater, automatically generating 'flow' lines from the node at the height of the floodplain edge to the river at pool level. In many cases the methodology caused errors due to the inaccuracy of the elevation model requiring changes to the divisions by editing the line boundaries.

The process developed 3,727 divisions of 250 metres width at the outer floodplain edge to varying width at the river's edge depending on the geometry of the floodplain. A typical division and the parameters used in the FIP model are shown in Figure 4.17 and a section of the floodplain showing the actual divisions is given in Figure 4.18.

The divisions were then analysed for floodplain width which is the division length by calculating the average of two sides of the division. The division width was calculated by the distance between the points half way along the edge of the division.

A highland division for each floodplain division was calculated as the area that extended on from the floodplain division until the first groundwater contour. These highland divisions were used to calculate the rate of groundwater inflow into the highland.

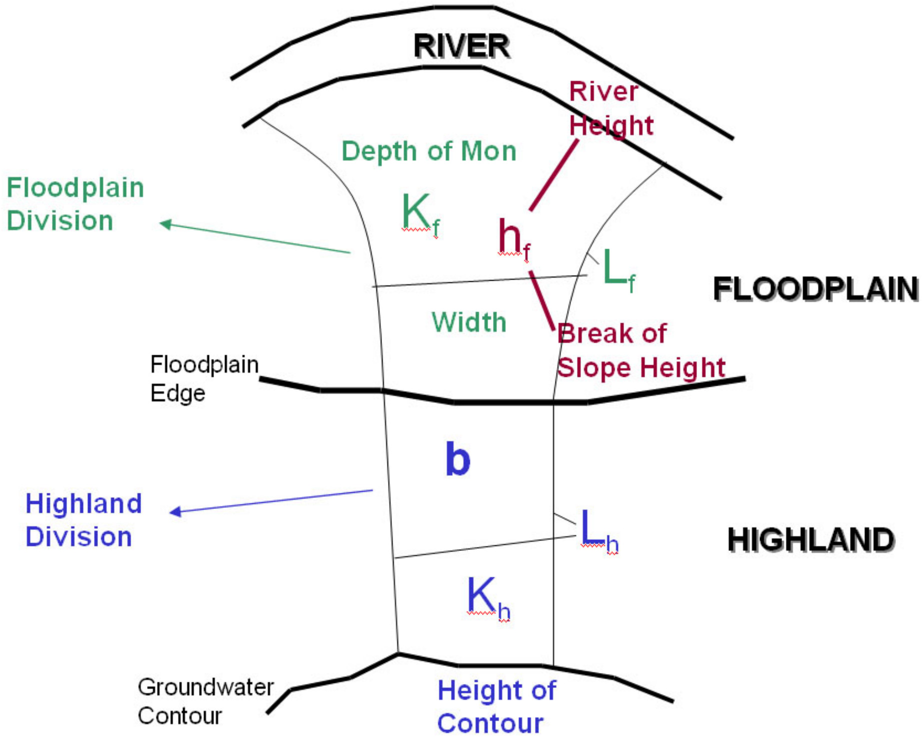

Figure 4.17 Diagram of a division showing the parameters used in the FIP model. Parameters of the model are explained in the text and Table 4.7.

Each division was assigned a river kilometre marker that was nearest. In the Chowilla and Pike floodplains there are major anabranch creeks which can intercept groundwater inflow into the floodplain before it meets the main channel. This intercepted groundwater then enters the river at the location where the anabranch creeks join back downstream.

A survey of river salinity at one kilometre intervals along the river is regularly undertaken by the South Australian Government (Porter, 2001). This run-of-river data is useful for identifying locations of high salt intrusion into the river. To compare run-of-river salinity levels and model predictions it was necessary to assign all divisions with these two anabranch creeks as the river kilometre downstream. Figure 4.17 and Table 4.7 show the range of parameters that are used in the FIP model.

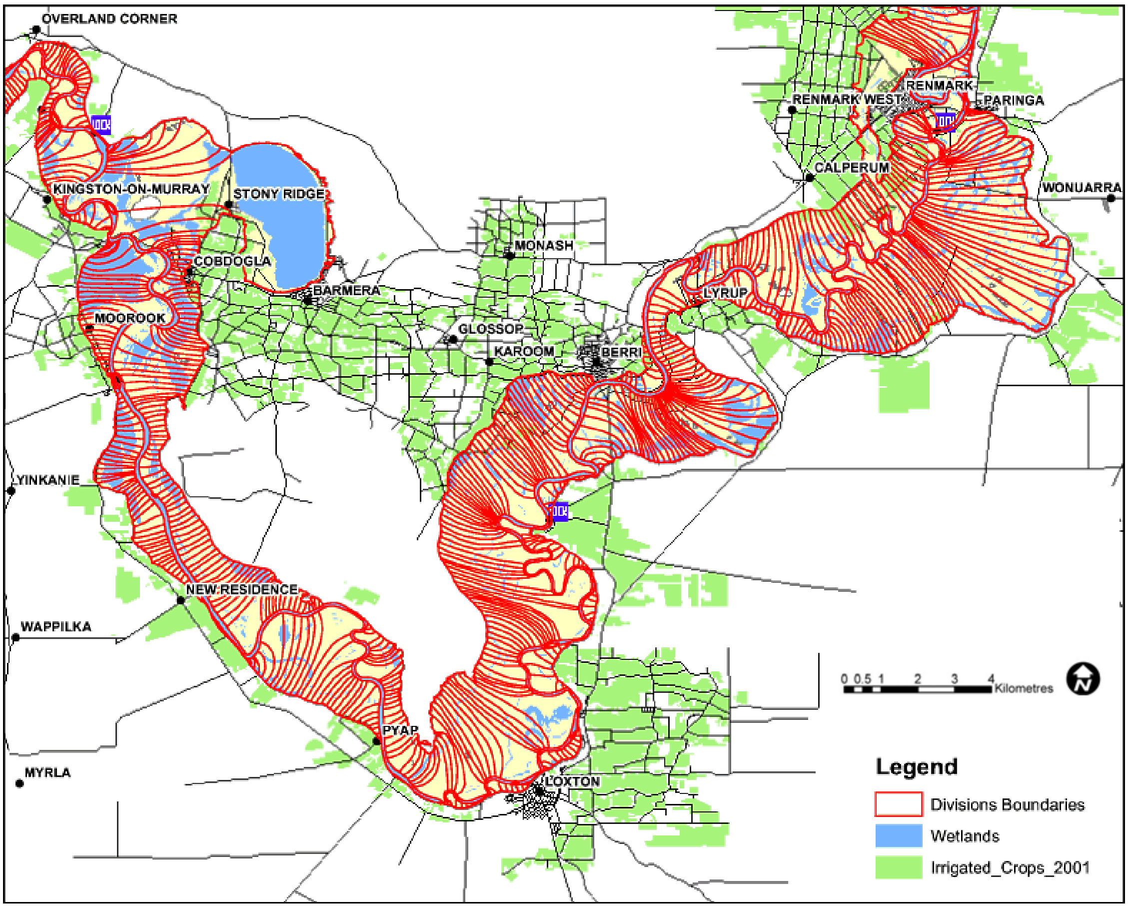

Figure 4.18 Map of part of the lower River Murray showing the floodplain divisions used to implement the spatial analytical model.

Table 4.7 Division coverage fields and descriptions.

Field	Example Value	Description	Units
DIVISIONID	1956	Unique division ID. The river is broken into 3727 divisions	
REACH	3	Lock reach. In this example lock reach 3 to 4	
AREA	1523440	Area of the division	m^2
PERIMETER	7964	Perimeter of the division	m
EDGE_L	267	Length of division at the edge of the floodplain. Determines the amount of inflow from the per metre rate below	m
COMB_L	3085	Width of the division (L in analytical model). Calculated from the average of the two sides	m
RIV_KMS	474	River kilometre marker from 0 at the mouth to 657 at the border. Exceptions are divisions in Pike and Chowilla which have been assigned river km markers downstream were the anabranch creeks enter the main channel	
RIV_ENT	985	River height at entitlement flow in AHD	cm
RIV_100	1315	River height at a 100,000 ML/day flow or 68,000 ML/day flow	cm
RIV_1956	1660	River height at the 1956 flood (~320,000 ML/day)	cm
HF	330	Difference between heights of river and break of slope (choice of RIV_100 or RIV_1956)	cm
K	2	Transmissivity (horizontal hydraulic conductivity)	m/d
B	7	Aquifer thickness below river level	m
Z_EXT	5	Depth at which evapotranspiration occurs. Constant defined by user	m
A	0.0001	Maximum transpiration rate. Constant set by user	m3/d
OBSERVED	0	Observed seepage from field and anecdotal evidence	
SIMPACTSAL	11550	Salinity of groundwater from SIMPACT	mg/L
REVINFL_M	0.0600	Inflows determined from bore observations and proximity of irrigation areas	$m^3/d/m$

Model Parameters and Spatial Data

The first dataset required was groundwater inflow rate into the floodplain. Initially the model used floodplain inflows predicted from the large-scale regional MODFLOW groundwater models (Barnett *et al.*, 2002). However, these proved to be inaccurate for determining inflows at the 250 metre intervals required for each floodplain division, based on known problems and gaps between models.

Instead, groundwater inflows to the floodplain from the regional aquifers beneath the highland that fringes the floodplain were calculated using highland aquifer parameters and an interpolated highland groundwater surface using Darcy's Law. The Department of Water, Land and Biodiversity Conservation (DWLBC) measured groundwater levels in a network of monitoring bores in 2000. These were interpolated to provide a groundwater depth surface for the regional aquifers beneath the highland areas adjacent to the floodplain. Bore numbers are limited over such a large area and known groundwater mounds were not all modelled sufficiently using this interpolation method. The inflows were adjusted based on the extent of irrigation adjacent to the floodplain in 2001, which provides a visual indication to the magnitude of groundwater inflows. Inflow values were increased in areas of extensive irrigation to better calibrate model predictions with measured river salt loads and seepage areas.

The second dataset required was groundwater salinity. An interpolated surface of groundwater salinity was obtained from the SIMPACT model (Miles *et al.*, 2001). This model interpolated data from the SA-GEODATA Drill hole Database and represents the salinity of the regional aquifers beneath the highland, but not the floodplain aquifer.

Other required data included the horizontal hydraulic conductivity or transmissivity of the floodplain aquifer. This value for the regional highland aquifers is known to vary in the range 1-5 m/day (Barnett, 1991). However, insufficient data is available to adequately characterise the spatial variation and so a commonly accepted value of 2 m/day was chosen.

The height of the river and the elevation of the edge of the floodplain were also required. River levels at each kilometre interval along the river were obtained from the RiM-FIM (Section 4.3) for a river at entitlement flow. The river heights were based on recorded and modelled backwater curves and are presented in Figure 4.19. The height of the floodplain at the edge of the river valley was determined from the river height of a 100,000 ML/day flood event using the RiM-FIM (Figure 4.19). This value was used to determine the maximum groundwater elevation at the edge of the floodplain before seepage occurs and is compared to the river height to obtain h_f for the analytical model. There are some problems in using these flood levels to determine the floodplain edge, as they will vary, although no actual elevations are available for this 'break of slope' edge.

The maximum transpiration rate of vegetation and soil limited groundwater discharge rate determines the rate of evapotranspiration. This varies according to vegetation and soil type, with species such as river red gum having higher transpiration rates than black box. Initial model calibrations used a single evapotranspiration rate of 0.1 mm/day consistent with the recharge mapping in Chapter 5.

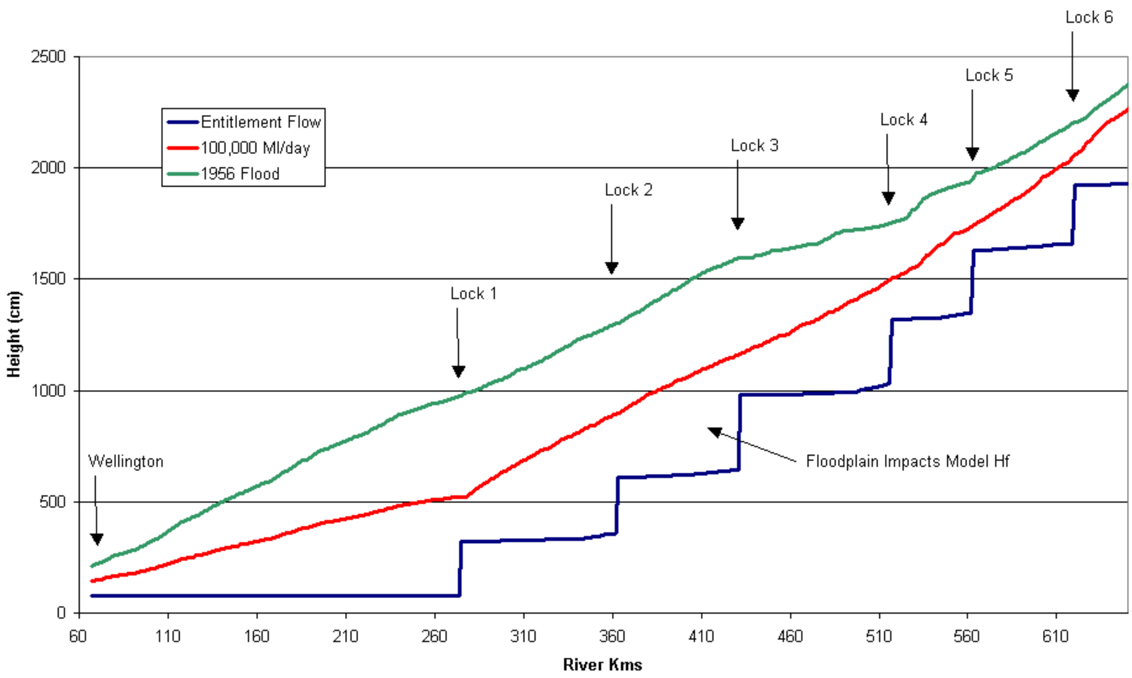

Figure 4.19 Graph of river heights for entitlement flow, 100,000 ML/day and the 1956 flood. The FIP model uses the entitlement flow for river levels and the 100,000 ML/day heights for floodplain edge height.

To validate the model three variables were required; salt loads, seepage and floodplain salinisation risk. The Run-of-river salinity values were used to validate the base flow salt load portion of the FIP model. A map of areas where seepage was observed at the break in slope was developed from field work and anecdotal evidence.

To calibrate the floodplain salinisation risk from evapotranspiration, and therefore vegetation health, component of the model, the 1:10,000 vegetation community maps presented in Chapter 3 were initially used. The maps were the only reliable vegetation information at the time and covered approximately 40% of the Lock 4 to Lock 3 reach. Once the DEH vegetation mapping was complete this was used to validate the FIP model.

Development of the GIS Tool

Holland (2002) initially implemented the mathematical equations for the model in a spreadsheet. These equations were then implemented in a GIS using a customised programming language to provide predictions for each of the floodplain divisions, thus providing a full spatial implementation of the Holland *et al.* (2009) analytical model.

Table 4.8 Division output values calculated from the parameters in Table 7.1 and the analytical model equations.

Field	Example Value	Description	Units
DIVISIONID	1956	Unique division ID. The river is broken into 3727 divisions	
Q0	0.0535	Qo (analytical model) (K x B x HF)/L	
QONET	0.0211	Qonet (analytical model) (K x B x (HF - Zext))/L	
ERROR	0	Scenario error value in analytical model (0 - no error, 1 or 2)	
SCENARIO	2	Scenario calculated in analytical model (1, 2, 3 or 4)	
QS	0.0000	Seepage rate	$m^3/d/m$
QET	0.0260	Evapo-transpiration rate	$m^3/d/m$
QR	0.0340	Base flow rate to river	$m^3/d/m$
TONNES	0.105	Salt load to river	tonnes/d
XCRIT	0.379	Percentage of division with ET	%
ATTENPERCE	0.434	Percentage of inflows attenuated by floodplain (Qs + Qet/inflows)	%

The analytical cross-section model equations were run for each division within the GIS and produced results that were stored within the GIS database. Table 4.8 shows the output variables generated.

The rate of seepage (QS) can be compared with the presence or absence of seepage known for that division under current inflows and can be used to predict seepage under future scenarios. The TONNES field is calculated from the rate of base flow (QR) and groundwater salinity and predicts salt loads to the river which can be compared to current river salinities with current inflows. QET gives the predicted rate of evapotranspiration which can be compared with potential evapotranspiration of the vegetation. High values of evapotranspiration mean that water is moving up through the soil profile which means salt can accumulate to cause salinisation and therefore risk to the vegetation.

A useful way of interpreting the rate of evapotranspiration was found to be the percentage of the division with high ET expressed in the XCRIT value. High XCRIT implies most of the floodplain in that division is at risk of salinisation. The percentage of inflows that are attenuated by the floodplain (QS and QET/Inflows) is another useful value for looking at the spatial variability in floodplain attenuation and the areas of floodplain that are most at risk from retaining groundwater.

A user interface was developed for the spatial FIP Model (Figure 4.20) to calculate the results from user defined scenarios. The results of the calculations are presented in a map in the FIP model (Figure 4.21) showing the areas of XCRIT classed into three classes of high, medium and low. The model is able to predict seepage, evapotranspiration and base flow from changing rates of inflow, extraction depth, aquifer thickness, hydraulic conductivity and maximum transpiration rate.

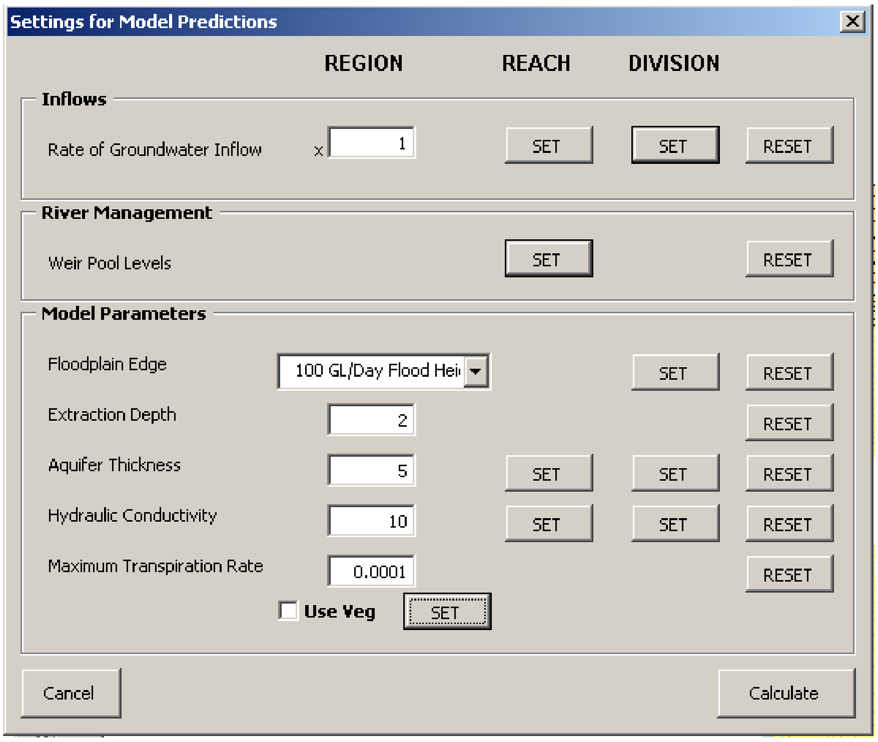

Figure 4.20 The FIP model input screen to set values for current and future scenarios.

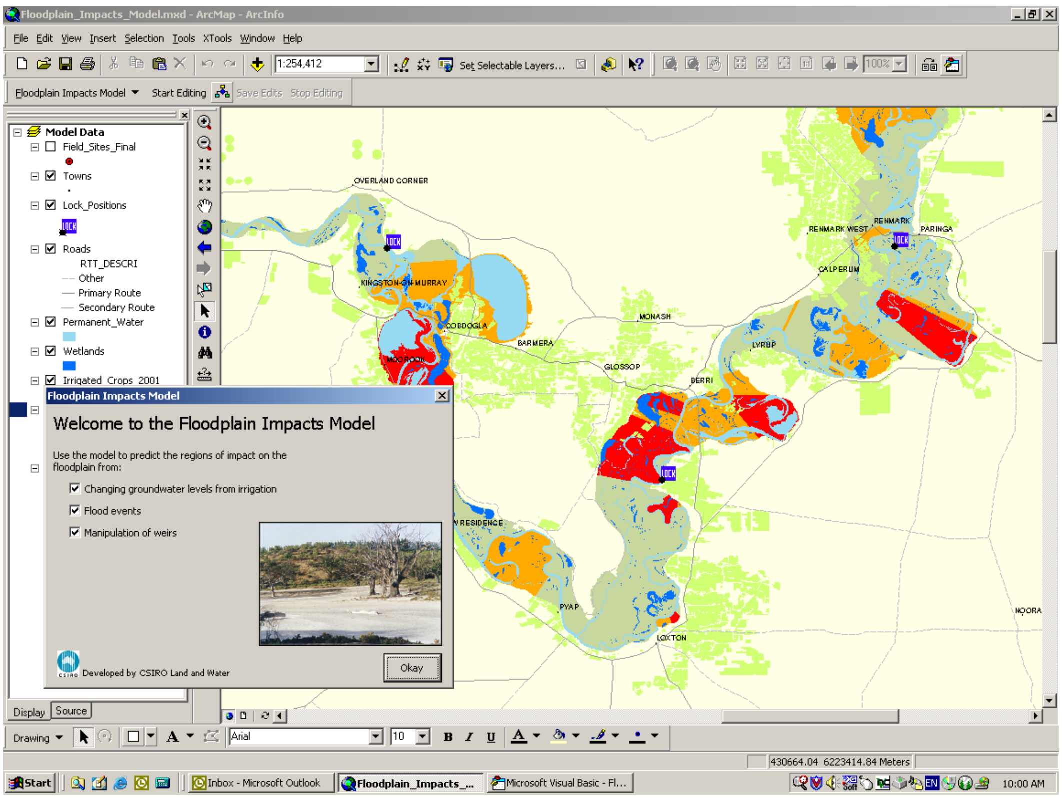

Figure 4.21 The FIP model in the GIS showing an output map of XCRIT classes.

Results of the FIP Model

Preliminary results from the model under current groundwater inflow conditions are shown in Figures 4.22 and 4.23. Areas at risk from seepage (red) and salinisation (orange) based on XCRIT are depicted. The model predictions of areas at risk from seepage and salinisation are summarised in Table 4.9.

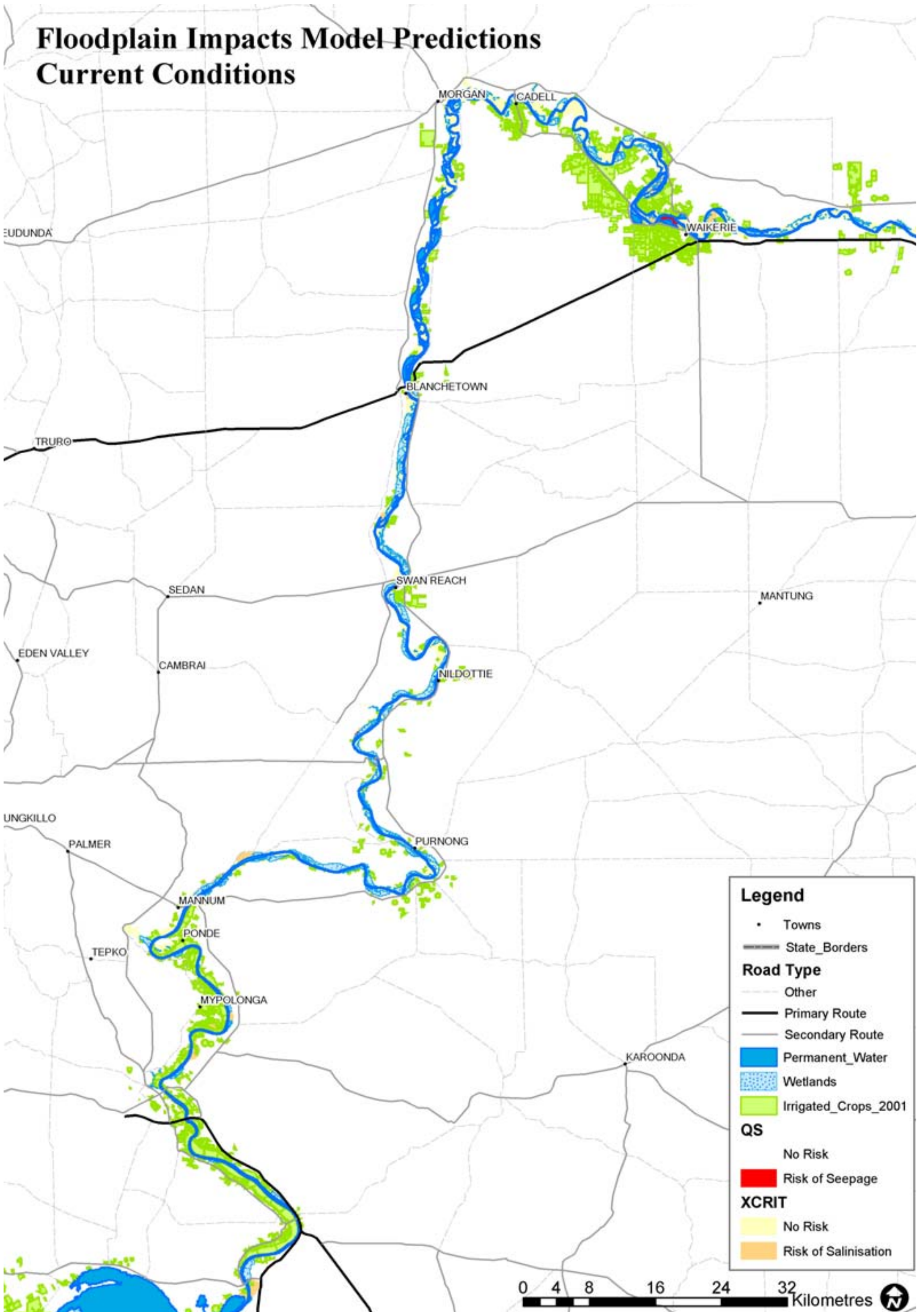

Figure 4.22 Predictions of floodplain risk for the lower River Murray in South Australia (southern section) under current inflow conditions. Areas of salinisation are those floodplain divisions were some or all of the floodplain has groundwater within 2 m of the modelled surface.

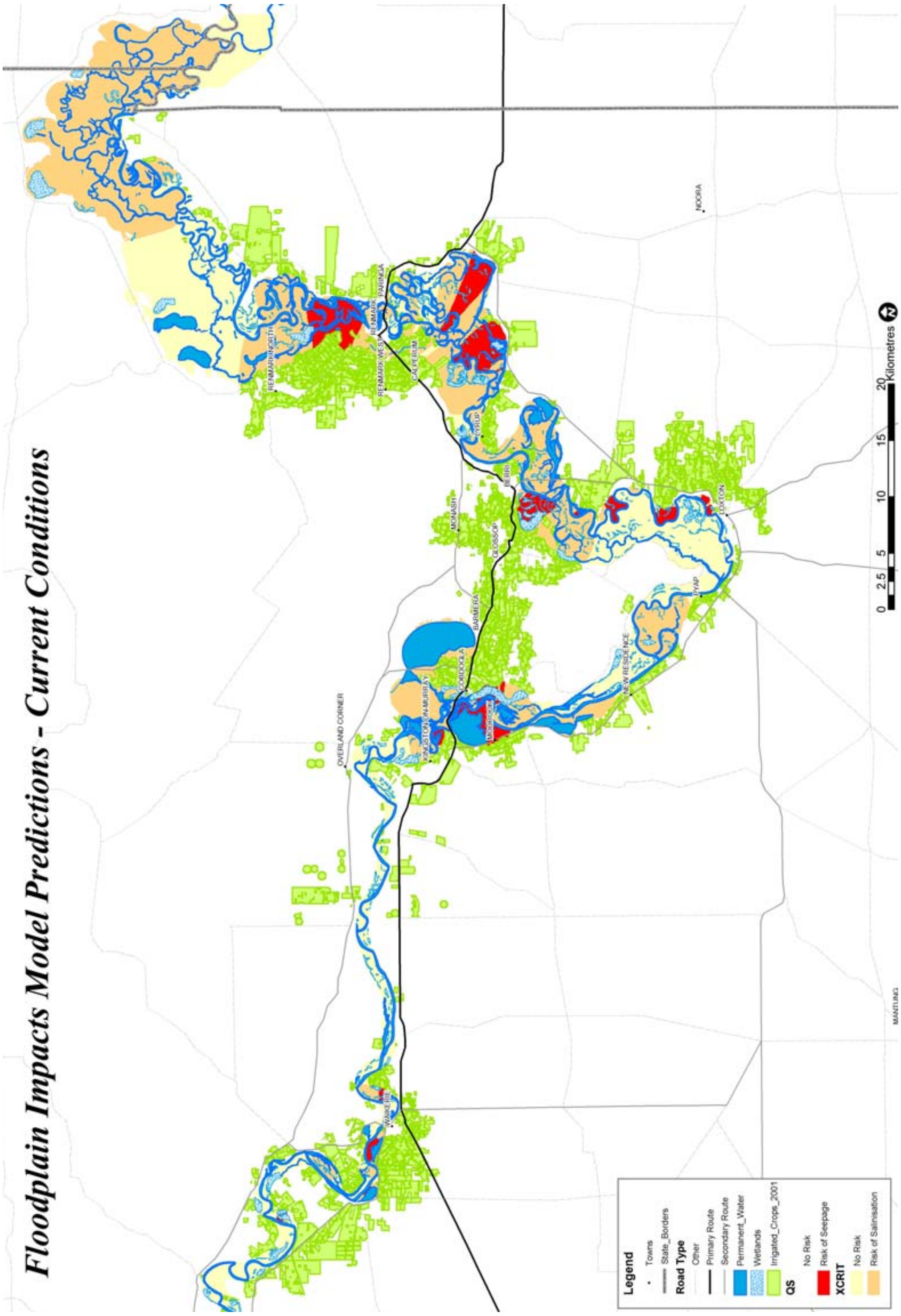

Figure 4.23 Predictions of floodplain risk for the lower River Murray in South Australia (northern section) under current inflow conditions. Areas of salinisation are those floodplain divisions were some or all of the floodplain has groundwater within 2 m of the modelled surface.

Table 4.9 Summary of preliminary model predictions for the lower River Murray in South Australia.

	Number of divisions	Area of floodplain (ha)	Length of floodplain (km)
Total	3736	119,900	1,205
Current inflows causing seepage	105 (3%)	5,100 (4%)	28 (2%)
Current inflows causing salinisation	1,116 (30%)	47,200 (40%)	347 (29%)

Floodplain attenuation is the percentage of salt inflow that the floodplain retains and therefore does not reach the river. The FIP predictions in Table 4.9 highlight that, while the average floodplain attenuation is around 30% (as assumed by Barnett *et al*., 2002), there is considerable spatial variability. This needs be taken into account when using regional groundwater models to predict river salinity. The degree of attenuation at any given location is dependent on the rate of groundwater inflow and the floodplain parameters that control the rate of evapotranspiration and seepage. Floodplain geometry, most notably width, floodplain aquifer depth and conductivity, evapotranspiration rate and river height are the major factors affecting attenuation percentages.

Figure 4.24 shows the spatial variability in floodplain attenuation. Areas upstream from the locks have high floodplain attenuation as the river is high in these areas, reducing the flow of groundwater to the river as base flow and increasing the height of the groundwater closer to the surface to increase salinisation. Areas further upstream from the locks have variable attenuation, which increases as floodplain width increases for those areas adjacent to an irrigation area. This has implications for the location of irrigation areas which would be better placed near narrow floodplains to provide a lower floodplain salinisation risk, but next to wide floodplains to lower the risk of high salt loads to the river. Seepage areas can occur in medium and high floodplain attenuation floodplains, as they are a function of the rate of groundwater inflow and river levels (Figure 4.25).

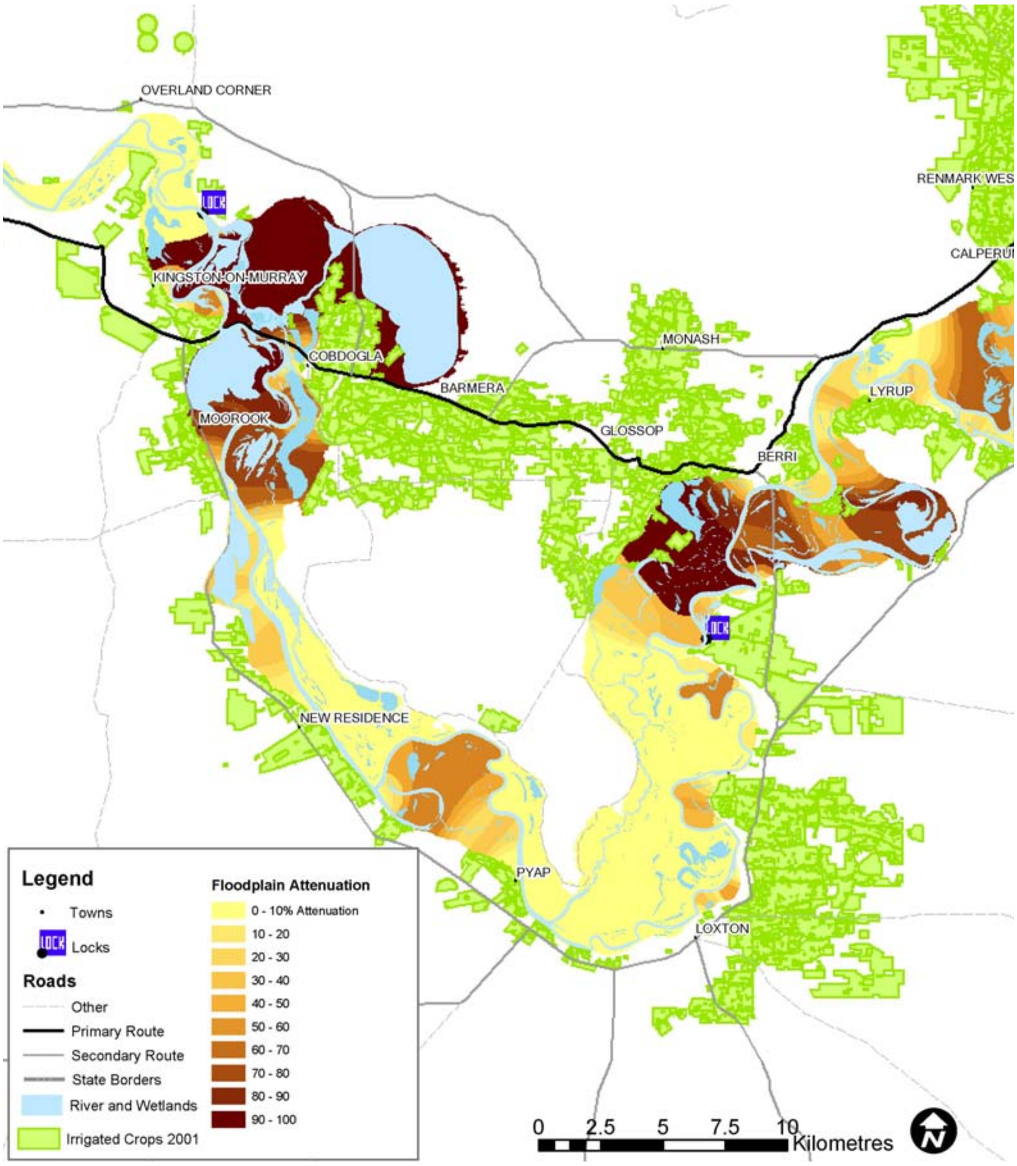

Figure 4.24 Predictions of the floodplain attenuation percentage for the area between Lock 3 and 4 under current inflow conditions. The degree of attenuation can be seen as a function of locking, floodplain width and of the magnitude of groundwater inflow.

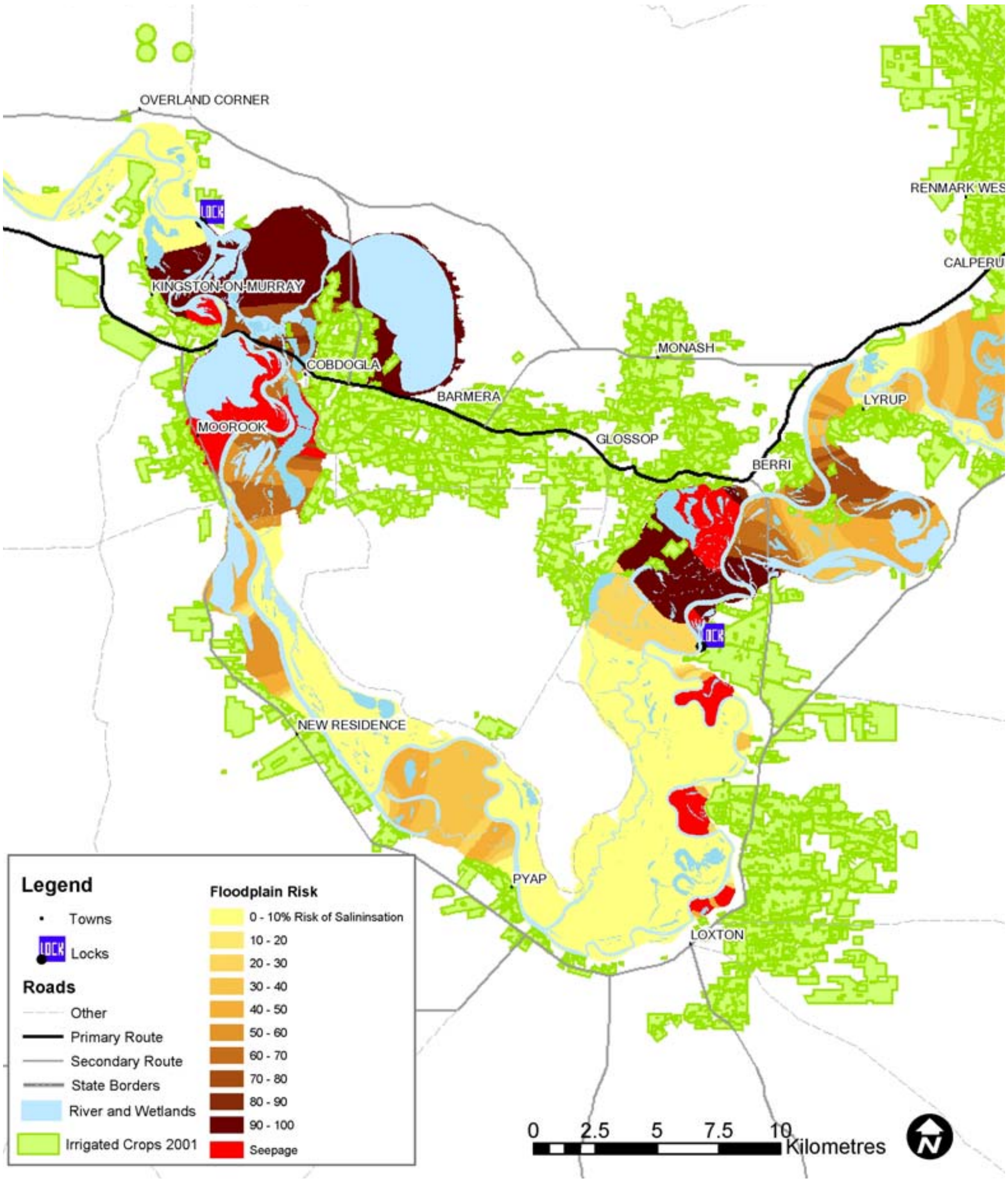

Figure 4.25 Predictions of floodplain risk of salinisation for the area between Lock 3 and 4 under current inflow conditions. Areas of salinisation have been given a gradational colour based on the XCRIT value.

An understanding of the salt loads in the River Murray is necessary to calibrate models of groundwater accession by the floodplain and to investigate the surface-groundwater interaction with the increased salt loads from flooding.

Model Validation

The run-of-river salinity values (Porter, 2001) recorded salt load into the river at a one-kilometre intervals along the river. An example of a portion of the data is given in Figure 4.26. An interesting aspect of the 2001 run-of-river data that was uncovered when visualised in this way, was that the highest indicated inflows were up to five kilometres downstream from irrigation areas (see for example downstream from Loxton irrigation area suggesting that the highest salt concentrations can be downstream from where the inflows are emanating. A potential cause for this could be that groundwater intrusion occurs at the bottom of the river and it would then take that distance before the salt load was measured in the top of the river profile. For this reason, a shift in the river distance between model outputs and run-of-river data needs to occur.

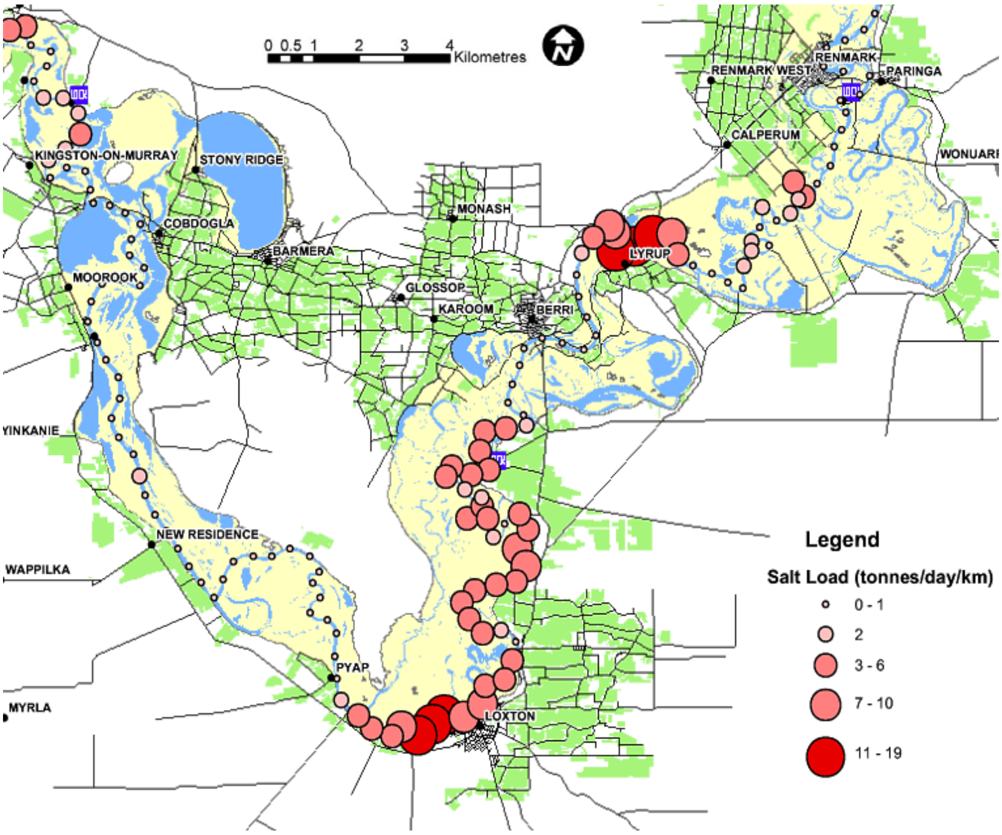

Figure 4.26 Map of part of the lower River Murray showing the spatial representation of the run-of-river salinity data.

The model was validated by comparing predictions of base flow salt loads (TONNES) to the river against run-of-river salt load data from 2001 which covered the whole length of the river in South Australia (Porter, 2001). Figure 4.27 shows the comparison of the run-of-river data and the model predictions under current conditions. There is a good comparison between the two datasets with two notable exceptions. The areas around Wachtels Lagoon and Ramco Lagoon have much higher predictions of salt load to the river than observed in the run-of-river data. These two areas are the only large permanent wetland complexes upstream of a lock that are likely to be in good hydraulic contact with the floodplain groundwater system. It is expected that they act as large salt storages with salt only being released during high flows. The storage of approximately 58 tonnes/day in these two areas has been removed from the graph in Figure 4.27.

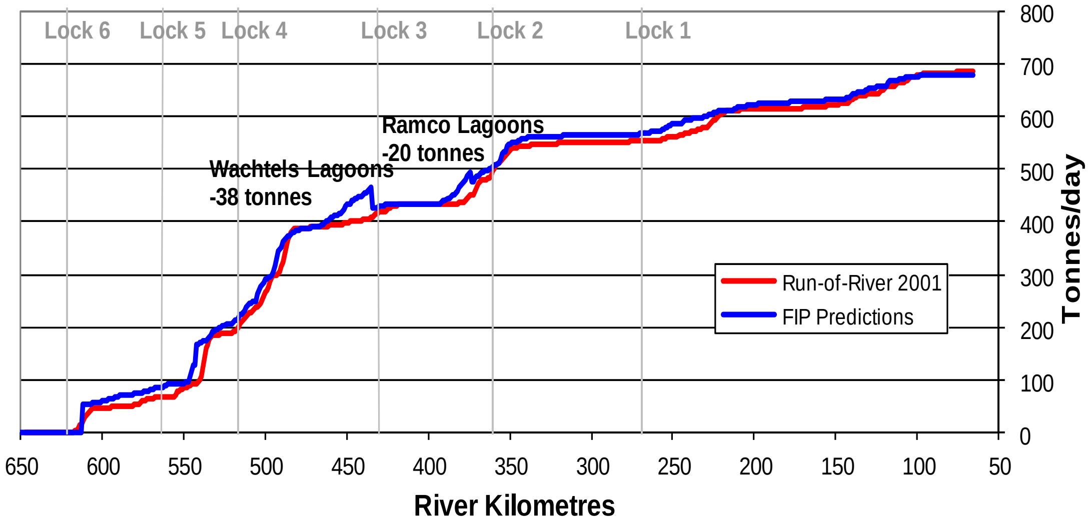

Figure 4.27 Comparison of FIP model predictions of cumulative salt loads to the river versus the 2001 run-of-river survey.

The model predictions of salt loads to the river for each reach can be compared with reported salt loads (Barnett *et al.*, 2002 from 1995 values). These values were based on an average 30% attenuation of groundwater inflows (Figure 4.28). The FIP model results closely follow the published estimates and run of river values, with the exception of Lock 1-2, Lock 2-3 and Lock 5-6, where the published salt loads are much greater. FIP salt loads are comparable to run of river, but higher than the published estimates in the Wellington-Lock 1 and Lock 3-4 reaches.

A similar comparison between published salt loads to the river valley divided into Land and Water Management Plan (LWMP) areas (Figure 4.29) shows that estimates for most LWMP areas are in agreement. However, in several areas the FIP predictions were 2 to 30 times smaller than the published values (Cadell, Pike River, Riverland Norwest and Woolpunda). These large variations in estimates of salt loads to the river valley result in the large differences in estimates of salt stored in the floodplain. Floodplain attenuation of salt loads varied between a minimum of 0% and maximum of 67% (FIP) and a minimum of 0% and maximum of 97% (AWE, 2003) (Table 4.10).

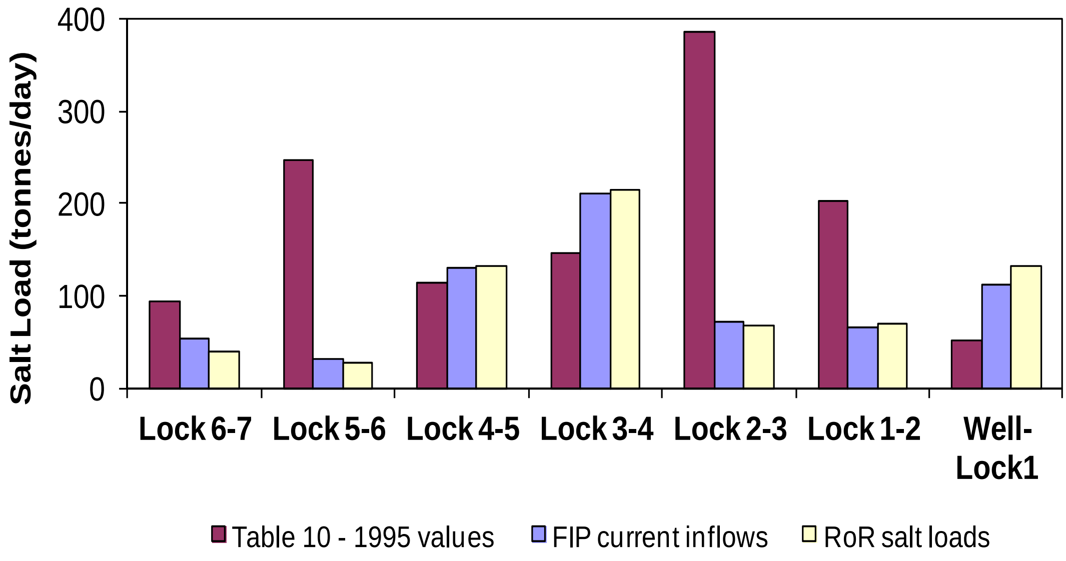

Figure 4.28 Total salt loads to the river for each lock reach showing comparison between reported values that assumed 30% attenuation (Barnett *et al.*, 2002 data from 1995),FIP results, and run-of-river values for 2001.

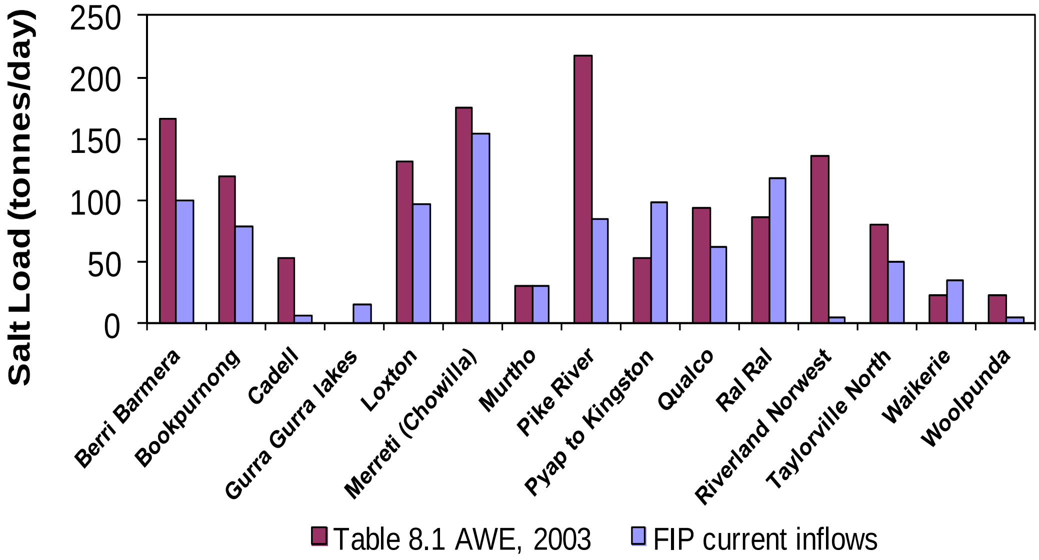

Figure 4.29 Total salt loads to the river valley for each Land and Water Management Plan Area showing comparison between reported values (AWE, 2003) and FIP results for 2003.

Testing of the model was also carried out by comparing those locations where seepage at the break of slope was predicted to occur and was in fact observed. These areas showed strong correlation even with other factors such as presence of clay layers at the edge of the floodplain that can influence the occurrence of seepage.

Floodplain vegetation health data discussed in Section 5.2 (Smith and Kenny, 2005) was used to validate the prediction of evapotranspiration. Understorey species were not rated for health but the presence of halophytic species was used to indicate the presence of salt. Holland *et al.* (2009) defined the halophytic species as *Halosarcia*, *Sarcocornia*, *Pachycornia* and *Suaeda* species. A total of 41% of the River Murray floodplain in South Australia was therefore mapped as poor health or salinised vegetation. This compares well with the 40% predicted from the FIP model (Table 4.9) for salinised floodplain. Given the localised variability of vegetation health, it is difficult to spatially compare results of the FIP model with vegetation health at the spatial resolution of the vegetation mapping. The FIP model is most useful in indicating the potential for floodplain risk.

Table 4.10 Summary of preliminary model predictions for the lower River Murray in South Australia divided into Land and Water Management Plan areas. Salt loads and floodplain attenuation percentages are compared to reported figures (AWE, 2003).

LWMP Area	RoR salt loads (tns/day)	Salt load to river (tns/day)	Salt load to river (tns/day)	Salt load to floodplain (%)	Salt load to floodplain (%)
	2001	AWE, 2003	FIP	AWE, 2003	FIP
Berri Barmera	55	15	54	92%	46%
Bookpurnong	58	66	69	45%	13%
Cadell	6	5	7	91%	0%
Gurra Gurra lakes	6		7		56%
Loxton	89	131	88	0%	10%
Merreti (Chowilla)	37	51	54	71%	65%
Murtho	25	17	25	45%	17%
Pike River	46	86	44	61%	48%
Pyap to Kingston	40	20	36	62%	63%
Qualco	63	94	60	0%	3%
Ral Ral	42	26	52	70%	56%
Riverland Norwest	5	4	4	97%	0%
Taylorville North	37	7	50	90%	1%
Waikerie	9	11	12	52%	67%
Woolpunda	16	19	5	22%	0%
Outside LWMP areas	151		116		11%
TOTAL	685	552	681	57%	30%

Model Limitations

A lack of field data to calculate groundwater inflows led to estimates from groundwater interpolation contours and proximity to irrigation areas and are therefore subject to some uncertainty. Transmissivity and aquifer thickness also had to be estimated and an average value was used for the entire lower River Murray. Refinements of these three values could improve the model.

Other limitations are related to the analytical cross-sectional model (Holland and Walker, 2003) and include the assumptions that the floodplain surface is flat and that the relationship between evapotranspiration and groundwater depth is linear and declines to zero at the extinction depth z_{ext}. Both assumptions are due to mathematical limitations in analytically solving the governing equations, and both impact on the selection of z_{ext} for each floodplain division.

The surface elevation plays a critical role in the health of the vegetation as well as determining the presence of seepage and the percentage of groundwater reaching the floodplain. The best surface elevation data available is from the RiM-FIM. The availability of a detailed digital elevation model could be used to improve the FIP model by improving the surface elevation at the break of slope of the floodplain.

The model does not attempt to simulate floodplain wetlands, backwaters or oxbows, which occur as the river meanders across the floodplain. This is because the role that these water bodies play in intercepting saline groundwater flowing towards the river is unknown, and a means of quantitatively accounting for these effects needs to be developed. The algorithms have subsequently been developed in a Floodplains and Wetlands Impact Model (FWIP) (Holland *et al.*, 2009).

In addition, the model is not applicable to areas with irrigated floodplains, in particular the dairy flats downstream of Mannum. These floodplains do not have large areas of native terrestrial vegetation. They also do not appear to be undergoing salinisation to any degree, because the low irrigation efficiency (i.e. high drainage) provides a means of keeping soil salinities low, and freshening the underlying groundwater.

4.4.3 Modelling Floodplain Tree Health from Groundwater Management Scenarios

The FIP model developed in Section 4.4.2 was used to predict areas at risk from future development and management scenarios. The following figures (Figures 4.30, 4.31 and 4.32) show the prediction of areas at risk from seepage and salinisation with an increase in inflows from increased irrigation or increased regional inflow due to vegetation clearing in the Mallee. Management scenarios to decrease inflows and lower the floodplain water table can improve the condition of the floodplain, reducing the risk of seepage and salt accumulation. A summary of the areas at risk predicted in these scenarios, compared to those predicted for current groundwater inflow conditions, is given in Table 4.11.

Table 4.11 Summary of the areas at risk for the lower River Murray in South Australia.

	Salt load (tonnes)	Divisions with seepage risk	Area of floodplain with seepage risk (ha)	Length of floodplain with seepage risk (km)	Divisions with salinisation risk	Area of floodplain with salinisation risk (ha)	Length of floodplain with salinisation risk (km)
Current inflows	760	105	5,100	28	1116	47,200	347
Increase of 20%	888	127	6,300	33	1162	48,400	359
Decrease of 40% (85% efficiencies)	481	21	1,200	5	1000	43,000	316
Groundwater 1 m lower	829	57	2,900	14	467	34,800	163

Figure 4.30 shows the predicted risk areas if all groundwater inflows to the floodplain were increased by 20%. This could occur if a large expansion in the area of irrigation was to take place as a result of small increases in irrigation areas spread along the River corridor or an increase in inflows as a result of large land clearing in the Mallee. In reality any new irrigation is likely to be concentrated in particular areas, such as the new development occurring in the Murtho area south of Chowilla.

Figure 4.31 shows predicted risk areas if inflows were decreased by 40%. This could occur from an improvement in irrigation efficiencies over the whole area currently irrigated. An increase in irrigation efficiency from a hypothetical current average of 75% to an average of 85% would decrease the inflows by 40%.

Figure 4.32 shows predicted risk areas if the floodplain water table was lowered by 1 metre. This could occur by lowering the weir pool level by 1 metre or by lowering the floodplain aquifer water table using interception schemes. Management scenarios have the effect of redistributing the ratio of salt stored in the floodplain to salt load to the river. A scenario that reduces salt loads (decrease of 40%) is not the best option for salinisation, whereas lowering the groundwater more than halves the number of divisions with seepage risk. This reduction however affects the smaller divisions more, as reflected by the less than 50% reduction in area of seepage risk from 47,200 ha to 34,800 ha, and it increases the salt load (Table 4.11).

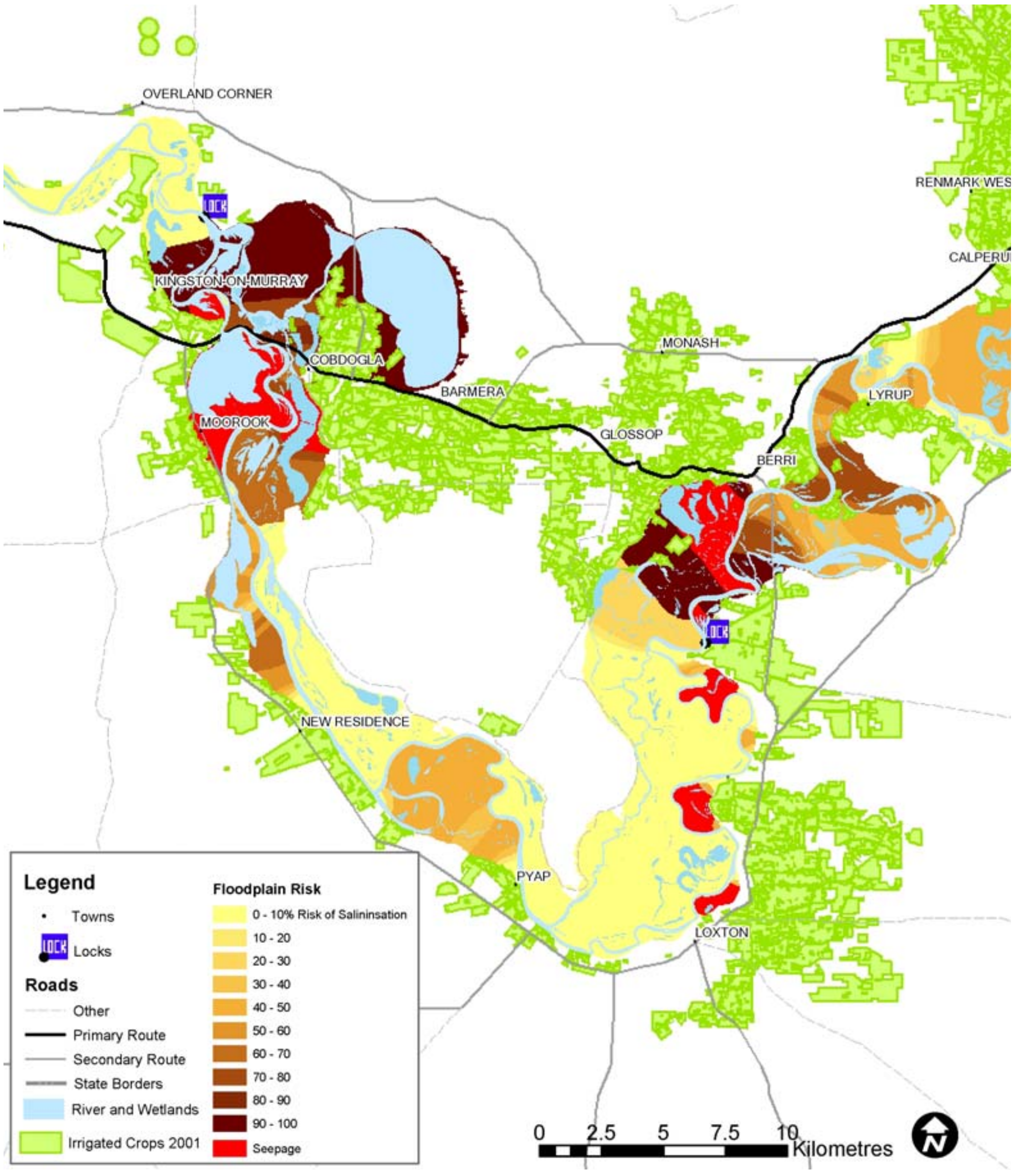

Figure 4.30 Predictions of floodplain risk for the area between Lock 3 and 4 under a 20% increase in current inflows due to increased irrigation. Areas of salinisation have been given a gradational colour based on the percentage of floodplain with groundwater within 2 m of the modelled surface.

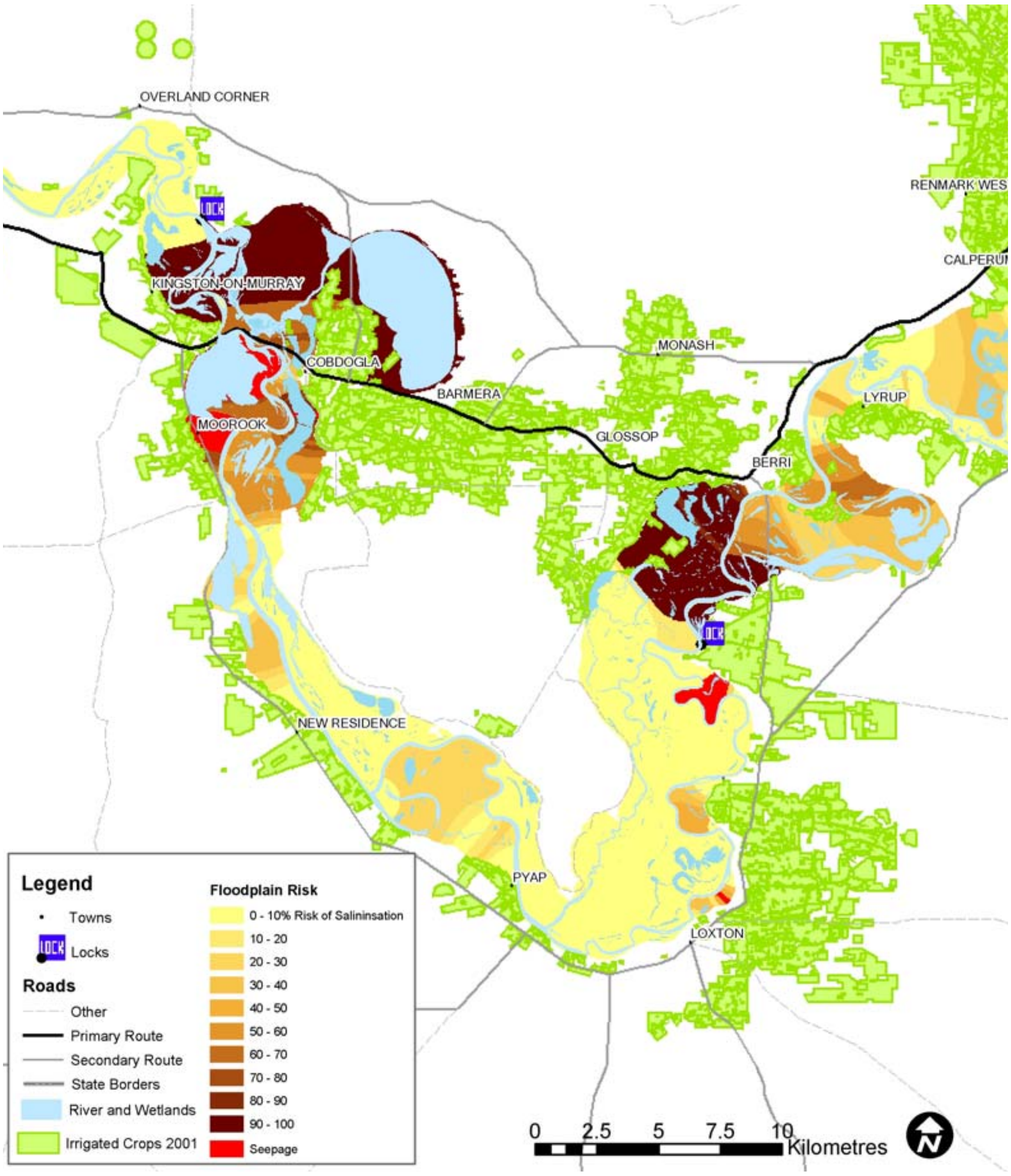

Figure 4.31 Predictions of floodplain risk for the area between Lock 3 and 4 under conditions were irrigation practices have been improved to have 85% efficiencies rather than the current average of 75%. The improved efficiencies equate to a reduction in the current inflows of 40%. Areas of salinisation have been given a gradational colour based on the percentage of floodplain with groundwater within 2 m of the modelled surface.

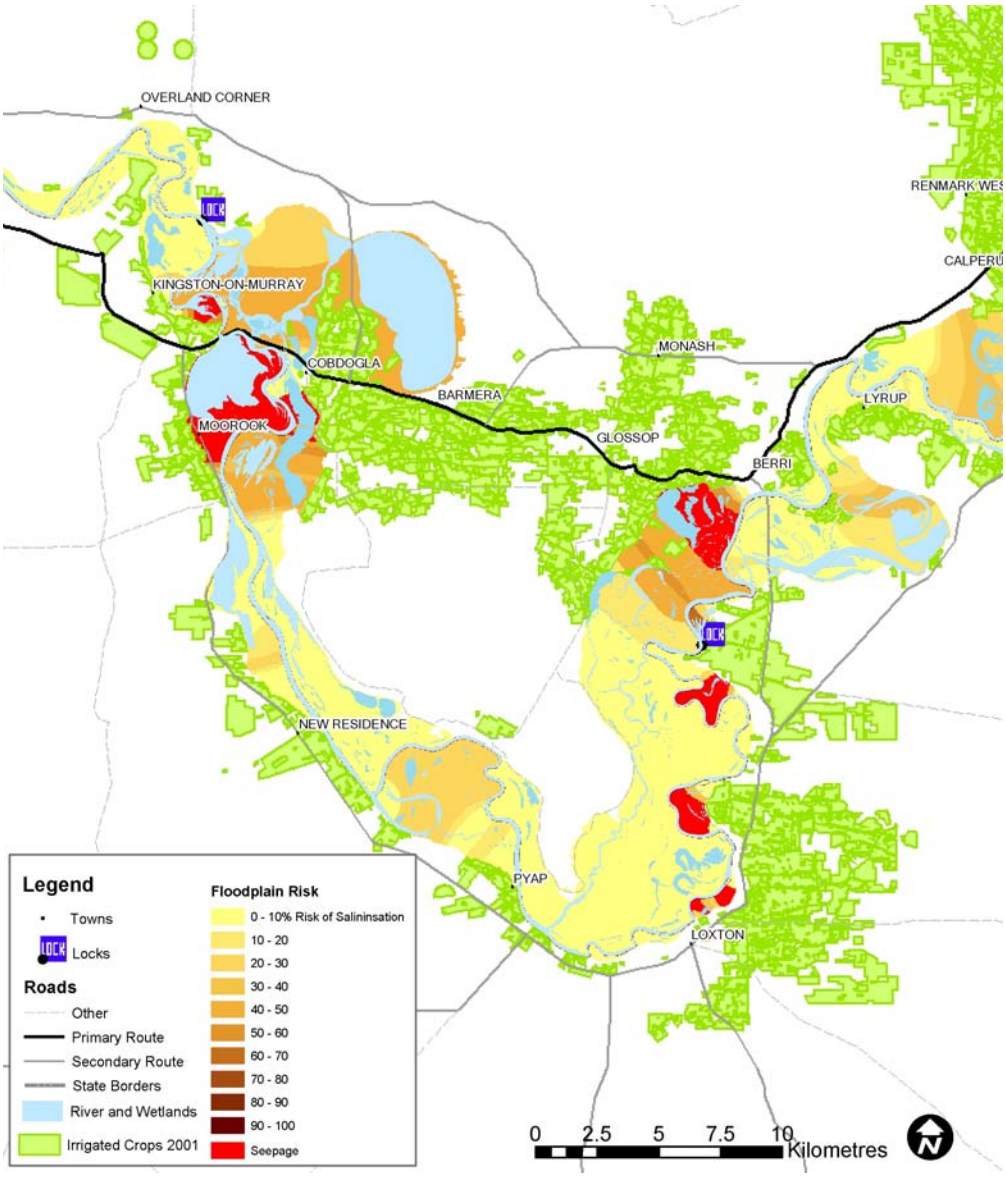

Figure 4.32 Predictions of floodplain risk for the area between Lock 3 and 4 under current inflows but with the groundwater surface lowered by 1 metre. The lowering of the groundwater could be achieved by lowering the weir pool or through groundwater interception schemes. Areas of salinisation have been given a gradational colour based on the percentage of floodplain with groundwater within 2 m of the surface.

4.4.4 Discussion of Groundwater Management Scenarios

This section presented the mapping of groundwater depth via groundwater flux through the floodplain on a regional scale.

A cross-sectional model of groundwater movement across the floodplain was implemented spatially on the whole of the lower River

Murray in South Australia by incorporating an existing analytical cross sectional model of the lower River Murray floodplain. The model has been built to operate within a GIS and allows spatial predictions of floodplain impacts. The FIP model has successfully modelled the distribution of salt and groundwater behaviour through the floodplain of the River Murray.

The model allows predictions of the effects of irrigation on vegetation health for entire sections of the river for the first time and allows the effects of drowning through locking, irrigation and flooding to be separated spatially.

The model can also provide the basis for the development of an irrigation zoning policy by identifying areas of the floodplain at risk from saline seepage, soil salinisation and large salt inflows into the River Murray. It is also useful for providing spatial floodplain attenuation estimates for use in regional and detailed salt load models such as SIMPACT and Land and Water Management Plan MODFLOW models.

River flow and groundwater management scenarios can be combined to predict the overall impact on floodplain health. The Floodplain Risk Methodology (Holland *et al.*, 2009) takes the RiM-FIM model and combines this with results from the FIP model to assess the major drivers of floodplain health and therefore the potential for management scenarios.

The FIP model has been improved by incorporating wetlands within the floodplains (FWIP model) (Holland *et al.*, 2009) and further enhanced by incorporating the impacts of flooding on vegetation health (RiM-FIM) into the Floodplains Risk Methodology (FRM) (Holland *et al.*, 2009). To identify areas at risk of reduced flooding frequency Holland *et al.* (2009) have used the 70,000 ML/day threshold from the 'active floodplain' concept discussed in Chapter 5.

4.5 CONCLUSION

The first objective of this chapter was to develop a method to identify the extent of floodplain inundation from surface water management scenarios at the regional scale, including increased flows and weir manipulation in the lower River Murray. A floodplain inundation model (RiM-FIM) was developed from satellite imagery for the whole of the lower River Murray in South Australia by linking observed inundation at increasing flows to a river height model with in a GIS. The model successfully predicted the area of inundation compared to observed inundation from aerial photography with approximately 15% underestimate of the flooded area. The mean difference between the surface elevation predicted from river height to inundate and surveyed heights was 0.25 metres allowing the RiM-FIM to be used as an elevation model in the absence of detailed survey or LiDAR elevation data. The RiM-FIM model achieves the objectives for this chapter and was used to indicate vegetation health from changes in current versus natural flow regime.

The second objective of this chapter was to develop a predictive model of vegetation risk for the lower River Murray from the impacts of groundwater management scenarios. An existing analytical model identifying seepage, evapotranspiration and river base flow from groundwater inflow to the floodplain was applied spatially to the lower River Murray. The model successfully produced predictive risk to vegetation on the floodplain when compared to known vegetation health, observed seepage areas and salt loads recorded in the river. The model can be used to support decisions on groundwater management scenarios in the lower River Murray.

5 ASSESSING FLOODPLAIN SCALE ENVIRONMENTAL FLOW AND GROUNDWATER MANAGEMENT SCENARIOS

5.1 INTRODUCTION

The objectives of this chapter are firstly to develop a predictive model of the extent of floodplain inundation from surface water management scenarios, at the floodplain scale for the Chowilla floodplain. The second objective is to produce a predictive model of groundwater changes from management scenarios for the Chowilla floodplain. This will involve identifying potential groundwater recharge, soil salinisation rates, benefits of flooding on soil water availability and the disadvantage of increased salt loads to the river.

In order to determine the environmental benefits of flooding on the soil salinity and groundwater recharge and to model the salinity impacts of increasing the area of inundation, it is important to examine the potential groundwater recharge that could occur.

This Chapter maps the potential groundwater recharge on the Chowilla floodplain, using a number of different methods including soil types, vegetation response to flooding and electromagnetic data. The methodology has implications for mapping this critical environmental variable across the whole of the lower River Murray.

Along with the benefits that flooding produces in soil leaching and groundwater recharge it also influences the volume of salt that is discharged into the River Murray following a flood event. This Chapter also considers this dis-benefit and determines a relationship between flood size and salt load that is consistent with present understanding of salt load processes.

The chapter is structured in the following way:

- Firstly the chapter presents models of environmental factors affecting floodplain tree health (5.2) including:

  - the development of a vegetation health map that can be used at the floodplain scale for validation (5.2.1);
  - flooding frequency and duration mapping (5.2.2);
  - soil properties and groundwater recharge (5.2.3);
  - groundwater depth and salinity (5.2.4); and
  - other freshwater sources (5.2.5).
- Secondly the chapter discusses the development of the tree health model (5.3). This is discussed progressively from a habitat model of tree health (5.3.2) to a process model of salinisation (5.3.3) and then a process model of tree health (5.3.4); and
- Thirdly the chapter uses the model produced to test a number of management options and to demonstrate its applicability at the floodplain scale.

5.2 MODELLING ENVIRONMENTAL FACTORS AFFECTING FLOODPLAIN TREE HEALTH

5.2.1 Mapping Tree Health at the Floodplain Scale

Some assessment and mapping of tree health was required to compare and validate model outputs. It was necessary to be able to map tree heath over the last 20 years as flooding history impacts on the current health of the trees. To examine the more detailed management scenarios on the Chowilla floodplain it was necessary to map the vegetation and its health at a fine scale and monitor changes over time to validate the predictive model. An automated process was sought to map vegetation community and health over the floodplain to allow a rapid and repeatable assessment that was independent of operator to allow future monitoring and comparison of vegetation condition. As discussed previously, a number of shortcomings in the quality of the mapping results, and the release of new mapping by the South Australian Government, led to the use of the 2003 Department for Environment data (Smith and Kenny, 2005).

To use the Smith and Kenny (2005) 2003 vegetation map for the Chowilla floodplain it was necessary to undertake a number of refinements to the map including:

- Addition of the NSW vegetation map for the Eastern portion of the Chowilla floodplain;
- Simplification of the 63 vegetation classes to 12 so that the major vegetation types can be easily identified and the SA and NSW vegetation classifications could be integrated;
- Edge matching at the border;
- Addition of vegetation health attributes to the NSW data from previous health assessment; and
- Dividing the river red gum/black box mix vegetation class of DEH into two distinct classes for river red gum and black box. This was

achieved by overlaying the vegetation classification onto aerial photographs and dividing the polygons along the visible boundary between river red gums and black box. Not all of the mixed polygons could be divided in this way as the two species do co-habit. Along the edges of creeks, however, there is usually a single line of river red gums in front of the black box woodland.

The resulting vegetation map is presented in Figure 5.1 and the resulting tree health map is presented in Figure 5.2.

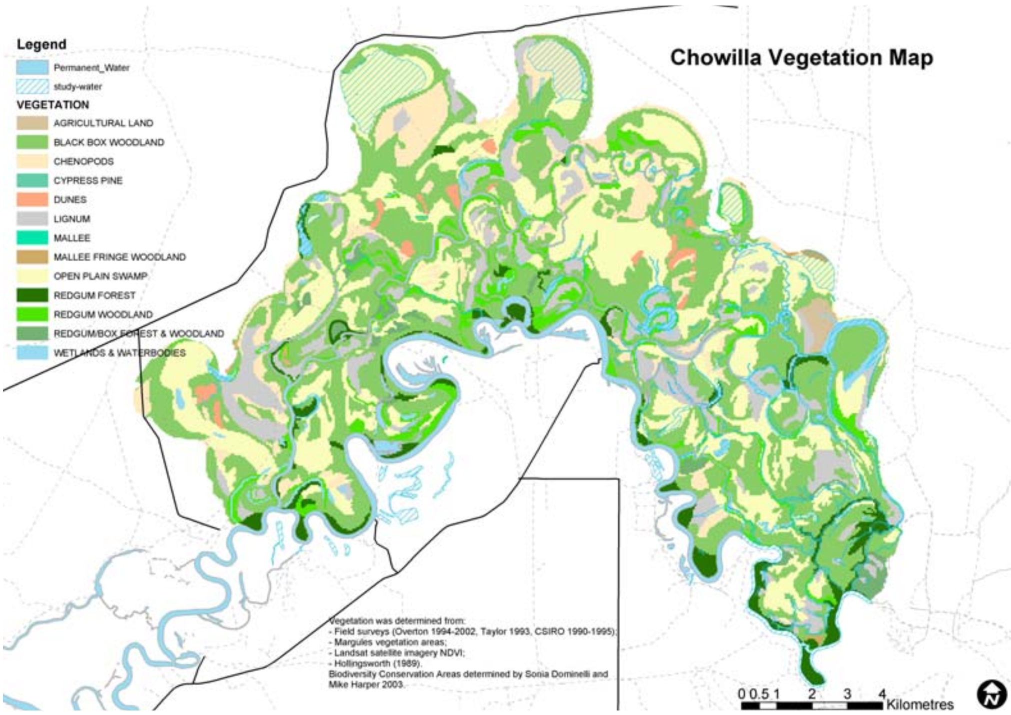

Figure 5.1 Map of vegetation types on the Chowilla floodplain.

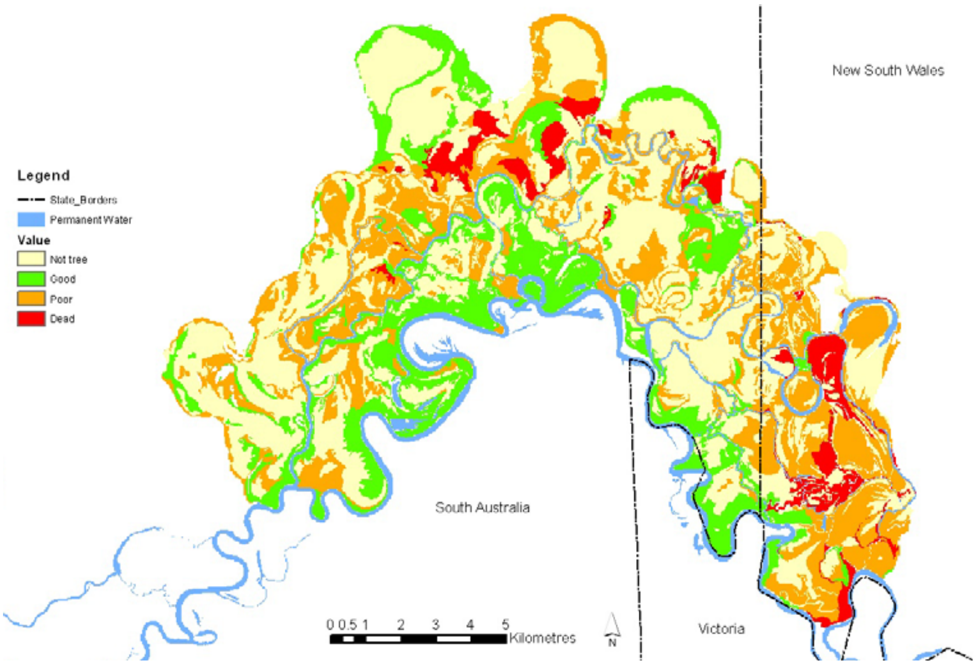

Figure 5.2 Map of vegetation health of the Chowilla floodplain (2003) from aerial photography and field assessment. This figure can be compared with 3.1 from 1994.

5.2.2 Flooding Frequency and Duration

RiM-FIM Model in Chowilla

The RiM-FIM model (section 4.3) was applied to the Chowilla floodplain by using the flow into South Australia and the height of Lock 6. One of its major applications has been the identification of threshold flows that are required to inundate the different vegetation types on the floodplain. The RiM-FIM model was also used to predict changes to the flooding frequency following environmental flow strategies such as the 350, 500, 750 and 1,500 GL scenarios proposed by the Living Murray program (MDBC, 2005). The model was also useful for mapping the extent of overbank flooding from Lock 6 management scenarios. Figures 5.3 to 5.7 show the increasing inundation of the Chowilla floodplain from increasing flows.

The RiM-FIM is based on satellite images of previous floods and is unable to predict changes in area of inundation from changes in flow paths caused by new flow control structures or the removal of existing ones.

For example, a new weir has been proposed for Chowilla Creek to block water as it by-passes Lock 6. A more dynamic model of surface flow is required to model these proposed changes.

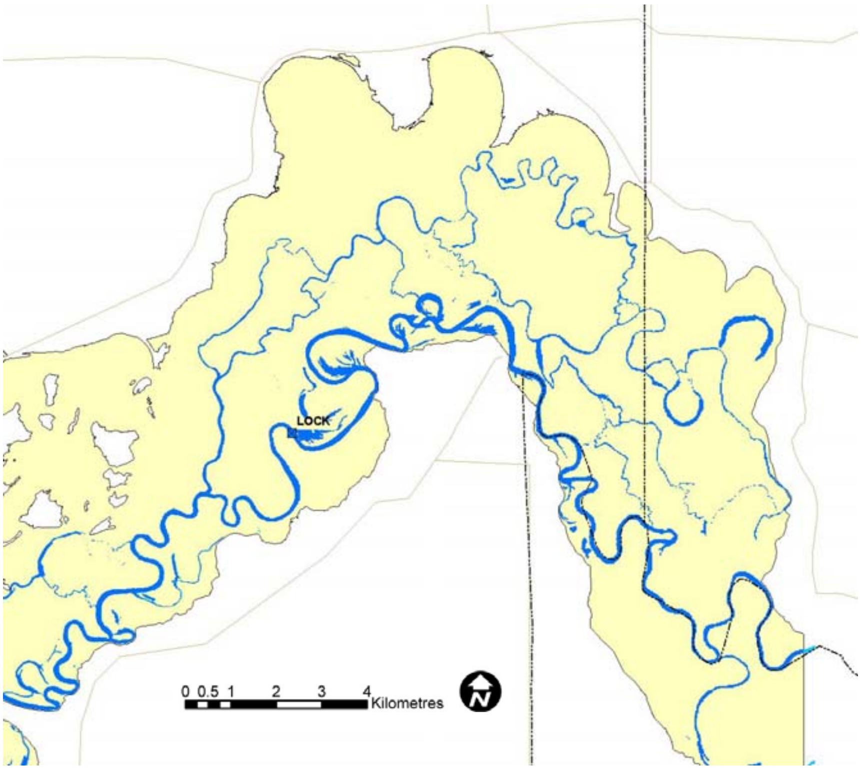

Figure 5.3 Floodplain inundation in Chowilla with 5,000 ML/day, Lock 6 at pool level and no weir in Chowilla creek.

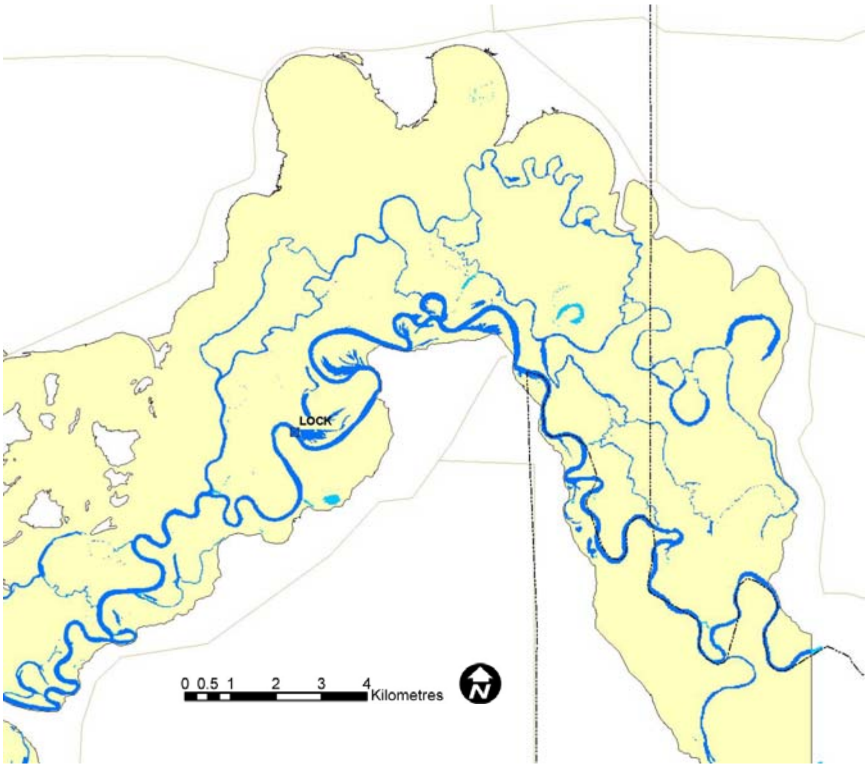

Figure 5.4 Floodplain inundation in Chowilla with 20,000 ML/day, Lock 6 at pool level and no weir in Chowilla creek.

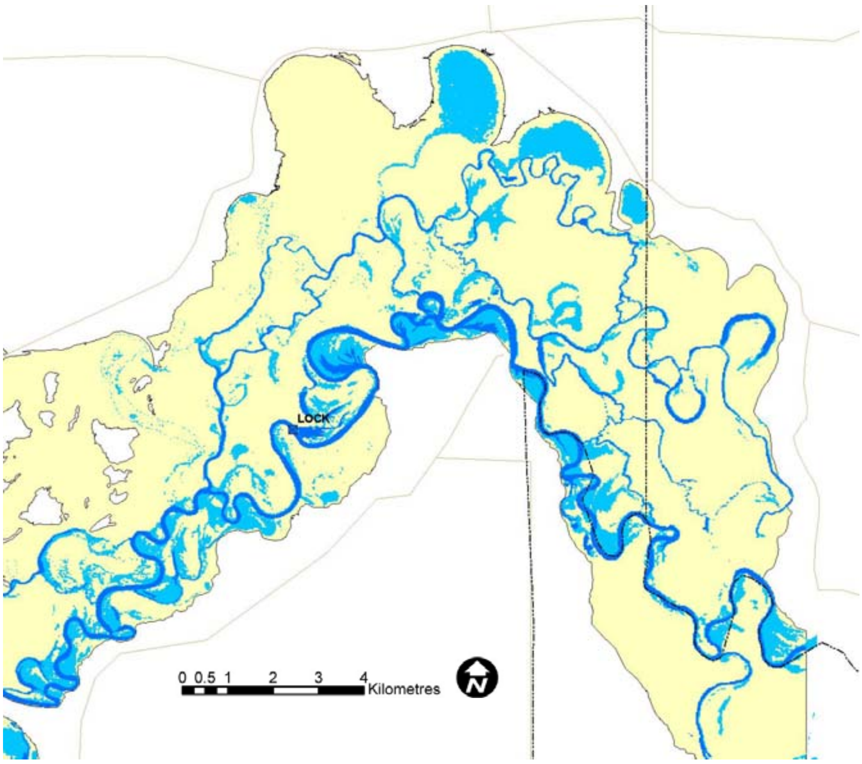

Figure 5.5 Floodplain inundation in Chowilla with 60,000 ML/day, Lock 6 at pool level and no weir in Chowilla creek.

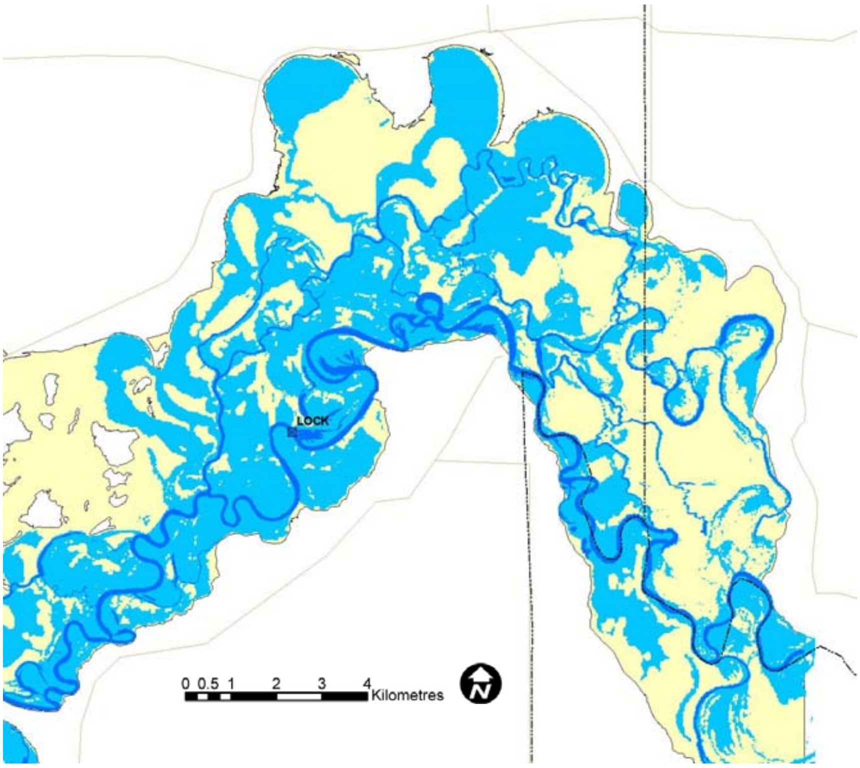

Figure 5.6 Floodplain inundation in Chowilla with 80,000 ML/day, Lock 6 at pool level and no weir in Chowilla creek.

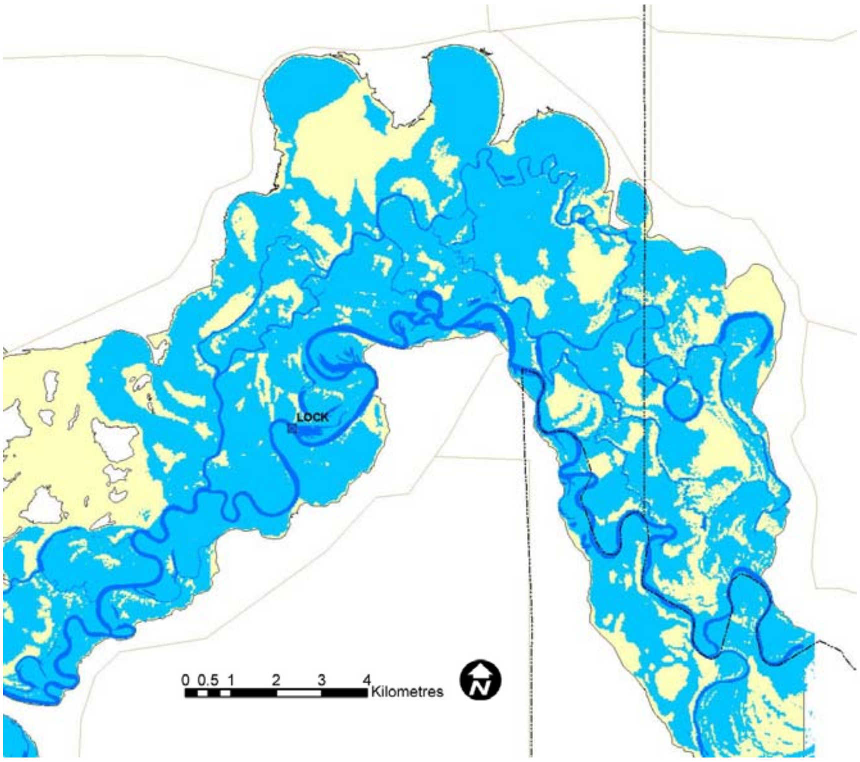

Figure 5.7 Floodplain inundation in Chowilla with 100,000 ML/day, Lock 6 at pool level and no weir in Chowilla creek.

Pseudo-Hydrodynamic Model

A simple River-Height-Flow model for the Chowilla floodplain was developed by Overton *et al.* (2005) to predict changes in the anabranch creek heights on the Chowilla floodplain from changes in flow and the installation of control structures. A full description of the River-Height-Flow model can be found in Overton *et al.* (2005). This River-Height-Flow model was used as the basis for a pseudo-hydrodynamic model of the floodplain. This model used a number of locations along the main floodplain creeks (Figure 5.8) and determined the cross-sectional profile.

Using the main river channel heights from the RiM-FIM (Section 4.3) the flow into each creek was calculated using Manning's Equation (Chanson, 2004) for surface water flow. The model then calculates backwater curves for each creek based on the height of any control structures. The model can also predict river velocities at the same locations. For the required flow and new control structure height, the creek heights predicted from the River-Height-Flow for each location were digitised into a GIS and a surface water elevation was generated using these heights and the height of the main channel using RiM-FIM backwater curves. The predicted water surface was intersected with the surface elevation layer for the floodplain, to generate a potential flood extent layer. A surface elevation model for Chowilla derived from LiDAR airborne laser data (DWLBC, 2006) was edited to remove areas that were predicted to be inundated but were not hydraulically connected to the creeks. These areas included low lying wetlands where the commence-to-fill threshold had not actually been reached because of flow blockages not seen on the elevation layer.

The layer was also edited to define the banks that would be necessary either side of the new weir to block the flood waters going around the structure. This pseudo-hydrodynamic model can be used for a range of flows and any number of control structure changes in the floodplain creeks.

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Figure 5.8 The locations used in the River-Height-Flow model for the Chowilla floodplain (Overton *et al.,* 2005).

The model was used to predict the extent of inundation caused by different weir configurations. Figure 5.9 shows the extent and depth of flood extent with a weir of 19.5 metres AHD in Chowilla Creek using a simple flat elevation of water compared to the surface elevation data.

The extent of inundation in Figure 5.9 can be compared with that shown in Figure 5.10. The area of inundation in Figure 5.10 is derived from the pseudo-hydrodynamic model and takes into consideration the effect of the backwater curve in the anabranch creeks. The extent of flooding in Figure 5.10 could only be achieved if no further flow was allowed to enter the Chowilla anabranch system and the water present pooled behind the weir. This situation is highly unlikely and using the simple elevation technique for flood extent is therefore not viable.

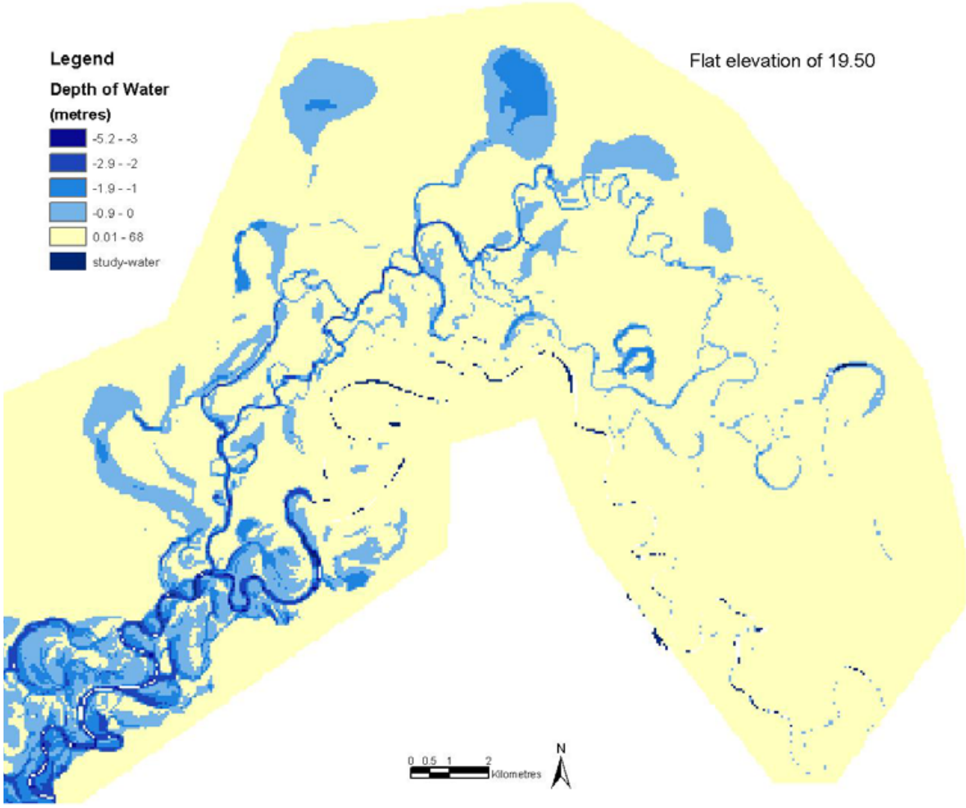

Figure 5.9 Inundation of flooding predicted using simple 19.50 m AHD elevation and DTM.

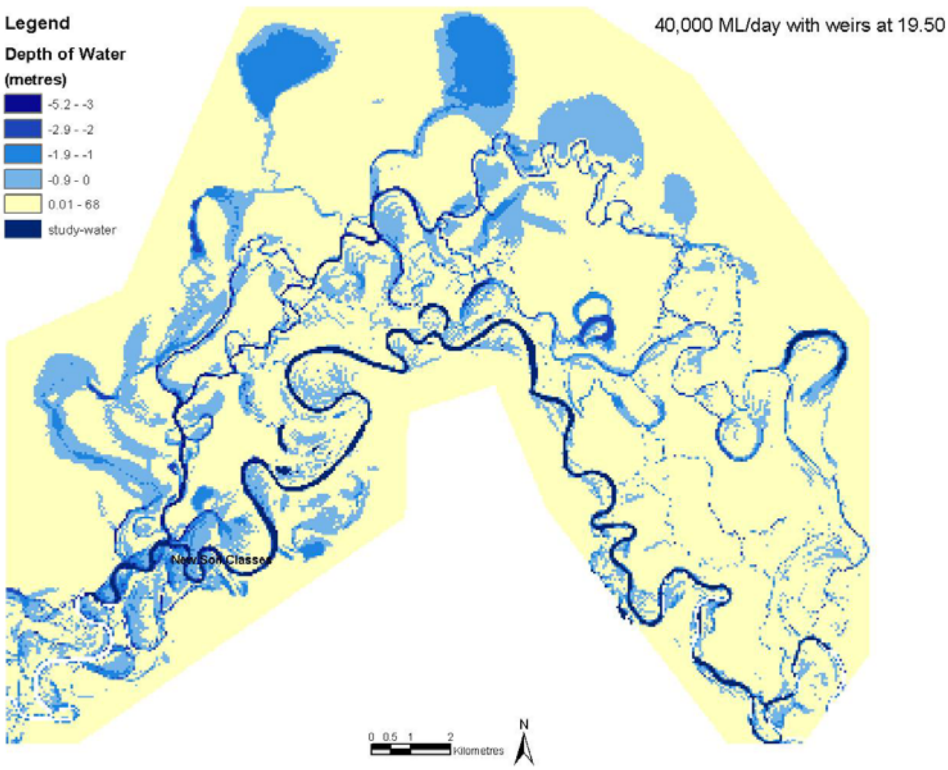

Figure 5.10 Inundation of flooding at 40,000 ML/day with both Lock 6 and Chowilla Creek weir at 19.50 m AHD.

Weirs can be held above normal pool operating to increase the height of the river upstream. Pool level for Lock 6, which occurs in the Chowilla floodplain, holds water above downstream levels by approximately 3 metres. Figure 5.11 shows the maximum flood extent from a weir of 19.87 metres at the bottom of Chowilla Creek under different flows in the River Murray.

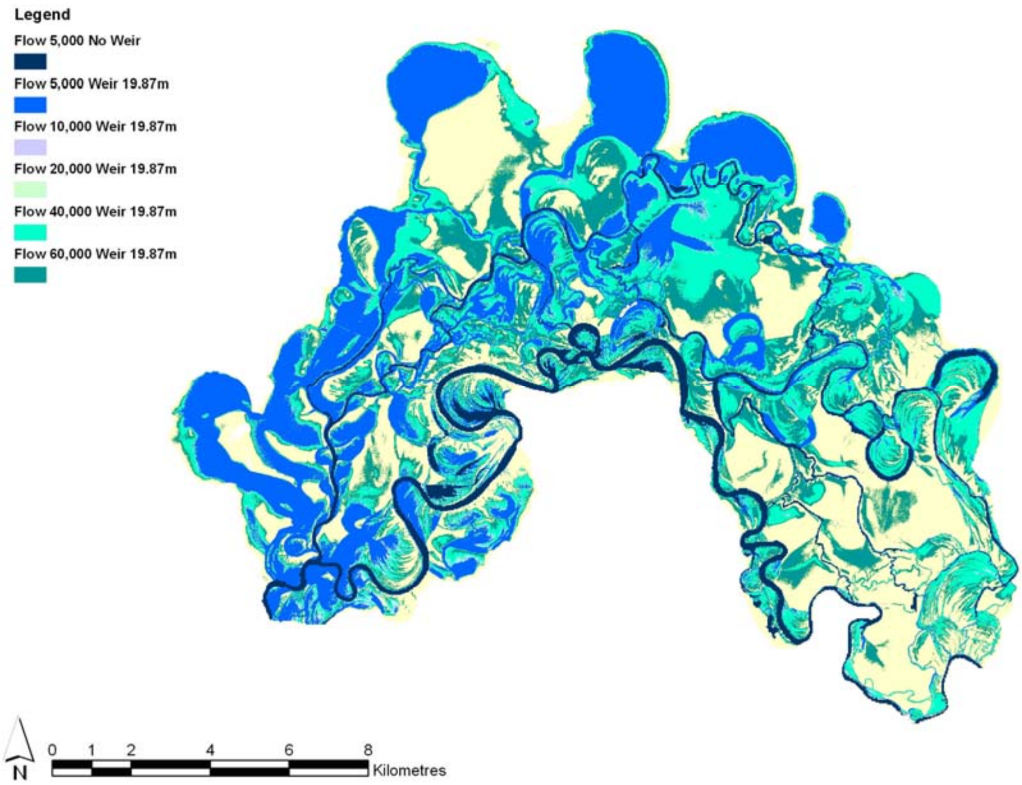

Figure 5.11 Extent of the maximum inundation on the Chowilla floodplain from a weir of 19.87 metres at the bottom of Chowilla Creek under different flows in the River Murray.

5.2.3 Groundwater Recharge and water sources

Soil properties impact on the rate of groundwater recharge into the unsaturated soil profile. The main property of concern is the amount of sand versus clay in the soil. A very sandy soil will allow water to pass easily through it and will have higher rates of recharge and therefore support healthier vegetation. A very heavy clay soil will have little or no recharge during rainfall or flooding and therefore likely to remain drier and less likely to support healthy vegetation in a region of similar flooding frequencies.

Identifying Recharge Areas from Soil Types

The soils on the Chowilla floodplain have been summarised into nine classes (Overton and Jolly, 2003) (Figure 5.12). These were based on a combination of the landform patterns and elements of Hollingsworth (1990), data on surface elevation, Landsat satellite Normalised Vegetation Difference Index (NDVI) and field soil profiles collected by CSIRO during the 1990s (McEwan *et al.*, 1994). Selected properties of these nine soil types are provided in Table 5.1.

This soil map does not rigidly adhere to a standard soil classification system or true pedological standards, but is useful in dividing the floodplain into soils that behave differently in terms of salt leaching and discharge. The values for properties presented are averages for the soil classes. Soils vary substantially across the floodplain, often with great variation within metres. The concept of an average soil type is therefore problematic and the characteristics of a soil pit at a particular location will most certainly be different to the average value for that class. However, it is both necessary and useful to group soils into a few classes for simplicity. Observed variation of tree health, for example one live tree with dead trees all around, may be accounted for by local soil variation. The soil hydraulic properties were assigned to each soil class for WINDS modelling.

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Figure 5.12 Soils of the Chowilla floodplain showing the nine different soil types (Overton and Jolly, 2003 based on Hollingsworth, 1990). Soil types are described in Table 5.1.

Table 5.1 Soil types and the soil hydraulic properties used in the modelling (Overton and Jolly, 2003).

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The soil types were then assessed for their potential recharge capability based on expert opinion (Overton and Jolly, 2003). Four of the soil types (3, 4a, 4b and 5) are likely to allow some recharge when inundated. Three of the soil types (2a, 2b and 2c) may allow some recharge to occur when inundated. The remaining two soil types (1a and 1b) are likely to allow minimal, if any, recharge when inundated. Recharge rates ranged from 1 mm/day for the heavy clay areas to 500 mm/day for the sand dune areas (Table 5.2). The recharge rates should be considered as maximum rates that the soil physical structure would allow. Actual floodplain rates are likely to be lower than these due to factors such as vegetation water use. When these recharge rates were used in a MODFLOW model of groundwater they were found to be too high for model calibration (Yan *et al.*, 2005). The actual soil recharge rates were then reduced on the floodplain soils to produce rates that were more likely to be actual than potential (Table 5.2). The recharge rates for the different soils are displayed spatially for the Chowilla floodplain in Figure 5.13.

Table 5.2 Recharge rates based on Ksat of soil types.

Soil Type	Comment	Maximum Recharge rate (mm/day)	Maximum Recharge rate (mm/year)	Recharge Rates (mm/day)
1a	Heavy Clay (Lignum)	1	300	0.5
1b	Clay (River red gum/black box)	1	300	0.5
2a	Sandy Clay (Open Plain Swamps)	2	700	1
2b	Sandy Clay (River red gum Along Creeks)	2	700	1
2c	Sandy Clay (River red gum/black box Forest)	2	700	1
3	Clay Loam (Black box Woodland)	6	2,000	2
4a	Sandy Clay Loam Salinised (Black box Woodland)	6	2,000	2
4b	Sandy Clay Loam (Black box Woodland)	6	2,000	2
5	Sandy Loam (Dunes)	500	200,000	500

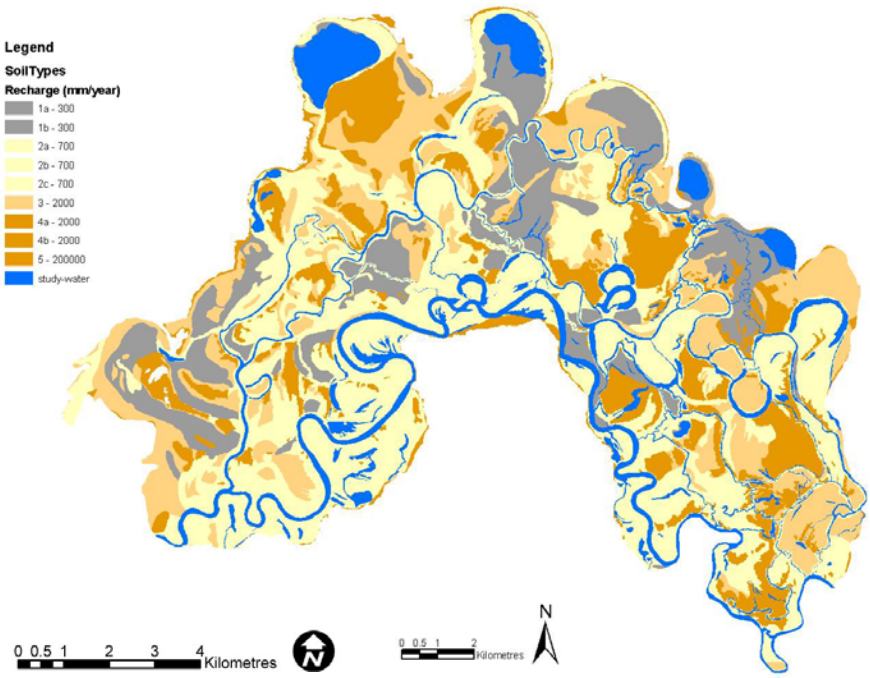

Figure 5.13 Maximum recharge rates on the Chowilla Floodplain based on soil types.

Identifying Recharge Areas from Vegetation Response to Flooding

Another indicator of recharge areas on the floodplain was the presence of vigorous vegetation on high elevations of the floodplain. Presence of healthy vegetation at high elevations suggests a fresh groundwater reserve which is most likely to have come from recharge as a result of surface flooding and rainfall. The 'Garden of Eden' near Hancock Creek is an example of such an area as it previously supported vigorous river red gums in locally elevated areas. Lack of flooding in the last 10-15 years has led to the depletion of the fresh groundwater reserve and this area is in poor condition today.

High recharge could therefore be inferred from areas that show a strong response to flooding in a sustained manner months after a flood event. It is anticipated that freshwater lenses under these recharge areas would be replenished by flooding and could then sustain good foliage growth for the overlying trees.

Landsat satellite imagery was used to map vegetation vigour of the floodplain before and after a flood event using the Normalised

Vegetation Index (NDVI). Two images were obtained for 1995 and 1997 after a flood in late 1996 (Table 5.3). Figure 5.14 illustrates the satellite imagery before and after the 1996 flood event.

Table 5.3 Satellite images used in the vegetation response to flooding study.

Date	Satellite	Row/Path	Flow at the border (ML/day)	Time since last flood
21 Jan 1995	Landsat 5 TM	96/84	6,309	12 months
05 Jul 1997	Landsat 5 TM	96/84	5,564	6 months

By comparing these images it is possible to identify areas which have responded with more vegetation growth than other areas. Areas flooded and not flooded show a distinct difference. Within the areas flooded, the best response, sustained after 6 months, was considered to indicate a local recharge area.

Areas that showed the strongest response to flooding were then overlaid onto the recharge map derived from soil types. Four recharge rates were simplified into two recharge rates and compared to the areas of high vegetation response. The areas classified as low recharge (1 or 2 mm/day) where changed to high recharge (6 mm/day) if the vegetation had responded to the recent flood. Figure 5.15 shows these areas of high vegetation response in relation to two recharge rate classes. A recharge map was developed using the above methods (Figure 5.16).

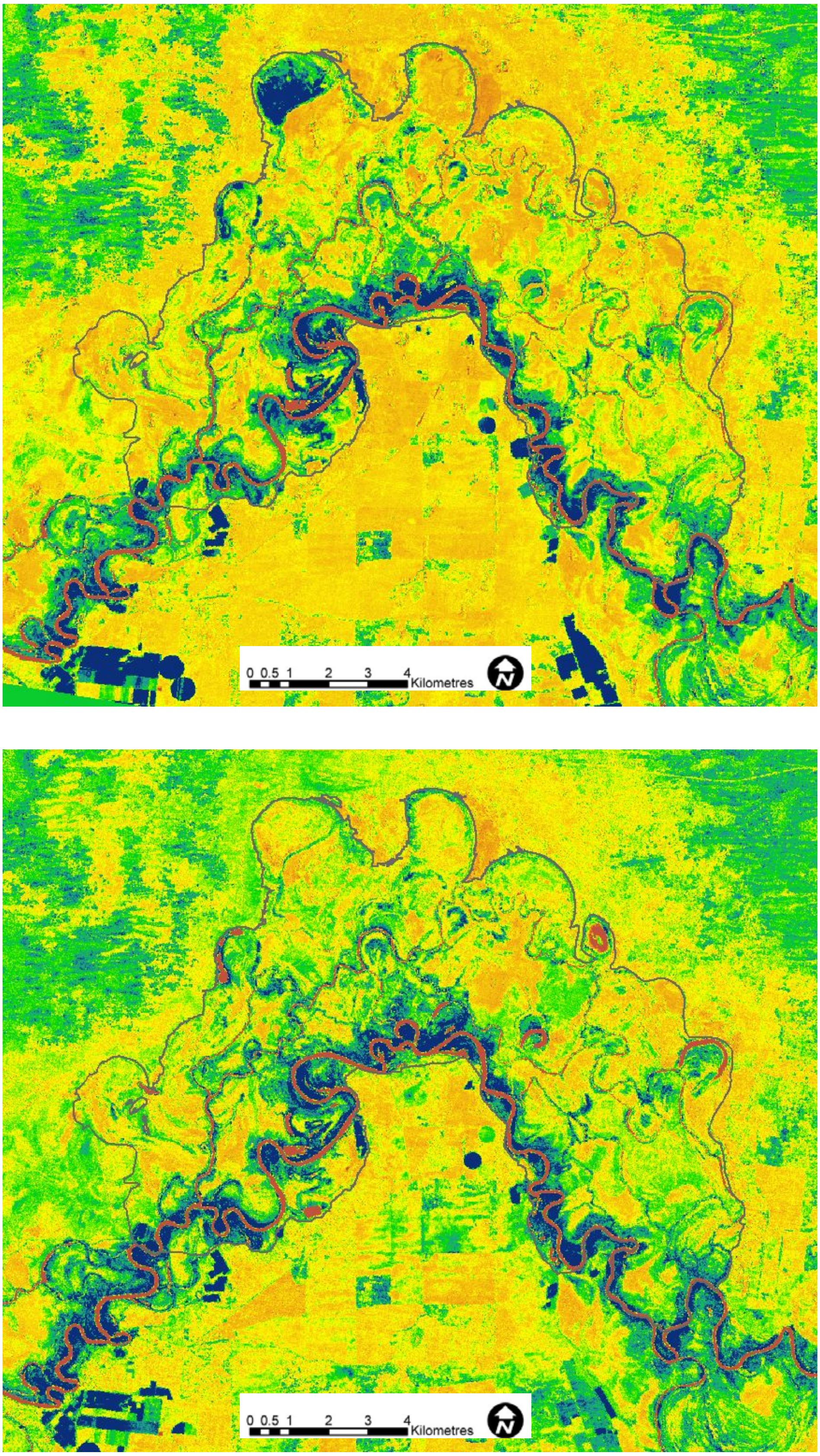

Figure 5.14 Satellite NDVI images for January 1995 (12 months following a flood and prior to the 1996 flood) and July 1997 (6 months following the 1996 flood). Blue and green areas show high vegetation vigour and orange areas show little vegetation vigour.

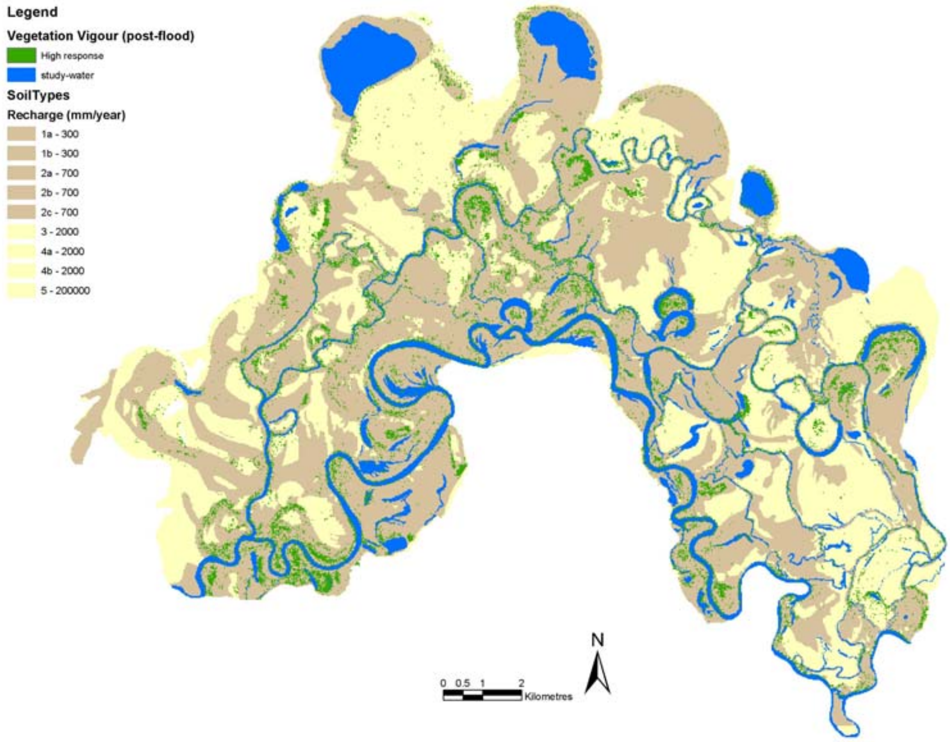

Figure 5.15 NDVI detected areas of high vegetation vigour response to flood (1996). The green areas are drawn over the recharge map based on soil types only.

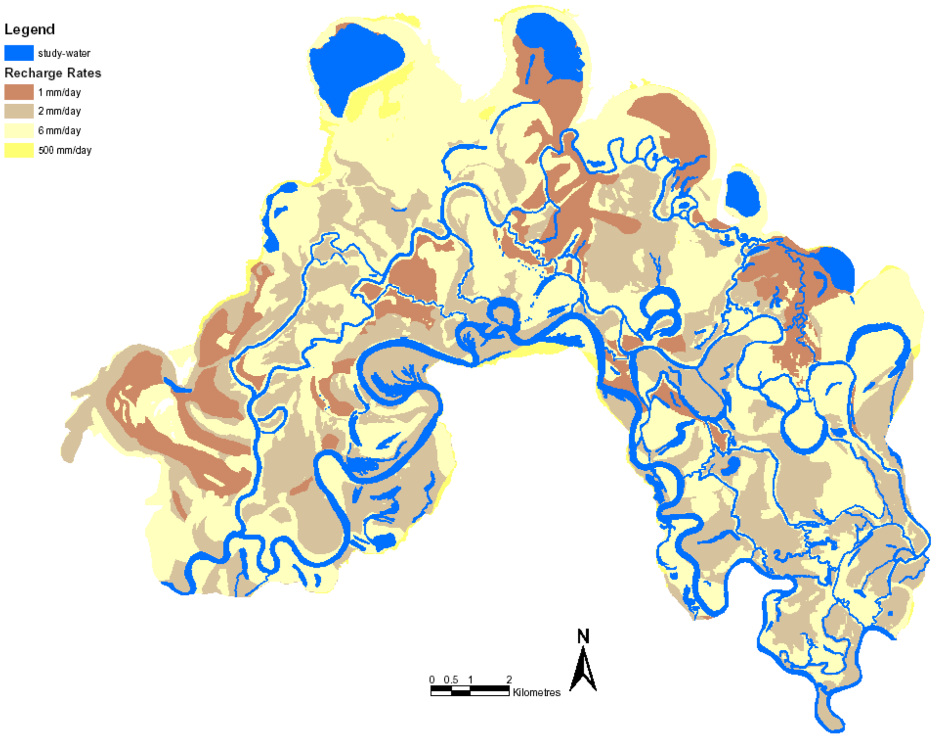

Figure 5.16 Recharge map of Chowilla showing the four recharge areas of 1, 2, 6 and 500 mm/day.

Identifying Recharge Areas from Electromagnetics

Airborne electromagnetic radiation data (AEM) was used to identify local recharge areas and lateral recharge for the WINDS model. The geophysical properties that AEM can detect are a combination of clay content, moisture content and salinity. High clay suggests non-recharge areas which may coincide with high salinity areas and high moisture content, as clay retains more water than sandy soils. The three properties therefore are complementary and high responses from the EM imagery indicate recharge areas.

Doble *et al.* (2006) showed that airborne electromagnetics could be used to identify groundwater recharge areas on the Bookpurnong floodplain. The electromagnetics are also useful in identifying lateral recharge from creeks. In regions where the creek water level is higher than the surrounding groundwater level the gradient for water flow is from the creek to the groundwater which produces a losing creek (Figure 5.17). Losing creeks are characterised by healthy fringing vegetation as the fresh creek water replenishes the bank storage to support the vegetation. In areas where the groundwater is higher than the creek the reverse occurs where saline groundwater enters the creek (Figure 5.18). Gaining creeks have poor or dead vegetation and can have salt crystallisation on their banks.

The airborne EM data captured in 2007 (Munday *et al.*, 2008) measured the conductivity of the soil at approximately 6-8 metres below the surface (Figure 5.19). Lateral recharge from creeks will occur in losing creek situations and not in gaining creek conditions and these two types of creeks can be identified in the image with low (blue) and high (red) conductivities respectively which correspond to known losing creeks (lower south-west creeks and the 'flush-zone' around Lock 6) and gaining creeks in the upper north-east. Figure 5.20 shows the areas of bank recharge identified from the AEM using a density slice from visual inspection.

Figure 5.17 Losing creek region in Chowilla Creek.

Figure 5.18 Gaining creek region in Salt Creek.

Another interpretation of the data provided conductivity at shallower depths of approximately 2-4 below the surface (Munday *et al.*, 2008) (Figure 5.21). The density slice image (Figure 5.22) shows the areas of local recharge which correspond well with known sandy areas and the 'Garden of Eden'.

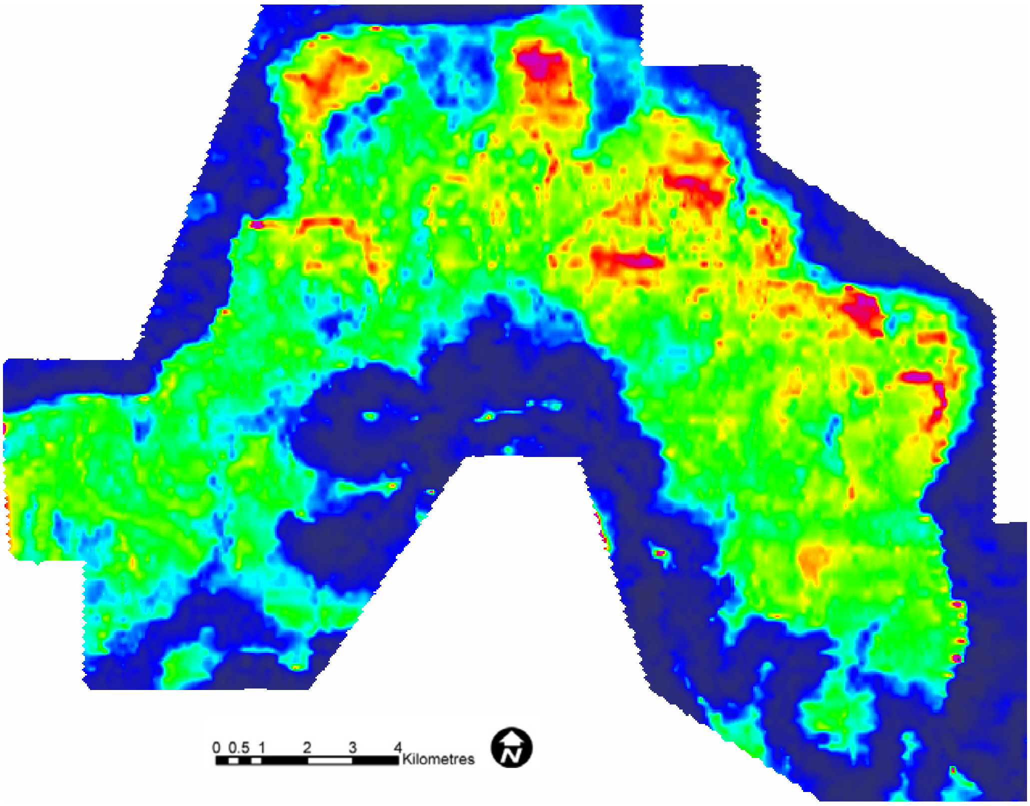

Figure 5.19 Airborne electromagnetic imagery over Chowilla showing the conductivity at approximately 6-8 metres below the surface. Blue areas represent low conductivity (min 0 mS/m) and red areas are high (max 2000 mS/m).

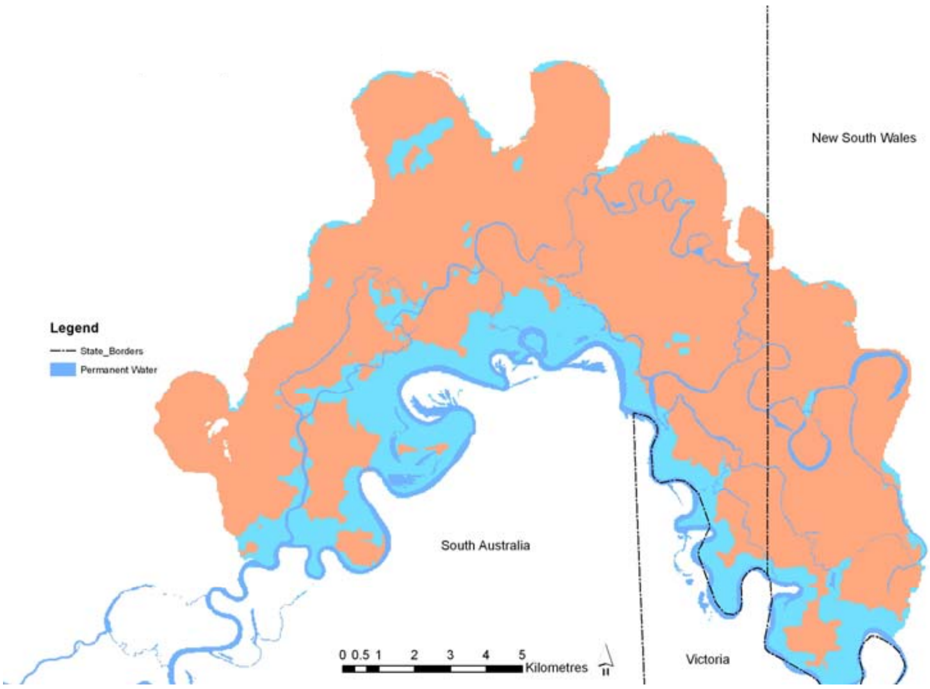

Figure 5.20 Bank recharge areas from losing creeks identified by the Airborne EM data.

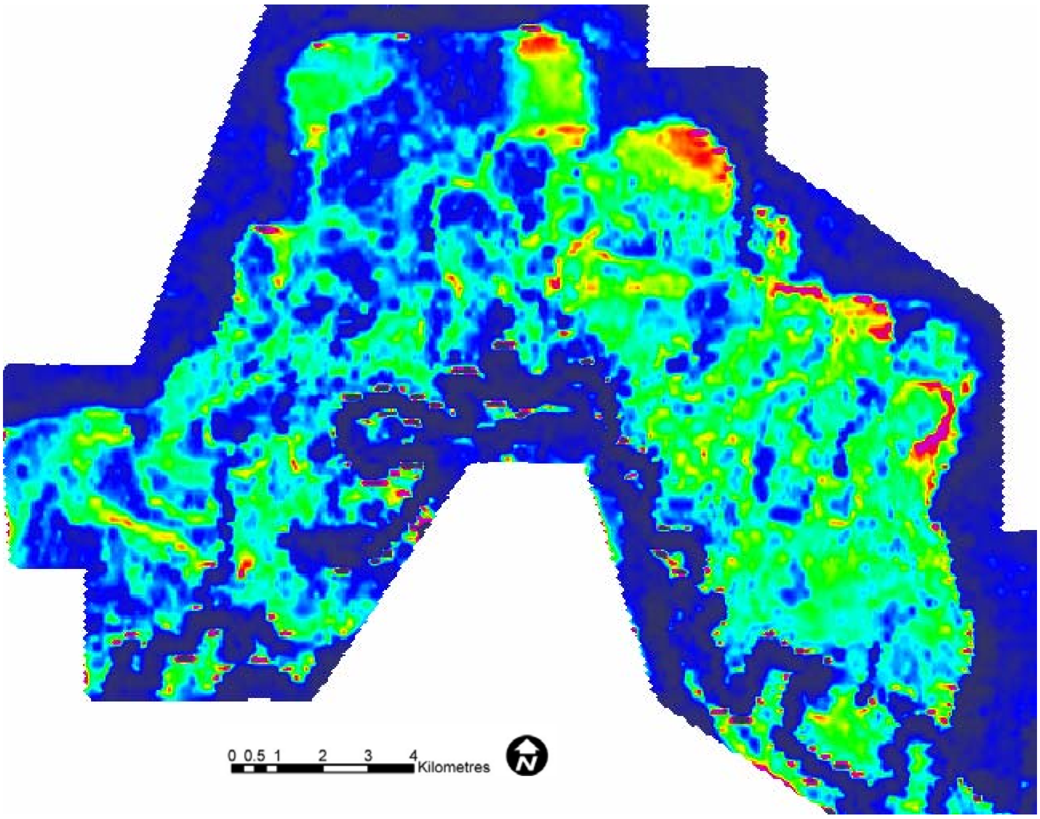

Figure 5.21 Airborne electromagnetic imagery over Chowilla showing the conductivity at approximately 2-4 metres below the surface. Blue areas represent low conductivity (min 0 mS/m) and red areas are high (max 2000 mS/m).

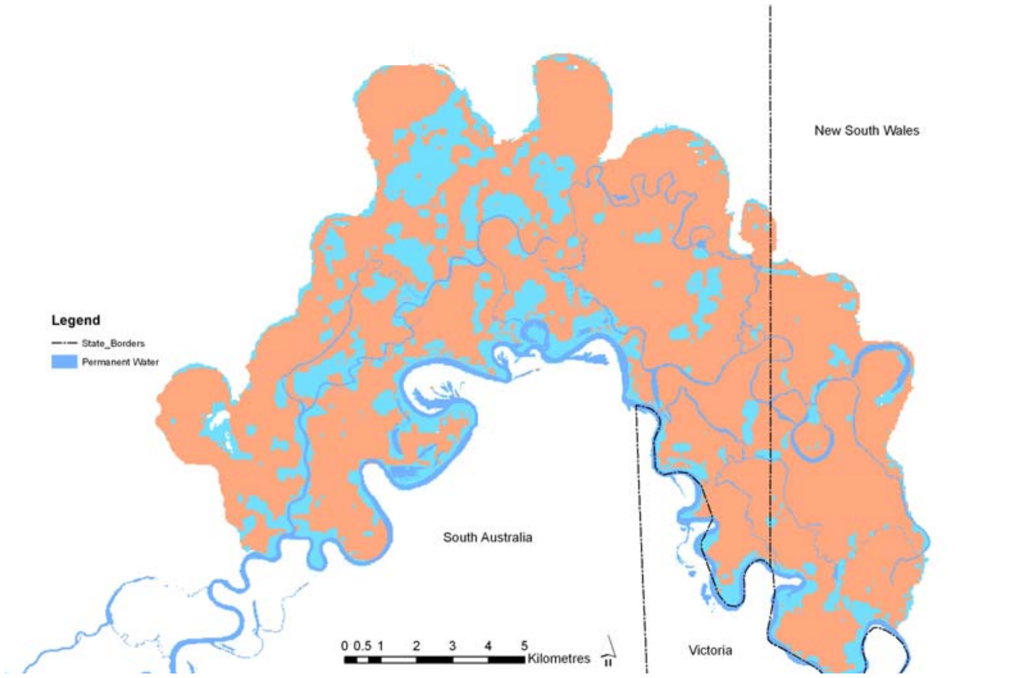

Figure 5.22 Local recharge areas identified by the Airborne EM data. The areas correspond to sandy areas.

These bank recharge and local recharge areas were used to develop a potential recharge map (Figure 5.23), which was used to refine the groundwater salinity map as explained in the following section.

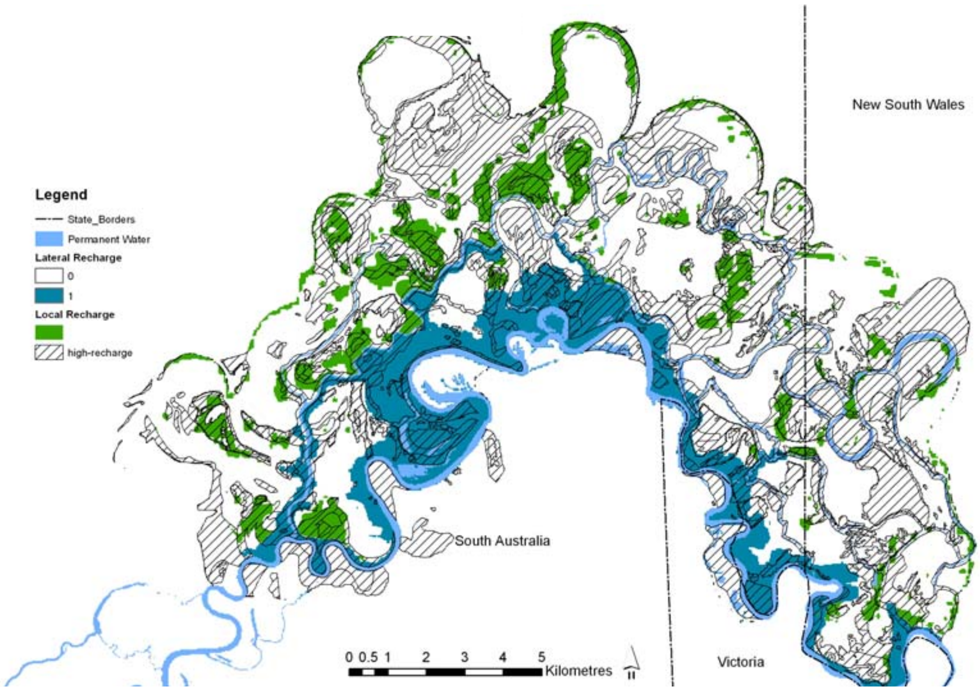

Figure 5.23 Potential groundwater dependent areas (green) and local recharge areas (hatched) identified from the Airborne EM data.

Recharge mapping is critical to identify the different water sources that can support tree health above flooding frequencies and for those areas where the response relationship for flooding frequency needs to be altered. The three methods described above, soil parameters, vegetation vigour and electromagnetic radiation, provided detection of different and complimentary aspects of the environment influenced by recharge. Soil parameters provide a means for altering the degree of leaching from flood events, vegetation vigour identified freshwater lenses and electromagnetics detecting lateral recharge from loosing streams.

5.2.4 Groundwater Depth and Salinity

The FIP model described in Chapter 4.4 is for regional scale management assessment but is inadequate when considering local scale floodplain management scenarios such as floodplain groundwater pumping. The floodplain tree health modelling in this chapter provides higher spatial resolution modelling of vegetation risk. In order to undertake the vegetation health modelling, groundwater depth and salinity maps are required at a scale similar to the modelling.

A MODFLOW model of the Chowilla floodplain (Yan *et al.*, 2005) was used in the floodplain tree health modelling. The MODFLOW model was used to determine current groundwater depth (Figure 5.24) and predicted groundwater depths as a result of management scenarios discussed in the next section.

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Figure 5.24 Groundwater depth (in metres) surface produced from MODFLOW model (Yan *et al.*, 2005)

A groundwater salinity map was also required for the vegetation modelling work. Previous information available on groundwater salinity included a map of the proposed flush zone (Collingham, 1990a; 1990b) which included two areas, within and away from the flush zone. The airborne electromagnetics data was used to derive a groundwater salinity map for a depth of five metres below the surface (Figure 5.25). The extent of the flush zone can be seen in dark blue (low conductivity indicating low salinity) (Munday *et al.*, 2008).

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Figure 5.25 AEM derived groundwater salinity, 5 m below standing water level (Munday *et al.*, 2008).

5.2.5 Discussion of Modelling Environmental Factors

This section has developed methods for determining flooding frequency and duration, groundwater recharge, groundwater depth and salinity, soil salinisation and the presence of freshwater sources on the Chowilla floodplain. The detection of vegetation response from flooding was shown to be useful for mapping recharge potential and could be used where detailed soil mapping is not available. Airborne electromagnetic data analysis was useful for mapping recharge potential, as well as distinguishing local bank recharge from deeper freshwater lens recharge. The electromagnetics was also used to derive a groundwater salnity map by taking a horizontal slice at a depth below the water table.

5.3 MODELLING FLOODPLAIN TREE HEALTH

5.3.1 Introduction

The objective of this section is to develop a predictive model of tree health for the Chowilla floodplain using the interaction of surface water and groundwater conditions.

This section builds the history of tree health modelling on the Chowilla floodplain and outlines the development of a process based tree health prediction model. The stages presented include:

- An initial model of black box health which was developed as a **Habitat Suitability Model** based on black box thresholds for tolerance of flooding, groundwater depth and salinity. This was developed first by Noyce and Nicolson (1993) and later enhanced by Hodgson (1993) (Section 5.3.2);

- Later Slavich *et al.* (1999a) used a complex plant growth model called **WAVES** to improve the predictive capacity to changes in flooding and groundwater (Section 5.3.3);

- Slavich *et al.* (1999b) simplified the WAVES model concept by developing a **Moving Salt Front (MSF) Model** by combining limiting vegetation characteristics, soil hydraulic properties and flood history to develop a salinity index which is indicative of vegetation health (Section 5.3.4). This reduced the number of parameters and allowed a more rapid assessment of salinisation at a point;

- This thesis then uses the equations developed by Slavich *et al.* (1999b) to develop an initial spatial model called the **Salinisation Risk Model** (Section 5.3.5). This model improves on the habitat suitability model as it is based on salinisation processes and is able to predict incremental changes in soil salinity;

- The final stage in the development of the research was to fully incorporate the moving salt front equations into a GIS to produce the **WINDS** model (Section 5.3.6). This model predicts soil salinity and then infers tree health based on tolerances of red gum and black box.

Components of the methodology, results and discussion of this section have been presented in the following publications:

Overton, I.C. and Jolly, I.D. (2004). 'Groundwater Lowering and Environmental Flow Scenarios for Chowilla'. Proceedings of the 9th Murray-Darling Basin Groundwater Conference, February 2004, Bendigo.

Overton, I.C. and Jolly, I.D. (2004). 'Integrated Studies of Floodplain Vegetation Health, Saline Groundwater and Flooding on the Chowilla Floodplain South Australia'. CSIRO Division of Land and Water, Technical Report No. 20/04, May 2004, Canberra.

Overton, I.C., Jolly, I.D., Rutherford, K., and Lewis, M.M. (2005). 'Integrated Spatial Tools for Managing the Chowilla Floodplain Ecosystem'. Proceedings of the Spatial Sciences Institute Biennial Conference, September 2005, Melbourne.

Overton, I.C., Jolly, I.D., Middlemis, H. and Lewis, M.M. (2006). 'Managing a Regulated River Floodplain with Altered Hydrology and Surface-Groundwater Interactions, River Murray, Australia'. Proceedings of the International Conference on Riverine Hydroecology: Advances in Research and Applications TISORSII, August 2006, Stirling.

Overton, I.C., Jolly, I.D. and Lewis, M.M. (2006). 'A Spatial Model of Riparian Vegetation Health Based on Surface and Groundwater Interaction'. Proceedings of the International Multidisciplinary Conference on Hydrology and Ecology: The Groundwater/Ecology Connection, September 2006, Karlovy Vary.

Overton, I.C., Jolly, I.D., Slavich, P., Lewis, M.M. and Walker, G.R. (2006). 'Modelling Vegetation Health from the Interaction of Saline Groundwater and Flooding on the Chowilla Floodplain, South Australia'. *Australian Journal of Botany* 54: 207-220.

5.3.2 Habitat Suitability Model for Tree Health

The first attempt to model black box health on a floodplain scale considered the use of the critical threshold values of the main factors which affect black box health. Noyce and Nicolson (1993) considered black box at risk to be in areas of the floodplain not flooded at least 1 in 10 years, with a groundwater depth of less than four metres. Areas of black box frequently flooded (greater than 1 in 10 years return period), or with a groundwater depth greater than four metres, or with a groundwater salinity less than 20 dS/m (the flushed zone) were considered in good health. The results concluded that the spatial distribution of health patterns can be modelled successfully using a small number of variables within the GIS. The GIS contains a number of datasets including elevation and topography, groundwater contours, groundwater salinity, flooding extent and vegetation type. The model used a 50 metre resolution 0.2 metre vertical accuracy elevation model, vegetation data from Margules *et al.* (1990), flood extent from the nine flood maps available, and groundwater depth and salinity from bore interpolation.

A GIS model of black box health was developed by Hodgson (1993) as a simple three parameter model that considers groundwater salinity, flooding frequency and groundwater depth. The class model does not consider such factors as soil texture, rainfall or runoff, which have more localised effects. These latter factors may be used to explain some of the remaining variance in tree health. The effect of soils on vegetation, leaching and soil salinisation may be significant in some areas.

The three main parameters were chosen as potentially mappable on a spatial scale sufficient for delineation of tree communities, as well as being relevant to the processes described above. This model was used to assign a health class to all the black box vegetation on the floodplain. Table 5.4 shows the combination of parameters and threshold values used and the area of the floodplain covered by each class.

Table 5.4 Parameters, thresholds and areas of black box classes for GIS model (Hodgson, 1993).

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The accuracy of the class model was tested at 80 field sites and association was shown to exist between the GIS predicted values and field values for groundwater salinity ($p < 0.05$) but not for groundwater depth ($p > 0.05$). Results suggest that the distinction based on groundwater depth between classes 1 and 2 is not valid and does not explain the variation seen in vegetation health. The influence of flooding was shown to account for 65% of the changes seen in satellite imagery.

This model was later used to explain broad scale spatial patterns in the health of black box (Taylor *et al.*, 1996) (Figure 5.26). Comparisons between aerial photographs and GIS classes were used to quantify the association. The analysis of black box at 80 field sites showed that the six classes from the GIS model could be divided into two groups, which were significantly different ($p < 0.05$) in health. Good health (Classes 2, 4, 5 and 6) was associated with fresher groundwater (< 40 dS/m) or where groundwater is saline (> 40 dS/m) but flooding was frequent (more often than 1 in 10 years) or where groundwater is deep (> 4 m). Poor health (Classes 1 and 3) was associated with saline groundwater (> 40 dS/m) and infrequent flooding (less frequent than 1 in 10 years) where groundwater was <4m (Table 3.1). Taylor *et al.* (1996) found that the degree of spatial matching between the GIS health prediction map of Hodgson (1993) and a health assessment map interpreted from aerial photography was 85% using a grid point method. After re-analysis of the GIS class model using the same classes as Table 3.1 but improved vegetation, groundwater depth and groundwater salinity maps, the spatial matching was found to be 69%.

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Figure 5.26 The Chowilla GIS showing good and poor black box health using the habitat suitability model (based on Taylor *et al.*, (1996)).

It is likely that the habitat suitability model can be improved by using more parameters, for example including a soil map (Hollingsworth *et al.*, 1990). Observation suggests that the soil texture difference between clay soils of the floodplain and dune sands accounts for variations in the health of black box, with healthier vegetation situated on the sands. This suggests that infiltration of water by the sands dominates water loss by evaporation.

While the GIS class model was useful for vegetation mapping, it is less useful for predicting the effects of changes in management, as the classes are relatively broad and do not give any indication of the degree of health. Taylor *et al.* (1996) called for improvements to reduce the main sources of error, particularly those associated with the model itself (parameter type and threshold values), the quality of the data coverages and the remote assessment of vegetation health.

Empirical models may provide strong relationships between modelled and observed data. However, they are limited by their inability to model environmental changes outside the range of observed conditions. Process-based models allow predictions for an infinite range of conditions depending on the assumptions made in the model development.

5.3.3 Plant Growth Model - WAVES

WAVES is a daily time step soil-vegetation-atmosphere model that simulates the movement of water and salt in soils as well as plant water use and growth, responding to climatic changes, flooding frequencies and fluctuating water tables and produces a predicted Leaf Area Index (LAI) or leaf mass. WAVES was first applied at Chowilla by Slavich *et al.* (1999a).

Overton and Jolly (2003) applied the WAVES model on a range of sites on the Chowilla floodplain. The results from the WAVES modelling showed that the model was useful for predicting impacts from future scenarios on vegetation health and thus is able to inform policy for floodplain protection and salinity mitigation (Figure 5.27).

The WAVES model was used to predict the impacts of increasing environmental flows and lowering the water table through groundwater pumping. Figure 5.27 shows that the impact of lowering water tables produced greater results and less risk of dying during drought periods than the increased flows from the Living Murray 1,500 Gl/yr management scenario.

The model had many limitations including the lack of spatial extrapolation across the floodplain and the large number of parameters required to populate the model.

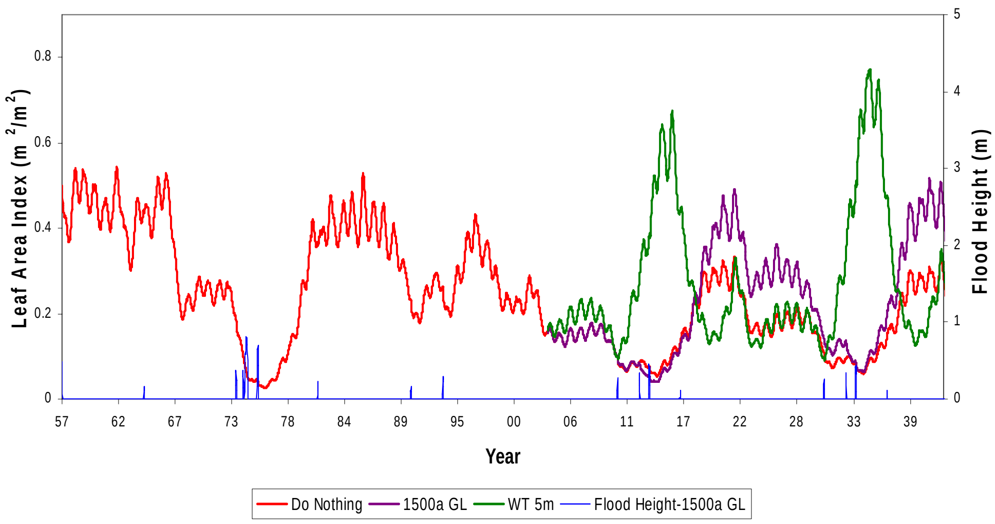

Figure 5.27 The WAVES model has been applied to the Chowilla floodplain and indicates that lowering water tables by 5 metres achieves more than the 1,500 GL flooding scenario (Overton and Jolly, 2003). The graph is for Black Box at Site 2 (Overton Site), soil type 2c, maximum WT Depth=3m, GW EC=18,000 uS/cm and elevation=20.2mAHD.

5.3.4 Steady State Moving Salt Front Model

Slavich (1999b) simplified the WAVES model concept by developing a Moving Salt Front (MSF) model by combining limiting vegetation characteristics, soil hydraulic properties and flood history to develop a salinity index which is indicative of vegetation health. The parameters for this quasi-steady state MSF can be feasibly represented within a GIS of the floodplain. Some of the soil hydraulic parameters of the WAVES model are incorporated into the MSF model so that long-term simulations with a fluctuating watertable can be compared with similar simulations conducted using WAVES. The WAVES model may also be used to evaluate the conditions for which the assumptions of the simpler steady state MSF model are satisfied.

Vegetation management in shallow water table environments requires an understanding of the interaction between the physical and biological factors which determine the rate of root zone salinisation. This rate is determined by the net groundwater discharge rate (vertical discharge minus recharge) and groundwater salinity. Groundwater discharge can occur by evaporation at the soil surface or by vegetation water use. The rate of groundwater discharge from bare soil depends only on the hydraulic properties of the unsaturated zone, whilst that from vegetated areas is affected by both soil hydraulic properties and vegetation characteristics such as leaf area, maximum rooting depth, the profile distribution of soil water uptake and the physiological tolerance to salinity of vegetation. The relative importance of biological and physical factors can be understood through the use of unsaturated flow and vegetation water use models.

The groundwater discharge rate, and hence soil salinisation rate, can be modelled using varying degrees of complexity. The least complex models assume that steady state conditions apply, (i.e. that the matric potential profile is constant) and that soil hydraulic properties are uniform with depth. With these assumptions, the potential groundwater discharge from the soil surface can be modelled using soil hydraulic properties only (Warrick, 1988). Warrick (1988) showed that the maximum upward water flux from a given water table depth could be represented using a simple power function whose coefficients depended on soil hydraulic parameters. Steady state groundwater discharge theory has been extended to consider a moving salt front (MSF) which is driven by transpiration of groundwater (Jolly *et al.*, 1993a).

This approach can be used to predict the effect of the salt front position on the groundwater uptake rate and to estimate the time scale for complete salinisation of soils on the Chowilla floodplain.

Jolly *et al.* (1993a) showed that soil salinisation occurring in the floodplains was due to increased discharge of saline groundwater caused by elevated water tables and reduced leaching of soils caused by less frequent flooding. While the salt front within the profile of a given soil may vary throughout the year in response to flooding, rainfall, water extraction by vegetation and groundwater fluctuations, over the long-term a dynamic salt 'balance' generally exists in which there is no net accumulation or leaching of salt (Jolly *et al.*, 1993a).

Soil salinisation can therefore be considered as a 'balance' between salt transport up the soil profile by groundwater discharge and downward leaching by flooding. This balance will be different for different soils, with more leaching on light textured sandy soils and less on heavy clayey soils. A full description of the derivation of the balance equation is given in Jolly *et al.* (1993a).

Steady-state soil limited groundwater discharge (q) is directly proportional to the power of the depth (z) of the water table below an evaporating surface (either the soil surface or the base of a plant root zone), and soil texture (Warrick, 1988) (Equation 5.1)

$$q = A\, z^{-n} \qquad (5.1)$$

where A and n are soil physical constants related to soil texture. Water available for leaching of salt from the soil (L) is related to the proportion of time the site is flooded (W) and the saturated hydraulic conductivity of the surface soil (K_s) (Equation 5.2).

$$L = K_s W \qquad (5.2)$$

So for a 'balance' between salt accumulation and leaching at a given site, (Equation 5.1) and (Equation 5.2) are equated and rearranged to define a 'critical depth', z (Equation 5.3).

$$z = (A \,/\, K_s W)^{1/n} \qquad (5.3)$$

The 'critical depth' is interpreted as the minimum water table depth (below the evaporating front) required to maintain the 'balance' and thus prevent long-term net accumulation of salt. An index of soil salinisation and therefore black box health can be given summarised from the above equations (Equation 5.4).

$$S = W^{-n} \,/\, z \qquad (5.4)$$

The soil salinisation index (S) is derived from the three major factors which affect the health; flooding frequency, groundwater depth and soil texture. This index can be used in areas of saline groundwater (greater than 40 dS/m). The advantages in using Equation (5.4) to give an index of soil salinisation over the threshold classes, is that it provides a value that can change with changing conditions. It therefore overcomes the limitations of the class model and allows the predictions of soil salinisation with varying management scenarios.

5.3.5 Salinisation Risk Model

To model the current spatial patterns of vegetation health using a process oriented model, the balance equation (Equation 5.4) was used. This required the use of the groundwater depth, flood extent and soil texture maps and is referred to as the Steady State Moving Salt Front Model as described above. As the equation does not include groundwater salinity, this factor was applied to the model as a separate step with areas of known recharge or the flushed zone given a zero salinity risk (S). Other areas were assigned the salinity risk based on Equation 5.3. Floodplain inundation percentages (W) were determined using Lock 6 pool levels recorded over the last 25 years. The model predictions for the vegetation health in 1988 are shown in Figure 5.28.

This model can be used to classify the black box vegetation as good and poor health. An example of predicted vegetation health in 1994 is shown in Figure 5.29 and can be compared to the **Habitat Suitability Model** (Figure 5.26) and the 'actual' health in Figure 5.2. A critical (S) value of 2.5 was chosen to separate the two classes. This value was determined to give the best spatial matching with aerial photograph interpretations.

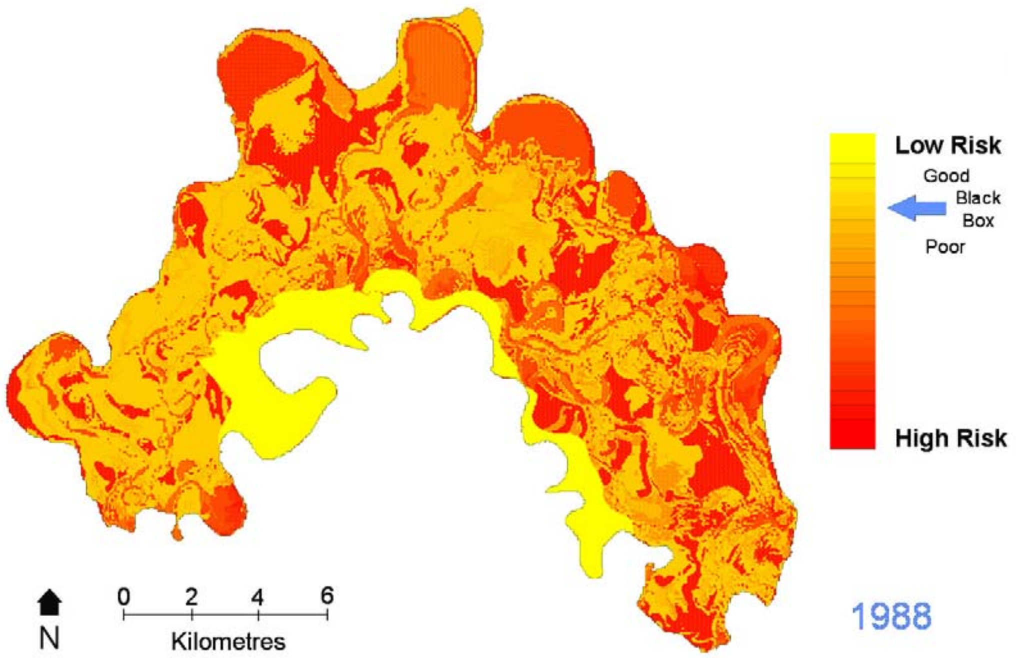

Figure 5.28 The salinity risk model for Chowilla showing good and poor black box health for 1988. Note the flush-zone has been given a zero salinity risk based on low groundwater salinity.

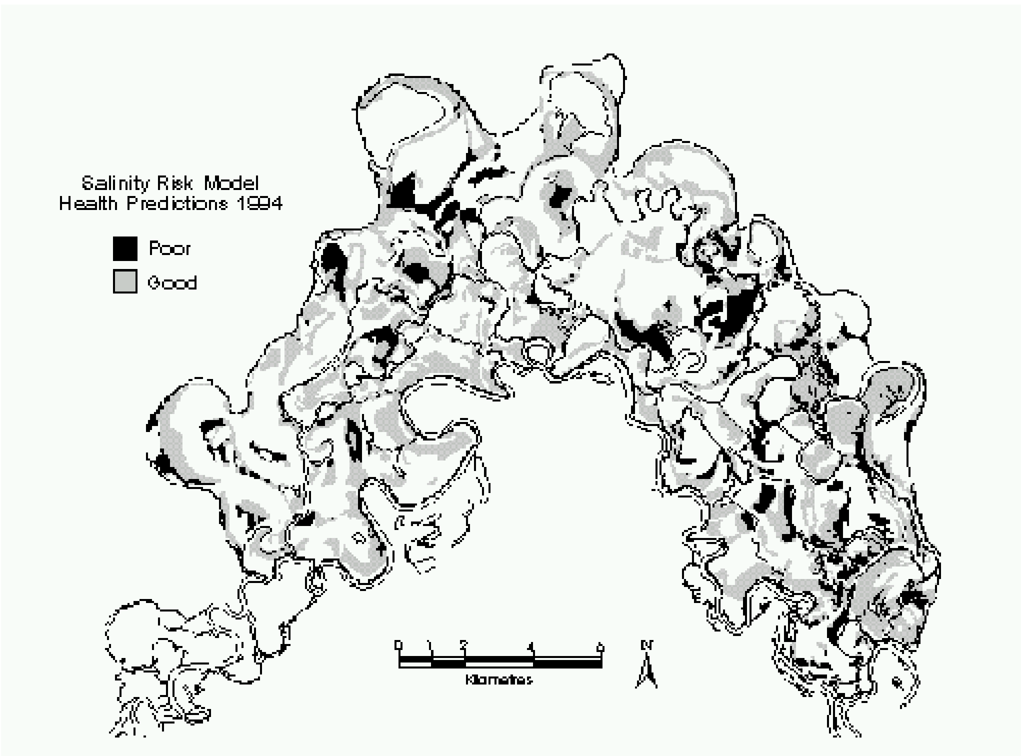

Figure 5.29 Salinity risk model predictions for black box health (1994) for the Chowilla floodplain. Poor and good classes divided on salinity value of S= 2.50.

It is interesting to note that using a constant (n) value for the soil hydraulic property (i.e. assuming the same soil type over the floodplain) does not reduce the spatial correspondence between the two maps by a large degree (65% instead of 70%) (Table 5.5 and Table 5.6). We assume that the soils do vary across the floodplain and that soil hydraulic properties do impact on soil salinisation. The small improvement in spatial correspondence when soils are divided into two texture classes is likely to be a factor of the accuracy of the soil data and their categorisation.

Table 5.5 Contingency matrix showing the degree of spatial matching between salinity risk model predictions of black box health (Figure 5.29) and interpretation of health from aerial photographs (Figure 3.1) as a proportion.

Spatial Correspondence: Salinity Risk Model vs Aerial Photograph Interpretation	Salinity Risk Model Prediction Good	Salinity Risk Model Prediction Poor	Matching areas
Aerial Photo Int. – Good	0.51	0.08	
Aerial Photo Int. - Poor	0.21	0.19	
			0.70

Table 5.6 Contingency matrix showing the degree of spatial correspondence as a proportion between flood extent differences and detected changes in the vegetation cover from satellite imagery between 1988 and 1995 for different soil textures.

Spatial Matching: Flood In. Diff. vs Change Detection for diff. Soils	Increase in Flood Extent	No Increase in Flood Extent	Matching areas
Light (Sandy) Soils			
Increase in Vegetation Cover	0.25	0.11	
No Change in Vegetation Cover	0.24	0.40	
			0.65
Heavy (Clayey) Soils			
Increase in Vegetation Cover	0.31	0.11	
No Change in Vegetation Cover	0.24	0.34	
			0.65

Although the differences were not significant, the percentage of black box that had increased (as detected by remote sensing) but was not predicted to have increased (GIS model) was greater for sandy soils (30%) than clay soils (27%). This suggests that the flood has a greater effect on black box on sandy soils than on clay soils. It could also imply that the critical depth to groundwater for soil salinisation is lower on sandy soils than for the clay soils, as was predicted by Jolly *et al.* (1993a).

The effect of considering groundwater salinity using the flushed zone and the recharge areas was to improve the spatial matching from 56% to the 70%. Again the accuracy, or more particularly the resolution, of the salinity data would limit improvements, although an increase of 14% is sufficient to demonstrate the importance of groundwater salinity to vegetation health. The major limitations in the model's predictive capacity are seen to be the accuracy of the groundwater depth and soil texture maps and the resolution of the groundwater salinity data.

Other factors will play a role in the vegetation health changes between these two dates including differences in the groundwater table, rainfall, soil salinity and groundwater salinity, as well as other factors such as grazing pressure, weed and intra-specific competition and density dependent inter-specific competition.

This process model is an improvement on the **Habitat Suitability Model** in assessing environmental flow and groundwater management scenarios on tree health. However, this approach does not incorporate groundwater salinity or allow the temporal effects of management scenarios to be assessed as it is a steady-state model.

5.3.6 WINDS

Model Equations

The simple balance model (steady state **Moving Salt Front Model** of the long-term salt balance) as described above is not dynamic and cannot model future changes based on changing conditions. The MSF model can be implemented over time by calculating the index of salinisation using Equation 5.4 (Slavich, 1997). Slavich (1997) defined a method for calculating an index of salinisation as the sum of the water availability for three recent time periods. Slavich (1997) proposed that the WINDS calculation could be used in a spatial grid model and used to predict vegetation health across the whole floodplain. Slavich (1997) defined a method for calculating a Weighted Index of Salinisation (WINDS) as the sum of the water availability for three recent time periods (Figure 5.30).

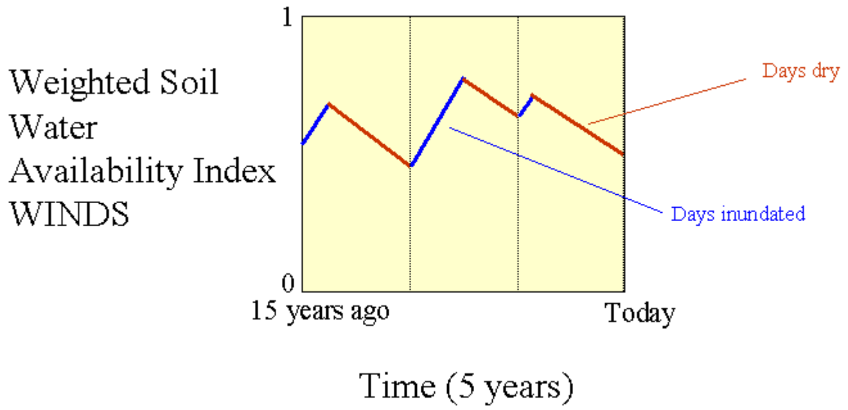

Figure 5.30 Hypothetical WINDS soil water availability index over a fifteen year period showing the increasing soil water availability during floods and decreasing availability during drought.

The WINDS index is calculated firstly from the groundwater discharge rate q (Equation 5.5) (Slavich, 1997).

$$q = A\ (Zw - Zf)^p \qquad (5.5)$$

Where A and p are soil parameters, Zw is the groundwater depth and Zf is the depth of the salt front. Within the GIS the first step is simplified to calculate the groundwater discharge rate q for each grid cell (Equation 5.6).

$$q = A\ (Zw)\ p \qquad (5.6)$$

Where A and p are soil parameters and Zw is the groundwater depth. This differs from Equation 5.5 as it does not consider a depth to the salt front. This information was not available and when estimated proved to make the discharge rate unrealistically high when the depth of the salt front approached the depth of the groundwater. The second step is to calculate the soil salinity for each 5 year time period (Csi) for each grid cell (Equation 5.7).

$$Csi = Csi\text{-}1 + Cg \,/\, Zwmax \,(\, q * td \,/\, qd - Ks * ts \,/\, qs \,) \qquad (5.7)$$

Where Cg is the groundwater salinity and Zwmax is the maximum water table depth. td is the number of days not flooded (dry) and ts is the number of days flooded (saturated). Ks is the saturated hydraulic conductivity and qd and qs are the soil water content during discharge and flooding periods respectively. The third step is to calculate the soil water availability for each 5 year time period (Xwi) for each grid cell (Equation 5.8).

$$Xwi = 1 - (\, Csi \,/\, Clmin \,) \qquad (5.8)$$

Where Csi is the soil salinity for that time period and Clmin is the limit of the soil salinity the plants can withdraw water against. The final step is to calculate the cumulative soil water availability (WINDS) for the last 15 years for each grid cell (Equation 5.9).

$$WINDS\ index = \Sigma\, Xwi * Wi \qquad (5.9)$$

Where Xwi is the average soil water availability for each time period and Wi is the weighting factor for each time period. Weighting factors can be chosen depending on the species and the time that water availability conditions can represent themselves in the health of the plant. Black box trees respond very slowly to changes in water availability as a result of their conservative nature in a highly saline drought environment. River red gum trees are more responsive than black box and will produce leaves sooner after a flood event (Overton and Jolly, 2003). The weighting factors were chosen as 1 = 0.55 for the last five years, 2 = 0.30 for the previous five years and 3 = 0.15 for the first five years. These weighting factors were chosen to be equal for both black box and river red gum as no data was available on which to base a difference.

The calculations are performed and a WINDS index between 0 (dead) and 1 (good health) is produced. The index is actually an indication of the soil water availability but is indicative of tree health. Trees in an environment of highly saline soil water at their root tips will have less water available for uptake by the tree for photosynthetic respiration purposes and will therefore have limited health potential. The WINDS index is calculated as the average water availability over the past 15 years (Figure 5.30) and is therefore a good indicator of the tree health at the end of the period, given the time response of floodplain trees.

The WINDS model is then incorporated into a GIS providing a spatial model of vegetation health which considers the impact of flooding history. The parameters used within the GIS model are similar to the ones used for the steady-state MSF model and the WAVES modelling above. The model outputs are highly susceptible to changes in the initial salinity of the groundwater at the beginning of the period and on the soil hydraulic parameters. These have to be estimated in most circumstances to produce reasonable results.

The combined index relates well to the weighted temporal average leaf area index modelled using WAVES, giving confidence for using the MSF model for management scenarios within the GIS.

Model Parameters and Spatial Datasets

Implementation of the WINDS model requires the following spatial layers which are described for the Chowilla application:

- Vegetation community map merged from the South Australian (DEH, 2006) and the New South Wales (DSNR, 2003) data and simplified to provide vegetation community classes;
- Vegetation health map of the three dominant tree species derived from vegetation mapping, updated based on field observations;
- Flood extent maps for a range of flows and weir configurations derived from the RiM-FIM;

- Soil map of hydraulic properties to provide potential soil leaching, aquifer recharge and salt accumulation rates;
- Groundwater depth map supplied by Yan *et al.* (2005); and a
- Groundwater salinity map derived from airborne electromagnetic imagery, vegetation vigour change detection and soil mapping.

The WINDS model assumes that current tree health is indicative of the soil water available to the plant during the last fifteen years, with greater emphasis on the last five years decreasing with each five year period. This is likely to be true for black box trees which respond very slowly to flood events. WAVES modelling has shown an approximate 10 year response time to a major flood event (Overton and Jolly, 2003). River red gum trees respond much quicker to flood events than black box and also decline much quicker during droughts.

The WINDS model parameters include the following.

Number of days inundated from 1957 to 2003

The numbers of days inundated and dry during five year periods from 1957 to 2003 were calculated from recorded hydrographs. Figure 5.32 shows the hydrograph for this period of time. The starting date of 1957 was chosen to follow the large 1956 flood which is the largest on record, estimated at over 350,000 ML/day. This allowed the soil salinity of the starting date to be considered as 0 dS/m, i.e. totally fresh, as a result of the long leaching period during the 1956 flood event.

Weighting on each of the three five year periods

Tree health predictions were based on a fifteen year period of flows broken into three five year units. The most recent five years has the greatest impact on the prediction of vegetation health at the end of that 15 year period, with decreasing influence from the middle and then the earliest five year period. The weightings used were, as discussed above: last 5 years = 55%, middle five years = 30% and 15% for the initial five years.

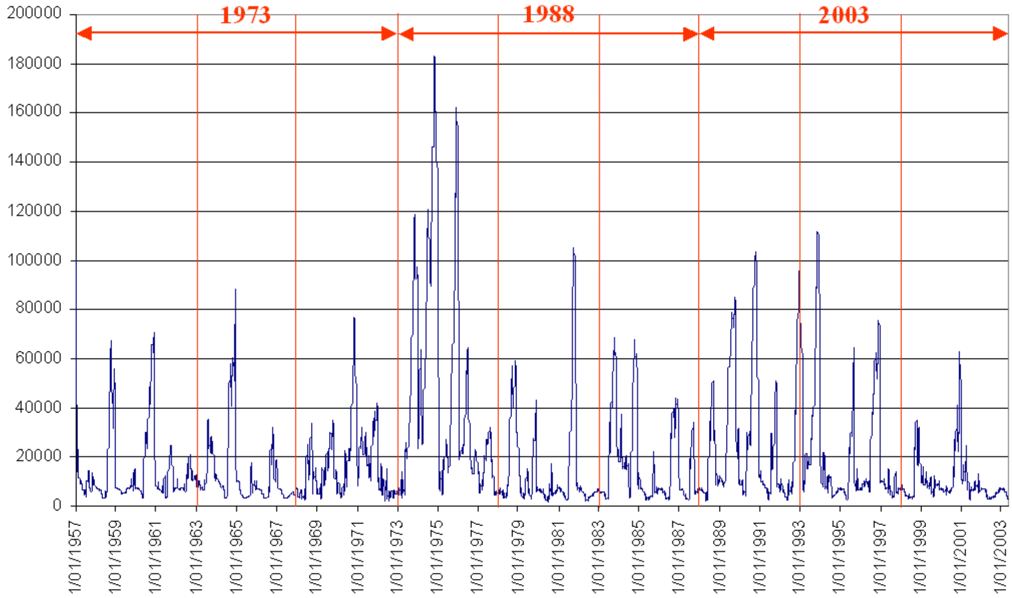

Figure 5.31 Hydrograph of River Murray flow at the border of South Australia from BigMOD. The graph clearly shows the dry period observed in the last five years. The red lines indicate the time period used for the 1973, 1988 and 2003 predictions.

Groundwater salinity

The groundwater salinity coverage was initially divided into three areas to allow for a much fresher zone of groundwater close to the river in the flush zone. It was noticed from bore readings that the 'flush zone' could contain groundwater with salinity as high as 33 dS/m. It is suggested that the proximity of the main river channel has a significant effect and the low salinity zone decreases rapidly to no effect within a few hundred metres. A thin band of 50 metres around the river was therefore set to 5 dS/m, the flush zone set to 19 dS/m and the remaining floodplain set to 50 dS/m. The initial groundwater salinity conditions at the start of 1957 were set at 5 dS/m in the flush zone and 30 dS/m outside this area. By incorporating the electromagnetic data interpretation and identifying the lateral creek recharge areas and local recharge areas a new groundwater salinity map was used in the WINDS modelling (previous section). The process of using the new groundwater salinity map was to change the initial salinity conditions to include areas of 5 dS/m in the areas identified as lateral bank recharge areas (Figure 5.20) and local recharge areas (Figure 5.22). The groundwater salinity map was also changed to include salinities of 19 dS/m for local recharge areas and bank recharge areas.

Groundwater depth

The groundwater depth coverage was derived from a MODFLOW model (Yan *et al.*, 2005).

Limiting soil salinity for river red gum and black box

A soil salinity limit for river red gum was set at 30 dS/m and 55 dS/m for black box. These values represent a conservative limit for healthy trees. Salinity of the water available to the plants is determined by the ability of the plant to lower it's root water potential below that of the adjacent soil, causing water to move down the hydraulic gradient from the soil into the plant. The minimum predawn water potential measured for black box trees growing on the River Murray floodplain is ψ_{lmin} = -3.5 MPa (Eldridge *et al.*, 1993; Zubrinich, 1996). This corresponds to a threshold osmotic potential for extraction by black box of C_g = -3.0 MPa, or a groundwater salinity approximating seawater salinity (approximately 35,000 mg/L equivalent to 60 dS/m EC or 60,000 µS/m EC). Similarly, the minimum predawn water potential measured for river red gum growing on River Murray floodplains is ψ_{lmin} = -2.2 MPa (Mensforth *et al.*, 1994). This corresponds to a threshold osmotic potential for extraction by river red gum of C_g = -2.0 MPa, or a groundwater salinity approximating two-thirds seawater salinity (approximately 25,000 mg/L equivalent to 40 dS/m EC or 40,000 µS/m EC).

The vegetation map developed in Section 5.2 was used to identify the distribution and health of black box and river red gum and was used to validate the model results.

Soil variables

The soil variables (Table 5.1) were derived from an assessment of average soil properties (Overton and Jolly, 2003). Very small changes in these parameters influenced vegetation health predictions. After an initial set of variables was employed, a poor spatial correspondence was achieved. This can be attributed to the generalised nature of the soil mapping, both in spatial distribution and the soil properties. The hydraulic properties were considered to have the least confidence so a process of refinement of these to improve the health predictions was undertaken (Table 5.1 lists the final values). This alteration of parameters to improve the model was considered necessary to calibrate the variables to achieve the best spatial correspondence of the tree health predictions at the current time period in order to have confidence in predicting future conditions.

WINDS Model Results

The results of the WINDS model show the weighted index of salinisation ranging from 1 (good health) to approximately -2 (dead trees). For ease of communication the WINDS values can be grouped into two classes of good (above or equal to 0.96) and poor health (below 0.96) determined by comparing results with actual vegetation health data. Figure 5.32 shows the prediction of vegetation health in 2003 and can be compared with known vegetation health in Figure 5.2, the Habitat Suitability Model Figure 5.26 and the Salinisation Risk Model Figure 5.28.

There is a 65% spatial correspondence between the predicted health versus the recorded health (Table 5.7). This figure is similar to the health predictions of the Class Model and the Salinisation Risk model. Although not an improvement, it does represent a reasonable accuracy for a simplified process model that can be used to predict changes in groundwater depth and salinity and flooding frequency.

The largest area of mis-classification occurred for the trees that were recorded as good in the field and predicted as poor in the model. This may be occurring due to rainfall improving the water availability or that the model predicts these areas as poor but there may be a time lag in vegetation response in the field, meaning they still fall in the 'healthy' category as seen on the ground.

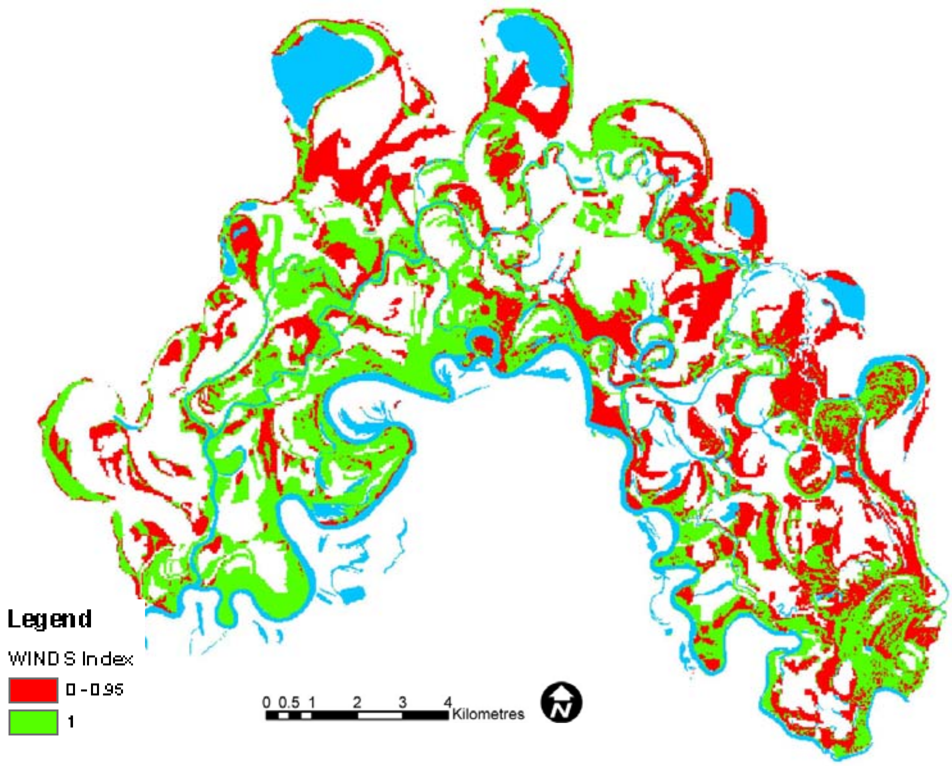

Figure 5.32 Vegetation health prediction for black box and river red gum trees (2003).

Table 5.7 Vegetation health prediction for black box and river red gum trees. Predicted from the WINDS model (2003) versus recorded health mapping.

	Predicted Good	**Poor**	**Total**
Recorded Good	1,900 ha 23%	1,900 ha 23%	3,800 ha 46%
Poor	1,050 ha 12%	3,500 ha 42%	4,550 ha 54%
Total	2,950 ha 35%	5,400 ha 65%	8,350 ha

The WINDS index values were then divided into three classes of good, poor and dead. This process was necessary to interpret the WINDS index into tree health classes as recorded by field assessment. The separation of these classes was achieved by comparing the accuracy results with the vegetation health map until the best accuracy was obtained. The class breaks chosen are 'Good' for 1 to 0.96, 'Poor' for 0.95 to 0.5 and 'Dead' equal to 0.5 to 0. Figure 5.32 shows the three WINDS index classes for vegetation health in 2003. Once these classes were defined they were then used in all predictions of tree health.

The WINDS model predicted decline in the high elevation black box areas even though the groundwater was deep. This is because these areas are infrequently flooded. However, it is known that black box trees on the sandy high elevations can survive on rainfall alone. To model this, and to include all water sources, rainfall records were analysed for volume of water and compared to the maximum infiltration rates of the soil. When the rainfall volume was large enough to provide infiltration rates equivalent to a flood (2 mm/day), the period was considered equivalent to a flood.

The WINDS model was also improved by incorporating a drought factor that addressed the dying trees due to drought or lack of water availability from water content more than salinity.

It was observed that river red gum trees died on the Chowilla floodplain after a period of five years without flooding (MDBC, 2005). This five year period was taken as a critical time threshold for when a trigger would be applied in the WINDS model to decrease the vegetation health if no flooding had occurred. A decrease of 0.3 WINDS was applied to trees if a five year period occurred with no flooding and referred to as the 'Drought Factor'. A 0.3 WINDS index was considered appropriate as it reduced good trees with a WINDS index of 1 to 0.7 and therefore the poor class. No good health class tree could therefore remain good after the drought factor was applied.

The drought factor is a rudimentary simplification of the consequences of lack of water to the soil profile but was found to be useful in identifying areas of trees that become poor for reasons other than salinisation. This drought factor was thought to only occur where trees receive water from an alternate water supply such as directly from a creek or through flooding recharge. Areas identified as bank and local recharge in the recharge mapping were then identified as the drought factor areas.

Modelling of water sources by including a simple decrease in groundwater salinity for creek and fresh water lenses, 'rain flood days' and a 'drought factor' improves the model prediction accuracy from 65% to 72% (Figure 5.33). This is an improvement in the spatial correspondence of the model, as well as an improvement in the capability of the model to consider changes in rainfall, creek heights and further analysis of flooding frequencies. Modelling of plant water sources could be improved by modelling the soil water balance.

The model was run using constant variables to test its sensitivity. The model sensitivity to groundwater depth, groundwater salinity, flooding frequency and soil types was tested by individually changing these variables to a constant surface, i.e. removing them as variables from the model. These model sensitivity outputs produced spatial correspondence results below 50% when compared with actual tree health. This suggests that these variables are all required to successfully model the spatial variation in floodplain tree health.

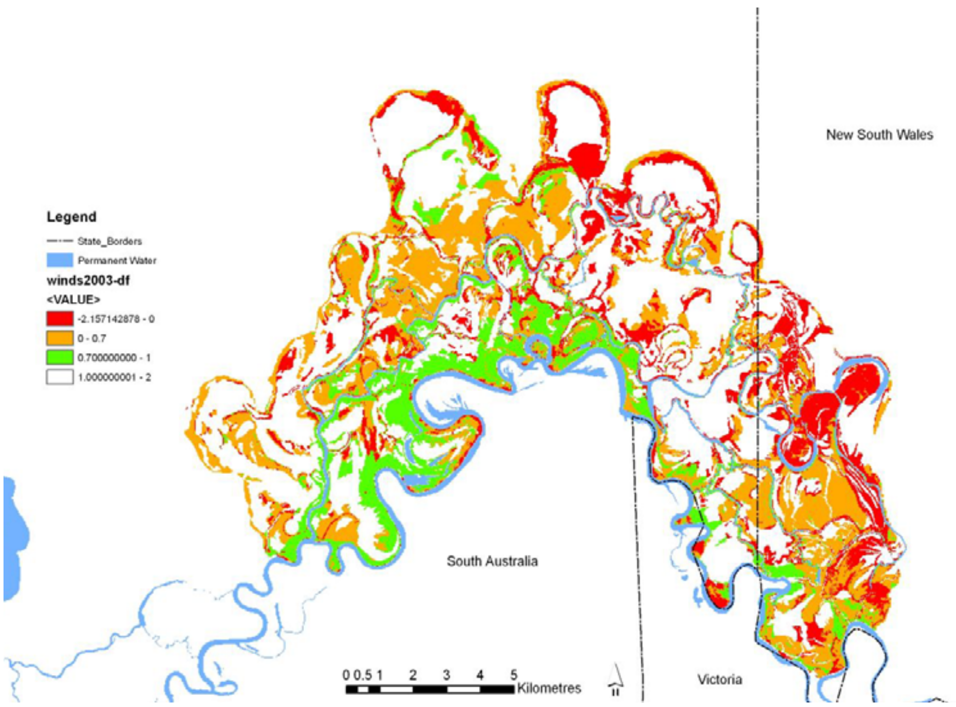

Figure 5.33 Vegetation health prediction for black box and river red gum trees (2003). Red for dead, orange for poor and green for good health.

The major predictions of the WINDS modelling are that approximately 45% to 55% of vegetation on Chowilla was dead or in poor health in 2005. WINDS modelling has produced a good spatial correspondence with observed vegetation health (72%). WINDS modelling allows changes in groundwater and flow regimes to be assessed for impacts on floodplain tree health, and define the areas which are most affected by the two management scenarios. The results also show that minor changes in soil hydraulic properties can have large effects on vegetation health. The exact timing of catastrophic events such as the river red gum dieback in summer 2003 is hard to predict, but future modelling should be able to predict the degree of risk of occurrence.

The WINDS model also produced a map of soil salinity as a result of the second step (Equation 8.2). The soil salinity map is useful for investigating the impacts on understorey vegetation types and for comparing the amount of leaching that is occurring in the floodplain. Figure 5.34 illustrates soil salinity predicted for 2003 before calculation of the WINDS index. The WINDS predicted salinity values will be used in the next section to examine the natural and altered distribution of vegetation to soil salinity.

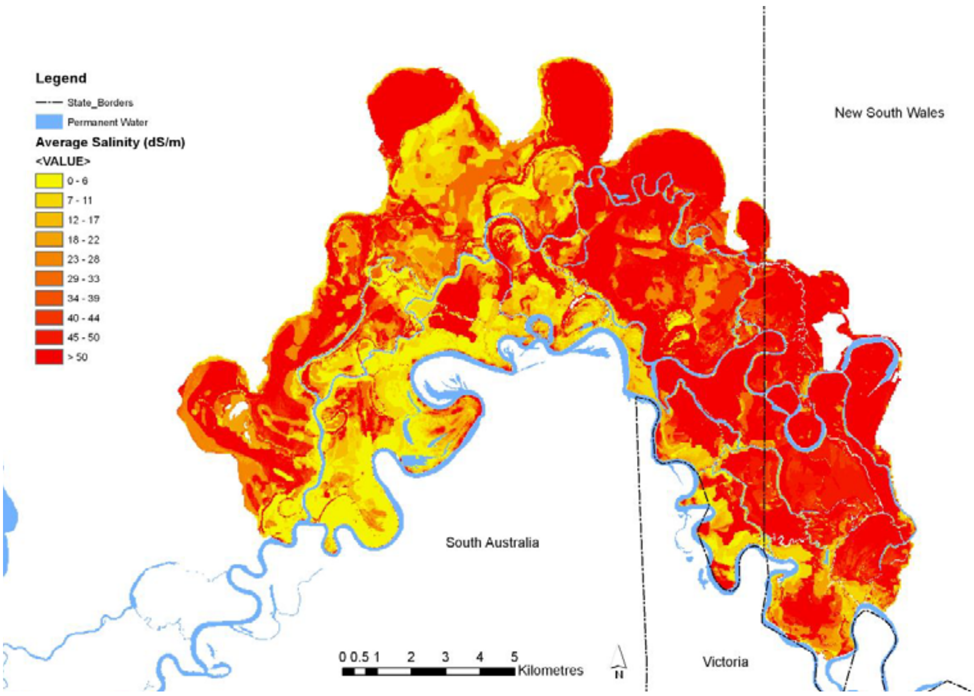

Figure 5.34 WINDS predicted soil salinity for the Chowilla floodplain (2003).

The distribution of tree health can be compared against predicted soil salinity (Figure 5.35). In areas of high salinity, above 30 dS/m, most of the trees are dead. However in the areas of low salinity there are numerous dead and poor trees. This may be due to drought, as a result of matric potential stress or lack of physical water present. Insect and disease are minor factors in relation to tree stress on the floodplain. The presence of so many dead and poor trees in low salinity areas suggests soil water availability, including soil moisture, needs to be considered when modelling floodplain vegetation even in a highly saline environment such as the lower River Murray floodplain.

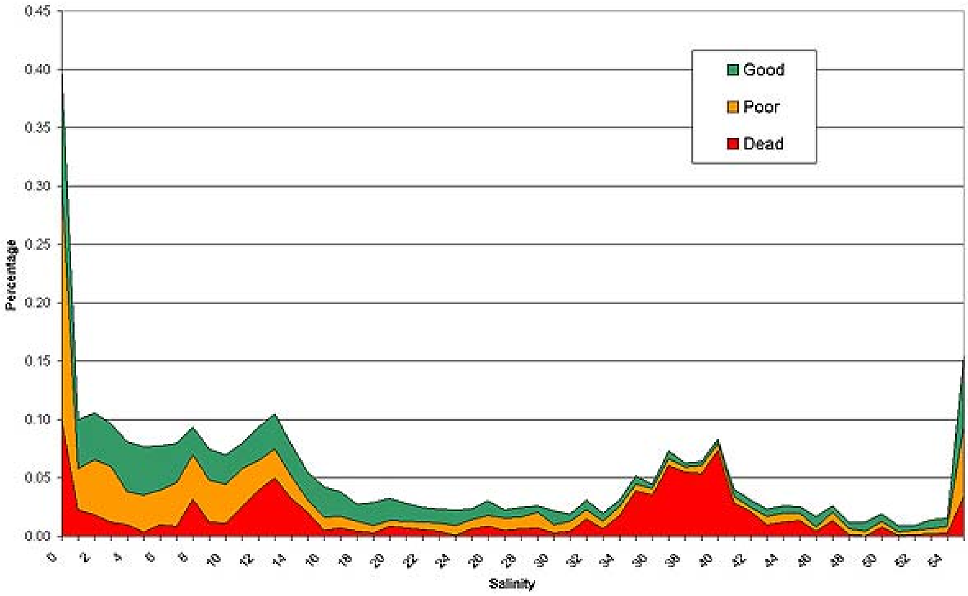

Figure 5.35 The distribution of tree health across the range of soil salinities (dS/m) modelled from the WINDS model.

Limitations of the WINDS Modelling

The main limitation of the WINDS model is that it does not consider tree density. Vegetation growth and health are not modelled directly but through the indicative variability of soil water availability. The model operates on a 30 by 30 metre grid, providing the soil water available for that area regardless of tree density. The model does not consider whether there is one tree or 50 trees in this 900 square metre area. With a given soil water availability, one tree could be very healthy but 50 trees would be competing for this same water and would therefore be poor.

Discharge rates of groundwater are modelled from evaporation rates and do not consider plant transpiration. The greater the number of trees in one grid cell, the greater the transpiration and the faster salt will accumulate.

A second major limitation is that soil water storage, and capillary fringe of groundwater and fresh water lenses are the sole water sources for the floodplain trees, whereas trees on the edges of creeks are known to use creek water directly.

The time lag of tree response to available water may not be modelled correctly. More research is required to understand the response of black box, river red gum and other vegetation on the floodplain to changing water availability conditions.

The model assumes that no structural damage occurs to the trees during poor conditions and age structure is not considered. Regeneration, recruitment and senescence are not considered but would have implications for the longevity of the population. Stress from insect attack, mistletoe and disease are also not included in the modelling.

The WINDS model does not provide for groundwater depth changing over time during flooding and drying cycles. In real flood events, groundwater rises closer to the surface and could therefore affect vegetation in these areas, especially if they did not actually receive surface flooding. After flood events, the groundwater levels drop to pre-flooding conditions. Given the short time span of most flood events, and the complexity of modelling required to simulate a fluctuating water table, a stable groundwater surface has been used.

The spatial resolution of the model does not represent the local variation in topography that can cause single trees to die or survive as a result of greater depth to groundwater or local accumulation of rain water.

5.3.7 Discussion on Modelling Floodplain Tree Health

A spatial model of soil salinisation (WINDS) was created using the major salt accumulation factors and used to infer vegetation health. The model is capable of combining the effects of groundwater depth, groundwater salinity, flood frequency and soil type to determine vegetation tree health on a regional scale. The model was then extended to consider recharge areas, rainfall and drought, as water source factors affecting health.

The WINDS model goes beyond traditional finite element groundwater modelling, as it is specifically focused on vegetation health, and is able to accommodate more processes than just groundwater depth-salinity relationships. The model has been shown to successfully predict the spatial extent of vegetation tree health on the Chowilla floodplain, with approximately 72% correspondence with field recorded health. The methodology is particularly useful due to its limited parameter requirements compared with most vegetation health models and can be used in a temporal predictive manner. The model has been instrumental in the development of strategies for groundwater management at Chowilla, the Living Murray Foundation Report (MDBC, 2005b), the development of the environmental regulator and the Chowilla Environmental Water Management Plan (MDBA, 2012).

The WINDS model also produces a soil salinity map which is useful for identifying reductions in salinisation from management scenarios and the impact on understorey vegetation.

The major limitation of the WINDS model is that it does not consider tree density in its assessment of health from water availability and its modelling of transpiration and therefore rate of salt accumulation. Although the model does take into consideration a moving salt front affected by leaching and discharge, the actual depth to water table across the floodplain does not fluctuate over time with flooding events that is modelled using the dynamic model WAVES. However, there are advantages in using the simpler quasi-steady state MSF model as it does not rely on complex environmental parameters being recorded over time or derived mathematically, and it can be adapted to be used within a GIS. The GIS provides a spatial model of the floodplain and can be incorporated into decision support systems for flow management.

However, the model was not designed to be used for predicting changes over short time periods. The GIS model does not incorporate changes in groundwater depths and therefore cannot accurately predict changes with management strategies. Jolly *et al.* (1993a) has previously identified that there is uncertainty in the understanding of how long the large amounts of salt stored in the soil profile would take to be leached under improved flooding regimes. Moreover, it is uncertain how a given degraded vegetation community will respond to changes in the depth of low salinity groundwater as a result of increased flooding. The presently available data and the model used are insufficient to answer these questions.

The mapping and modelling undertaken has simplified the spatial variability of the floodplain into distinct classes and regions that depict homogeneous environments and vegetation. In reality, there can be large variability in soil types, groundwater depth and salinity and vegetation health at a localised scale. Large differences can exist in depth of overlaying clay, affecting soil hydraulic properties, groundwater salinity and depth over small distances. It is not uncommon to see a live tree in the middle of a patch of dead trees, and vice versa, with no apparent cause. It is likely that the spatial variability of soils may account for these health differences and the aim has been to model the patch as a whole with an average health prediction.

The modelling has contributed to understanding of floodplain vegetation health processes by modelling the presence of freshwater lenses and the prediction of drought related die-back of riparian vegetation. The previous modelling has determined that the major cause of discrepancy between actual and modelled vegetation health from soil salinisation was the physical lack of available water. The conceptualisation of this soil water availability and freshwater lens storage of flood recharge advances the understanding of floodplain recharge processes. A soil water budget model could be developed and incorporated into the spatial floodplain model. The resulting water availability model could then be used for temporal predictions of drought and soil salinisation.

The WINDS model is potentially transferable to other floodplains and other floodplain species, using different parameters or different thresholds of the existing parameters.

5.4 IMPACT OF FLOODPLAIN SCALE MANAGEMENT SCENARIOS

5.4.1 Introduction

The objective of this section is to apply the WINDS models developed in Section 5.3.6 to determine the impact of groundwater and surface water management scenarios and climatic change on riparian vegetation on the Chowilla floodplain as a demonstration of the capability and limitations of the WINDS model.

Management scenarios proposed by the South Australian Government for the River Murray include environmental flow strategies and groundwater interception schemes. Groundwater interception schemes are intended to intercept salt before it enters the river, to reduce salt loads in the River Murray. However, recently salt interception schemes have also considered the addition of bores to target environmental benefit. Environmental flow strategies include increased river levels and floods using greater river flows or new control structures, manipulation of river levels through the operation of weir and control structures, and the supply of water through channel irrigation and pumping.

Once the WINDS model had been calibrated with vegetation health, a number of scenarios were considered for their impact on soil salinity, tree health, vegetation distribution and inundation of creeks and wetlands. The results of the different scenarios were analysed for improvements in floodplain tree health and are presented below. The scenarios included:

- Do-nothing and predictions over a 30 year period with the last 30 years of flows repeated;
- Do-nothing and predictions over a 30 year period with the last 15 years of flows repeated;
- Groundwater lowering by a proposed groundwater pumping scheme;

- Increased environmental flows from the 1,500 GL/yr MDBC Living Murray program;
- Raising of the Lock 6 weir to 19.87 metres; and
- Operation of an environmental regulator at the downstream end of Chowilla creek.

These management options were chosen to provide key inputs into management decision making as these management scenarios were requested by the South Australian Government. They were also chosen to demonstrate the range of possible management options for which the WINDS model can predict tree health outcomes.

A critical factor in predicting floodplain condition is the future predictions of river flows. The methods chosen here to estimate future river flows were to repeat the period from 1998 to 2003, which have been the worst on record since 1956 and are very relevant given that they represent current conditions, and to repeat the fifteen years from 1957 to 2003. Another future flow scenario that has been considered is the increase in flows from the Living Murray initiative of 1,500 GL/yr. The Living Murray also investigated 750 GL/yr and 500 GL/yr enhanced flow regimes which could also be modelled.

When considering the impact of management scenarios on vegetation health, it is important to not only consider the extent of inundation but also the impact of inundation on fringing trees. It is known from field based studies that the impact of flood extent can extend beyond the edge of flood waters, through groundwater recharge. The distance can vary and is dependent on soil type, vegetation type and depth of flood water at the edge of the flood. Field observations indicate this can extend from 0 to 100 metres, but usually at least to the extent of the fringing river red gum. For modelling, the effect of changing flood extents has been considered by including a 50 metre buffer from all water courses.

Components of the methodology, results and discussion of this section have been presented in the following publications:

Overton, I.C., Rutherford, J.C. and Jolly, I.D. (2005). 'Flood Extent, Groundwater Recharge and Vegetation Response from the Operation of a Potential Weir in Chowilla Creek, South Australia'. Report for the South Australian Department of Water, Land and Biodiversity by the CSIRO Division of Land and Water, Canberra.

Overton, I.C., Slarke, S. and Middlemis, H. (2006). 'Chowilla Management Options'. Report for the South Australian Department of Water, Land and Biodiversity by URS Pty Ltd, the CSIRO Division of Land and Water and Aquaterra Pty Ltd, Adelaide.

Overton, I.C. and Doody, T.M. (2008). 'Groundwater, Surface Water, Salinity and Vegetation Responses to a Proposed Regulator on Chowilla Creek'. Report for the South Australian Murray-Darling Basin Natural Resource Management Board by the CSIRO Water for a Healthy Country National Research Flagship, Canberra.

Overton, I.C. and Doody, T.M. (2010). 'Ecosystem Response Modelling in the Chowilla Floodplain and Lindsay-Wallpolla Islands'. In: Saintilan, N. and Overton, I.C. (eds.) 'Ecosystem Response Modelling in the Murray-Darling Basin'. CSIRO Publishing, Canberra.

5.4.2 Tree Health Response to Management Scenarios

The WINDS model was used to predict tree health outcomes from a range of management scenarios. Overton *et al.* (2006) undertook a detail analysis of management options at Chowilla using the WINDS model and prepared cost benefit assessments of each management option. Each scenario is discussed below and a summary of the tree health response to the different management scenarios and combinations of scenarios is presented in table 5.8. The inundation or water supply to 63% of the river red gum forest and woodland and 19% of the black box with the proposed environmental regulator is a long way toward achieving the targets of 70% and 20% set by the MDBC's Living Murray program.

The results indicate that the best management option is the combination of the construction of an environmental regulator and the instalment of a salt interception scheme. This option achieves 40% of vegetation in good health.

Table 5.8 Results for current and future predictions (total area is 8,582 ha).

Management scenario (30 year predictions)	Dead (Area ha %)	Poor (Area ha %)	Good (Area ha %)
Current	2,630 30%	3,909 46%	2,043 24%
Do-nothing	4,060 48%	2,854 33%	1,668 19%
Groundwater interception (SIS)	3,831 44%	3,073 36%	1,678 20%
Flow enhancement	3,860 45%	2,670 31%	2,052 24%
Weir raising	4,043 47%	2,743 32%	1,796 21%
Proposed new environmental regulator	3,043 35%	2,386 28%	3,153 37%
Channels	4,079 47%	2,826 33%	1,677 20%
Wetland pumping	4,009 46%	2,806 33%	1,767 21%
SIS and weir raising	3,703 43%	2,985 35%	1,894 22%
SIS and proposed environmental regulator	2,536 30%	2,610 30%	3,436 40%
Weir raising, channels and wetland pumping	3,840 45%	2,604 30%	2,138 25%

Do-Nothing Scenario Future Predictions

The do-nothing future predictions are necessary to firstly investigate the future of the floodplain with no intervention and secondly to provide a benchmark for the comparison between the different management scenarios. Groundwater depth for a 30 year prediction with no management intervention was provided by Aquaterra Pty Ltd (Overton *et al.*, 2006) and is presented in Figure 5.36.

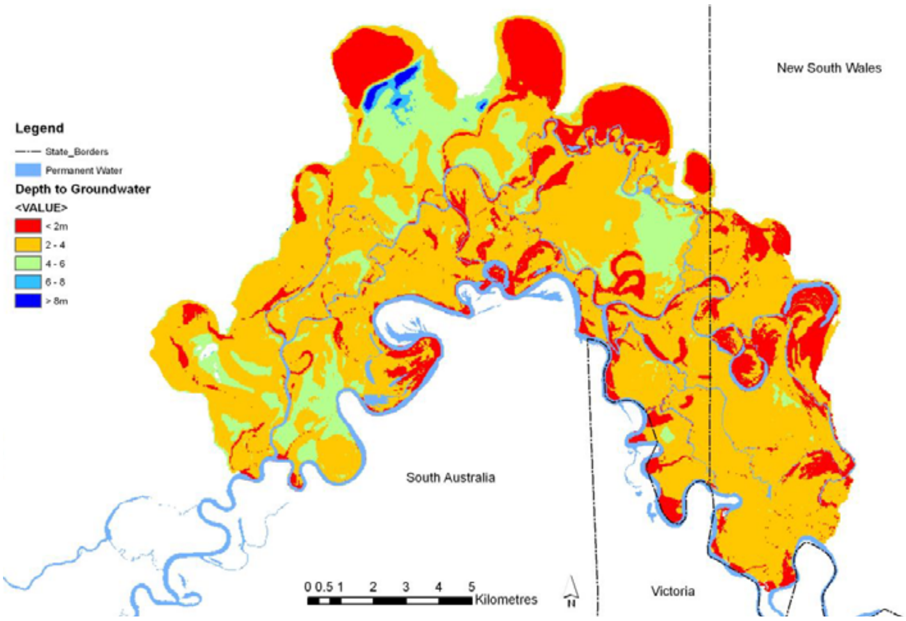

Figure 5.36 Depth to groundwater for 30 year prediction (2033) with the do-nothing scenario (MODFLOW modelling data provided by Overton *et al.,* 2006).

The WINDS model was used to predict the soil salinity in 2033 (30 year prediction) and is shown in Figure 5.37. This figure can be compared with Figure 5.35 from 2003. The tree health in 2033 for a no-intervention scenario is presented in Figure 5.38. This was based on the last 15 years of flows repeated twice. This figure can be compared with Figure 5.33 for the current day WINDS predicted tree health. The future of the Chowilla floodplain with no intervention is indicated by an increasing salt concentration in the soil profile and further decline of floodplain tree health away from the flush zone.

The prediction for a do-nothing scenario is an increase in decline of tree vegetation. It is likely that this decline will reach a limit where there will continue to exist healthy vegetation along the main channel of the river and along the loosing creeks on the floodplain.

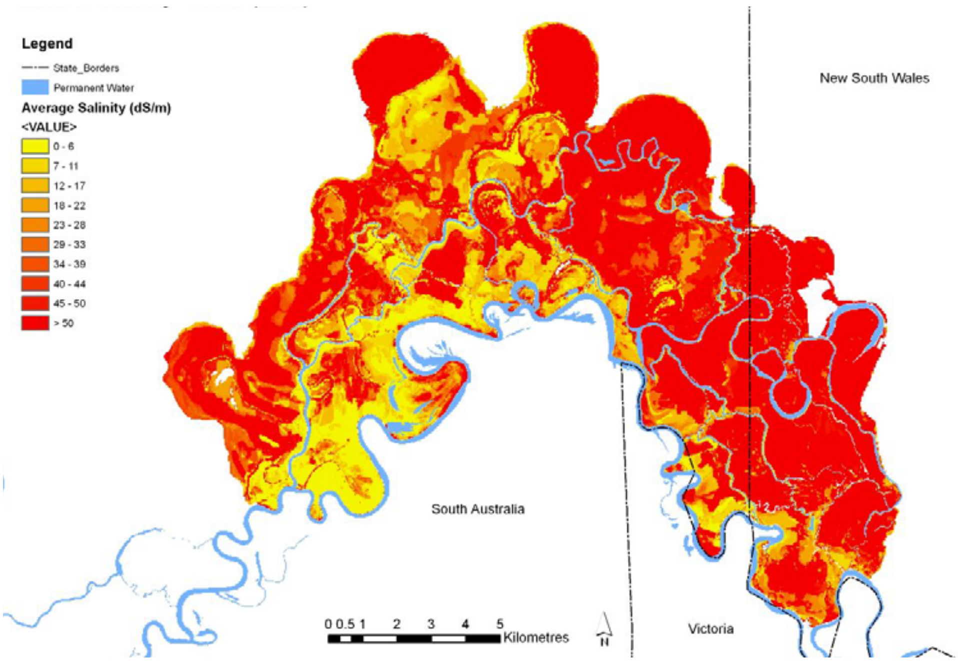

Figure 5.37 Soil salinity for 30 year prediction (2033) with the do-nothing scenario (WINDS).

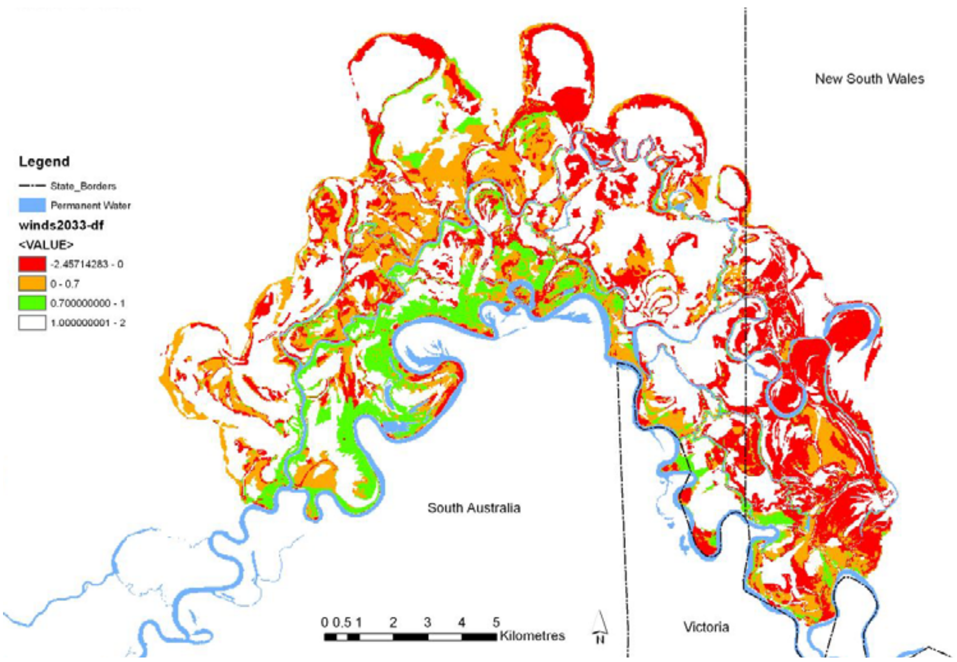

Figure 5.38 Tree health for 30 year prediction (2033) with the do-nothing scenario (WINDS), last 15 years repeated.

Groundwater Lowering Scenarios

The South Australian Department of Water, Land and Biodiversity Conservation has proposed a 38 bore salt interception scheme for the Chowilla floodplain. The scheme, which is in its concept design stage, is designed to pump out saline groundwater from 38 sites across the floodplain. Saline water pumped from the bores is then deposited into salt disposal basins away from the river. Figure 5.39 shows reduction in groundwater depth from the current depth to the 30 year depth under this groundwater interception scheme.

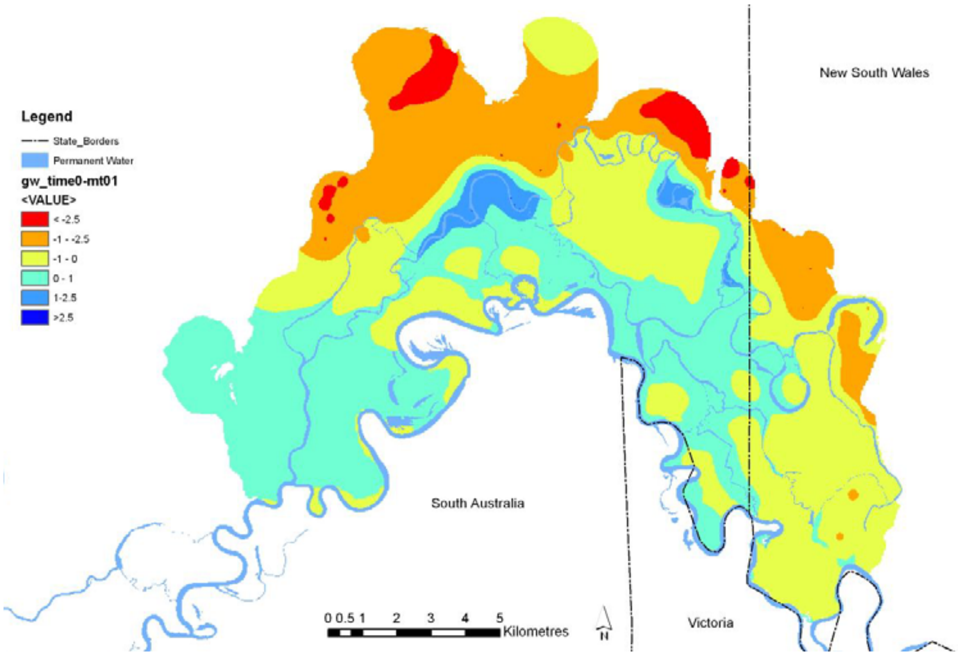

Figure 5.39 Change in groundwater depth for 30 years (2033) under the 38 bore groundwater interception scheme.

The WINDS model was used to model the soil salinity in 2033 under a groundwater interception scheme scenario (Figure 5.40) and the consequent tree health prediction (Figure 5.41).

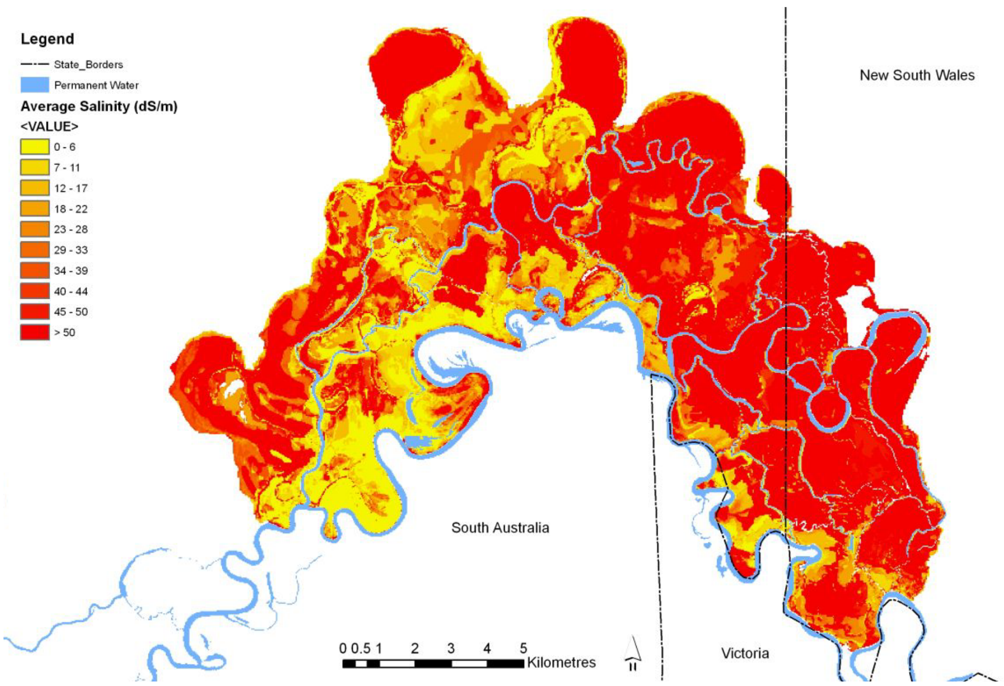

Figure 5.40 Soil salinity in 30 years (2033) under the 38 bore groundwater interception scheme.

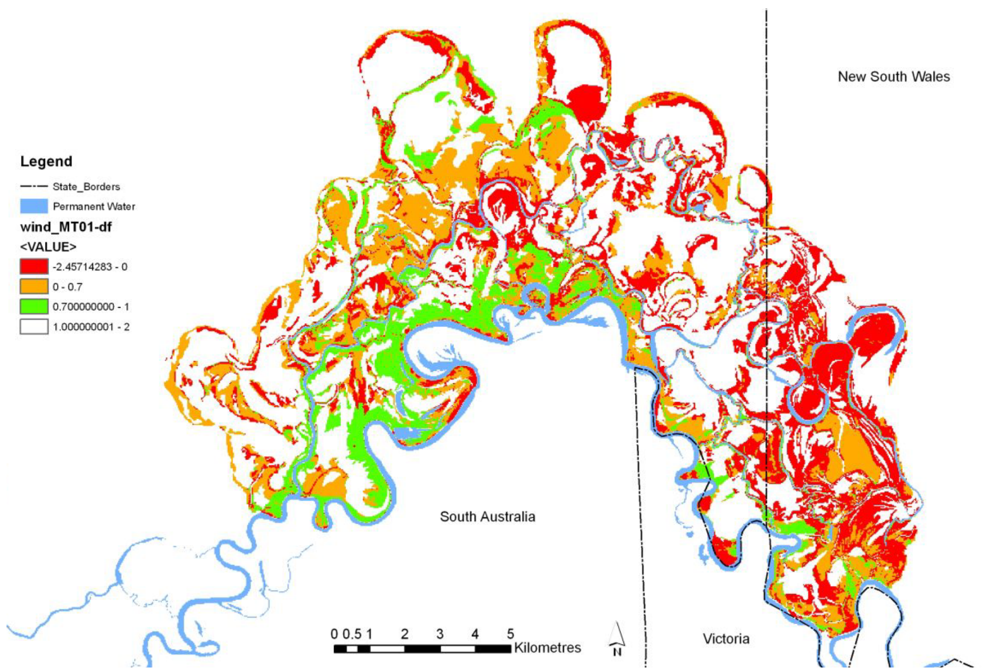

Figure 5.41 Tree health in 30 years (2033) under the 38 bore groundwater interception scheme.

The prediction for tree health under this management scenario is better than under the do-nothing scenario. However it is not as good as the current condition indicating that the groundwater lowering would not halt all of the decline. The lowering of groundwater is likely to create a buffer during drought periods as there is more soil profile for freshwater to be stored and the rate of soil salinisation will be reduced. However the immediate response of lowering groundwater is not to reduce the salt concentration in the soil which will need flooding to leach out what salt can be removed.

Another management option that could be considered to lower groundwater depth across the floodplain is to lower the pool level of Lock 6. This lowering would need to be in the order of two to three meters to have an effect on the floodplain and would have to be kept this low for a substantial proportion of the year. This option is unlikely given the problems for navigation and irrigation water supply such lowering would produce. The lowering of the weir would also dry out a number of permanent wetlands, which would be beneficial for a more natural wetting and drying cycle, but could potentially expose acid sulphate soils that could then trigger major water quality impacts (Lamontagne *et al.*, 2005).

Environmental Flow Scenarios

The MDBC, in its Living Murray program, has released hydrographs that are predicted to occur under three different release mechanisms of an extra 1,500 GL/yr, 750 GL/yr and 500 GL/yr of water each year. This volume of water would be released for environmental purposes across the River Murray and would mean approximately 50 GL/yr (for the 500 GL/yr scenario) of water for the Chowilla floodplain which could be stored and released when required. Figure 5.42 shows the modelled increase in flow from the option of 1,500 GL/yr proposed under the MDBC Living Murray program.

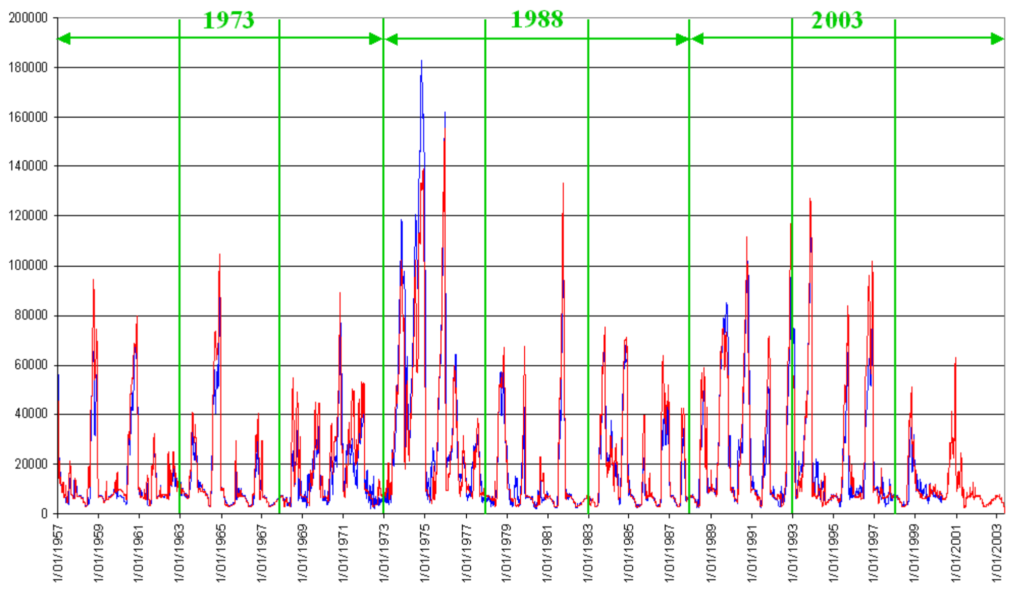

Figure 5.42 Living Murray MDBC proposed flow strategy to increase flows by 1,500 GL/yr. The hydrograph represents actual flows (blue) and enhanced flows (red).

The prediction from increased environmental flows is to maintain the same percentage of good vegetation from 2003. This would improve vegetation health in areas flooded with an increase in frequency compared to the do-nothing scenario but those areas high on the floodplain would continue to decline.

Environmental flows could also be increased from water released from storages. Lake Victoria has been used to increase the peak of a flood by 17,000 ML/day maximum, which is constrained by the stability of the outflow creek.

Weir Raising

The weir at Lock 6 can be managed to alter the upper pool level from its normal level of 19.25 m AHD up to a maximum level of 19.87 m AHD. To do this, stoplogs and Boulé panels could be added to the Lock 6 weir to gradually raise the upper pool level. The effect of raising the weir upper pool level by 620 mm (maximum) is to increase flows into the Chowilla floodplain through the numerous existing bank and weir structures. Flows within the Chowilla Creek system are also increased and creek water levels raised slightly.

The prediction under a weir raising management scenario would be to improve the vegetation only in the low lying areas. Overall the percentage of healthy vegetation declines.

This raising has impacts on creek and control structures by increasing the risk of bank erosion. The areas of inundation under different flows are shown in Table 5.9. The area of inundation impacted by weir raising of Lock 6 is presented in Figure 5.43.

Table 5.9 Area increases from the raising of Lock 6 weir by 62 cm.

Flow with MT2 management option	Area above percentage of permanent floodplain (ha)	Approx natural flood (ML/day)	Approx natural flood (ML/day)
5,000 no weir	-	-	-
5,000 with weir (19.87m)	-	-	5,000
10,000 with weir (19.87m)	-	-	5,000
20,000 with weir (19.87m)	100	-	5,000
40,000 with weir (19.87m)	1,700	10%	45,000
60,000 with weir (19.87m)	2,100	12%	60,000
150,000 (whole floodplains)	17,960	100%	

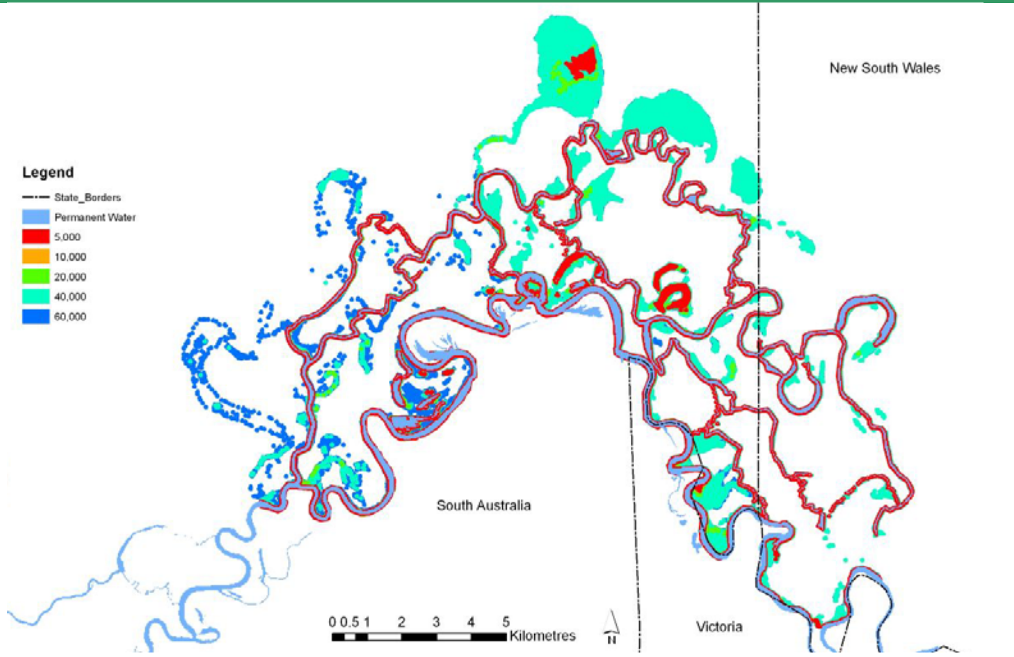

Figure 5.43 Extent of flooding for a 5,000, 10,000, 20,000, 40,000 and 60,000 ML/day flow with the Lock 6 weir raised to 19.87m and the area of influence buffered by 50 metres.

New Flow Control Structures

Flow control structures can be used to improve the health of the floodplain. As an initial assessment of potential flow management and floodplain water delivery scenarios, the feasibility of a single environmental regulator or weir at the bottom of Chowilla creek or a combination of regulators in Chowilla and Monoman Creeks were assessed. The construction of a flow control structure was proposed by the South Australian Government to enable the management of water levels throughout the Chowilla anabranch and provide the capacity to improve the health of significant areas of floodplain and wetlands.

A proposed structure at the downstream end of Chowilla Creek (Figure 5.44) as it enters the River Murray was considered for its impact on floodplain health (Overton and Doody, 2008a). The assessment included the benefits and dis-benefits from the impact on flood extent through the operation of the proposed environmental regulator, calculation of the water usage and an assessment of the potential ecological benefits. The modelling of salinity impacts and dis-benefits was undertaken by the South Australian Government Department of Water, Land and Biodiversity Conservation (Yan *et al.*, 2005).

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held by the University of Adelaide Library.

Figure 5.44 Chowilla Creek environmental regulator proposal location of the weir and the blocking banks (based on designs by URS Pty Ltd (Overton *et al.*, 2006).

The pseudo-hydrodynamic model described in Chapter 3 was used to identify the extent of inundation from this new regulator. Figure 5.45 shows the extent of inundation from five different flows in the River Murray with a Lock 6 weir height of 19.87 metres AHD. Table 5.10 shows the areas of flood extent and the approximate natural flood size. The extent of inundation from a flow of 5,000 ML/day can be increased from the current extent (permanent water) to approximately 4,900 hectares using an environmental regulator height of 19.87 metres AHD. This is the area equivalent to approximately a 65,000 ML/day flood event, which is approximately a 1 in 6 year event. The environmental regulator at 19.87 metres would be totally inundated at flows over 60,000 ML/day.

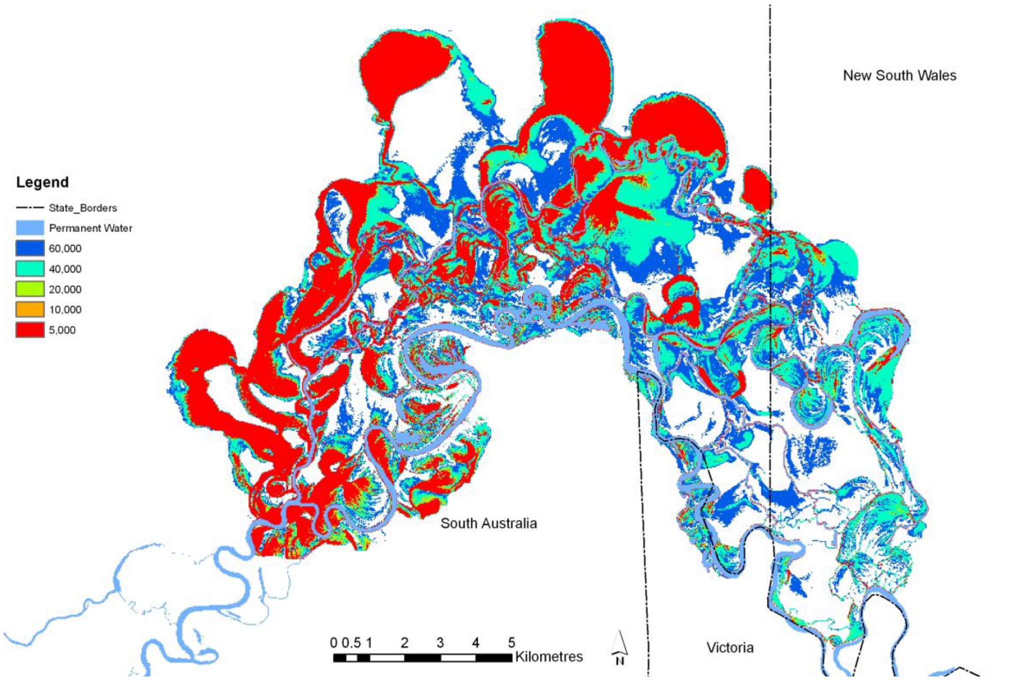

Figure 5.45 Extent of flooding for a range of flows with the proposed new environmental regulator in Chowilla creek.

Table 5.10 Area of inundation of different flows in the river using the proposed environmental regulator.

Flow with MTA environmental regulator	Area above permanent (ha)	Percentage of floodplain	Approx natural flood (ML/day)
5,000 no environmental regulator	-	-	-
5,000 with environmental regulator (19.87)	4,900	27%	65,000
10,000 with environmental regulator (19.87)	5,000	28%	65,000
20,000 with environmental regulator (19.87)	5,400	30%	65,000
40,000 with environmental regulator (19.87)	8,800	49%	82,000
60,000 with environmental regulator (19.87)	12,400	69%	95,000
150,000 (whole floodplain)	17,960	100%	

The modelling of water requirements and environmental benefits have been based on operation of the environmental regulator every year for flows under 60,000 ML/day. In some cases, this provides areas of inundation at a frequency greater than the natural return period. This may be necessary in the beginning to leach out the accumulated salt stored in the floodplain soils. A higher than natural return period would also be necessary to combat shallow groundwater tables in halting the rate of salt accumulation. Over many years the frequency of using the environmental regulator is expected to be reduced to a more natural flood frequency.

Figure 5.46 shows the hydrograph of 1957 to 2003 from the River Murray at Chowilla. The current flow regime is compared to the natural and a flow enhancement option of 500 GL/yr modelled by the MDBC. The graph shows the effective hydrograph achieved by using the environmental regulator.

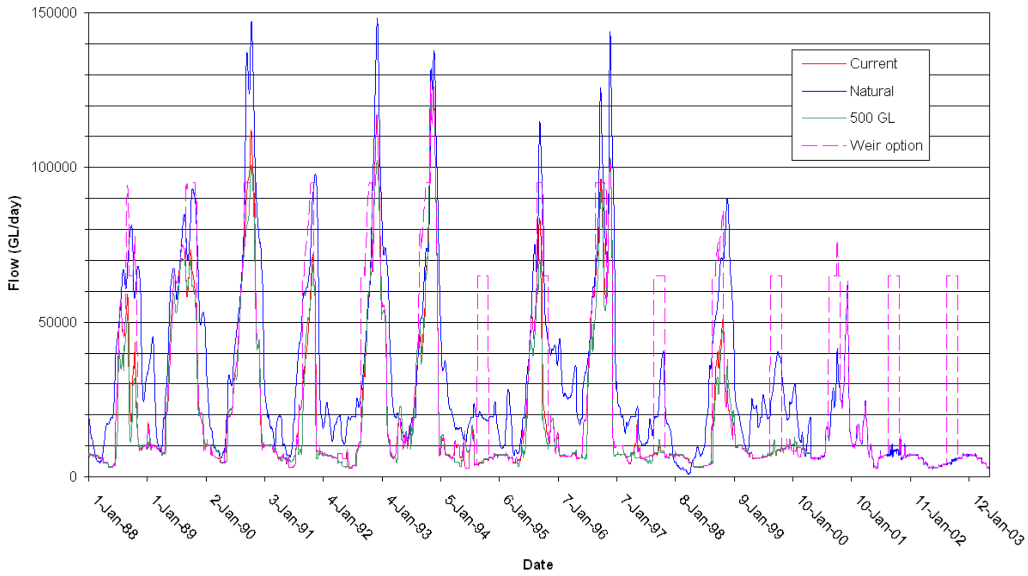

Figure 5.46 Hydrograph of flows into South Australia (through Chowilla) from 1988 to 2003 showing the current flow, the modelled natural flow and a modelled flow enhancement of 500 GL/yr. The dotted line shows the effect of operating the environmental regulator each year on the current hydrograph.

Figure 5.47 shows the percentage of time inundated during the 15 years of flows shown in Figure 5.46. The effect of the environmental regulator option is to return the floodplain to a near-natural flooding frequency. The floodplain surface inundates when creek levels reach over-bank flow at approximately 35,000 ML/day. The area inundated by the environmental regulator will not be exactly the same as a natural flood of similar size. For example, a flow of 5,000 ML/day with an environmental regulator held at a height of 19.87 metres, will inundate the same total area of approximately 4,900 hectares as a 65,000 ML/day flood with no environmental regulator. However the flooded area will be different with more flooding in the west (e.g. Coombal swamp) and less in the east (overbanking Salt Creek etc), as the main force of the water is back from the environmental regulator rather than upstream from a higher flow.

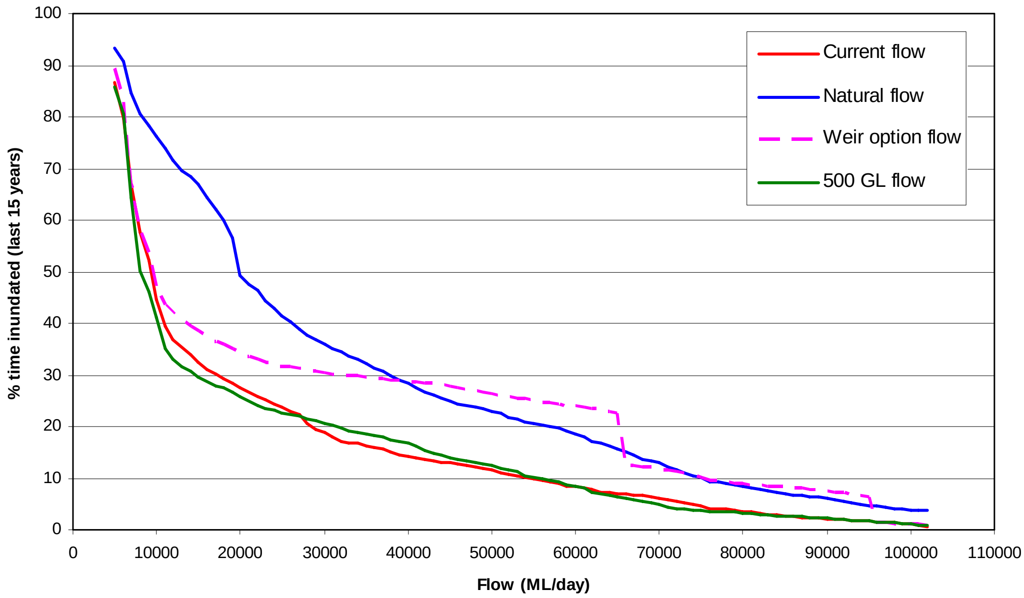

Figure 5.47 Percentage of time inundated during the last 15 years (1998 – 2003) versus the size of the flow across the border into South Australia.

An estimation of the water required to operate the environmental regulator at 19.87 metres is given below. The water loss would occur through evaporation and seepage into the soil. Wetland watering trials at Chowilla have suggested that the loss of flood water to evaporation and infiltration is approximately 23 % over three months. This figure was calculated for the water lost during the flooding of an anabranch creek on Monoman Island after three months inundation. This figure would vary depending on the duration of inundation, the depth of water, the antecedent soil moisture conditions, inundation of recharge versus non-recharge soils, as well as other factors such as temperature and relative humidity.

Given a flow of 5,000 ML/day and an environmental regulator height of 19.87 metres, an area of 4,900 hectares would be inundated with a volume of standing water of 23,750 ML. This would take approximately 14 days to fill, given the flow into Chowilla. Using a loss method based on the 23% as seen in the watering trials with this environmental regulator configuration the loss would be 5,450 ML and the total water used would be 29,200 ML. Loss from the watering trial will be an underestimate as the clay surface of the bottom of the wetland will allow less seepage than the sandy soils of the floodplain when considering the large area of inundation from the proposed environmental regulator. Also the surface area to depth ratio for a large floodplain inundation will be much higher than the small wetland trial.

The WINDS model was used to predict tree health on the floodplain from a 30 year scenario of using the proposed environmental regulator every year (Figure 5.48).

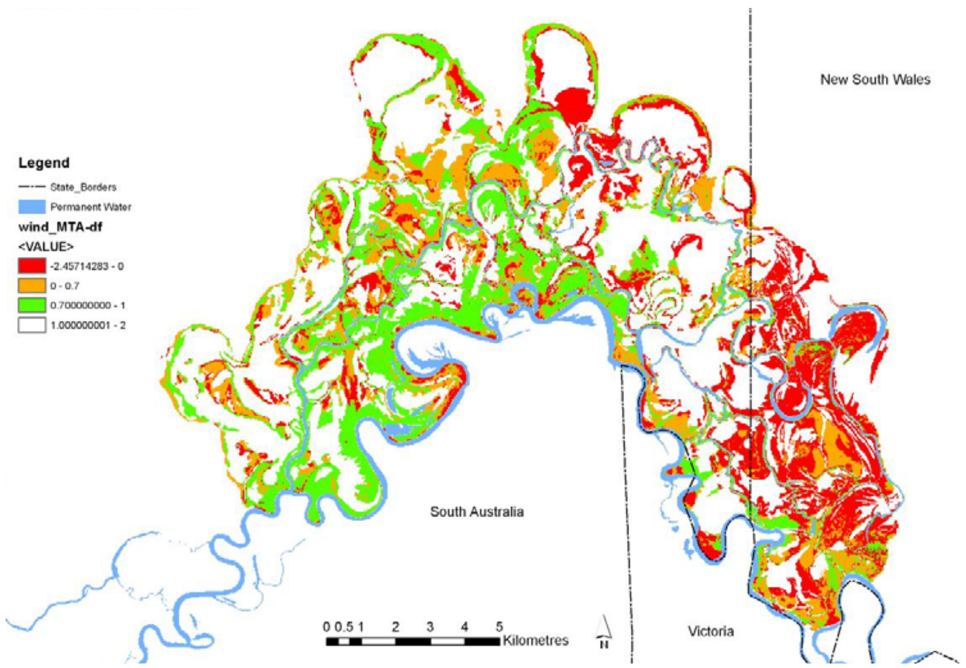

Figure 5.48 Tree health predictions for 30 years (2033) using the proposed environmental regulator in Chowilla creek (WINDS).

The areas that could be affected by the proposed environmental regulator operation were inundated for different times and cover areas of the floodplain with different soils and depth to groundwater. The

WINDS model was used to determine which parts of the areas affected by the proposed environmental regulator could show a response with improved health. Figure 5.49 shows the area inundated divided into parts that showed a vegetation health response and those that did not. Only the area of trees that were poor or dead in 2003 was considered for the identification of change. This was because a healthy tree in 2003 could not increase the WINDS index and would therefore remain as no change, which would be confusing. Not all poor or dead tree vegetation that received extra water from the environmental regulator shows a predicted improvement in health. Those areas that do not show an improvement in health occur in the eastern part of the floodplain where shallower groundwater creates greater salt accumulation rates. These areas are likely to need groundwater management via salt interception schemes as well as the increased inundation.

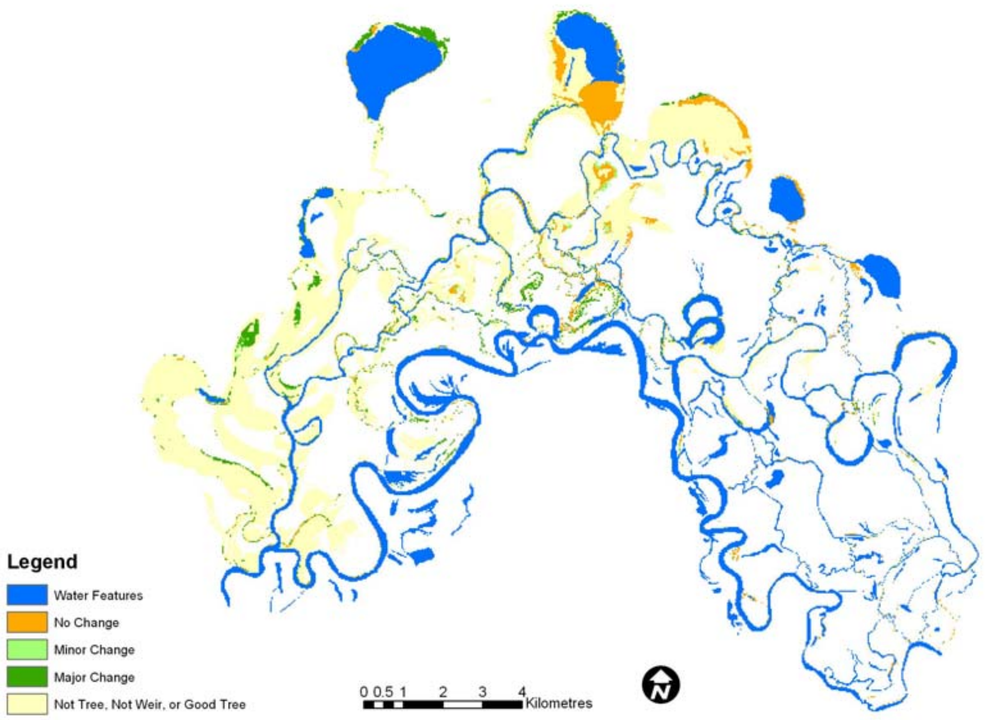

Figure 5.49 Change in tree health prediction for 2033 from 'do nothing' to the proposed environmental regulator operation on the Chowilla floodplain from WINDS modelling. Operation is based on the weir being used every year ay 19.87m with the last 15 years of flow repeated. The figure only shows those areas affected by the weir that are trees and were not in good health in 2003.

Approximately 45% of the poor or dead trees that were inundated showed a minor or major improvement in predicted health (Table 5.11). An increase of 0.2 in the WINDS index can be considered as approximately a 20% increase in its health and will take all the dead class to poor class and some of the poor into good. The major change of 0.2 to 1 sees some areas classified as dead, change to a classification of good health. The WINDS model predicts areas that can support adult trees. If the trees in an area die and conditions change to later support trees then the model will predict healthy trees. These areas may recruit naturally or require planting.

The change in the soil salinity in the areas affected by the environmental regulator is presented in Figure 5.50. Some areas have decreased their soil salinity by 40 – 50 dS/m, almost completely freshening the soil profile. A large area in the west has had its soil salinity reduced by approximately 20 dS/m. This area does not support many trees at present but the 20 dS/m drop could prove beneficial to the understorey vegetation.

Table 5.11 Tree vegetation on the Chowilla floodplain and the change in WINDS index.

Tree vegetation poor or dead	Area (ha)	Percentage of area affected
Change in WINDS index		
No change	409	55%
Minor change (0-0.2)	31	4%
Major change (0.2-1)	300	41%
Total	740	

From the history of salt loads and flow in Chowilla, and from the salt load modelling of DWLBC, it is predicted that the proposed environmental regulator at 19.87 metres will create a salt load of approximately 60,000 tonnes during the following 12 months. This amount is dependent on the creek levels during the following 12 months with lower than expected salt loads occurring when river levels remain high. The speed at which the water levels in the creeks are lowered from 19.87 metres to the low flow levels of approximately 16.8 metres will determine the rate at which the salt load will enter the creeks and then the River Murray. There is a possibility of using the draw down of the environmental regulator to aid salinity management on the floodplain.

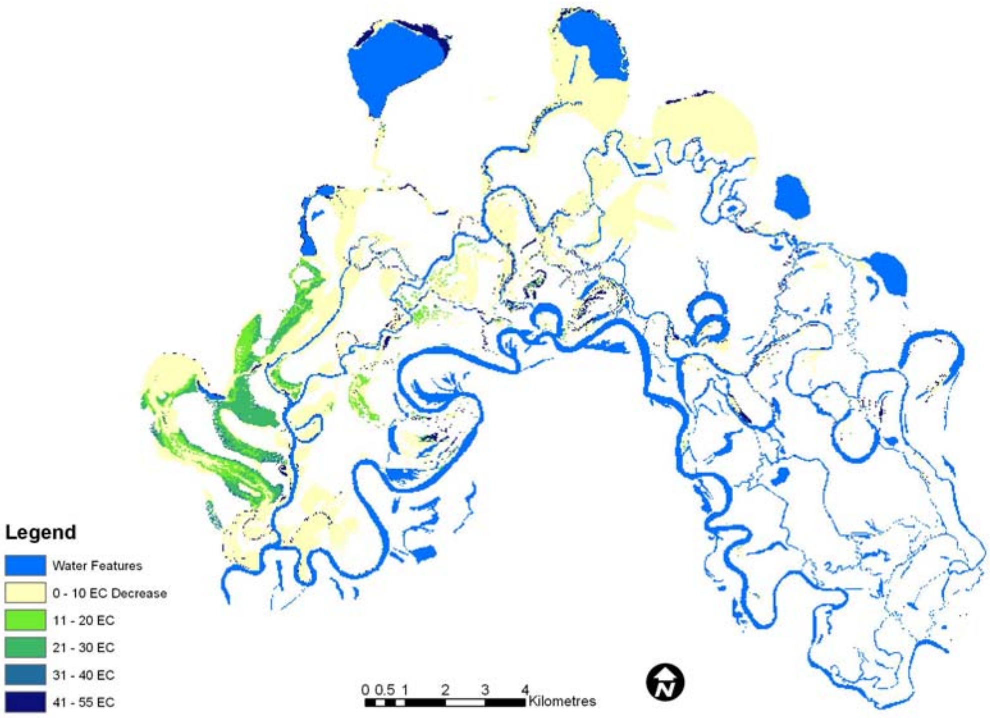

Figure 5.50 Soil salinity decrease from proposed environmental regulator operation on the Chowilla floodplain from WINDS modelling at 2033 with last 15 years flow repeated. Units are in dS/m EC. Operation is based on the weir being used every year ay 19.87m with the last 15 years of flow repeated. The figure only shows those areas affected by the weir that are trees and were not in good health in 2003.

As the major component of salt load is increased groundwater pressure, not the leaching and wash-off, the salt loads for the environmental regulator operation of approximately 60,000 tonnes over the following 12 months, is unlikely to reduce very much over time with successive Chowilla Creek proposed environmental regulator operations, unless a freshening of the groundwater occurs.

The benefits, and dis-benefits discussed here relate to a specific environmental regulator operation of an elevation 19.87 metres. The environmental regulator could potentially be raised to 20 metres by raising the Lock 6 weir to this level as well. The top of the Lock 6 weir is at an elevation of 19.87 metres but an assessment has been made that indicates it could be raised. An extra 13 centimetres of water height could impact on the extent of inundation and therefore the benefits / dis-benefits of the environmental regulator. Environmental benefit of the environmental regulator was also modelled using a flow to South Australia of 5,000 ML/day. Should the flow be greater than this, then the extent of inundation will be greater. Higher flows also mean the environmental regulator may not need to be used every year. The proposed salt interception scheme (38 bore scheme discussed above) is able to mitigate most of the salt increase from the environmental regulator operation.

The environmental benefits of the proposed Chowilla environmental regulator in terms of vegetation health and wetland inundation can be summarised as:

- 27% of the floodplain returned to a natural flow regime (influences 32% with water supply to creeks and wetlands);
- Ability to flood more frequently a larger area than under natural conditions to reverse salt accumulation;
- Uses approximately 70 GL/yr, takes 14 days to fill and can be held for approximately 3 months (5,000 ML/day flow at 19.87m);
- Creates a medium size flood from entitlement flows only, making it not dependent on increased flows in the River;
- Increases a 60,000 ML/day flood from 27% to 68% of the floodplain;
- Approximately 370,000 tonnes of salt removed from soil profile in 30 years (from approx 4,000,000 tonnes in the floodplain);
- The environmental regulator at 19.87 at 5,000 ML/day would inundate:
  - 37% of the river red gum Forest (influence 63%);
  - 24% of the river red gum / black box (influence 34%);
  - 57% of the cooba;
  - 46 % of the lignum;
  - 46% of the samphire;

- 85% of the grasslands;
- 24% of the high and medium biodiversity value areas; and
- 23% of the target area.

Artificial Irrigation

The construction of channels on the Chowilla floodplain can provide increased flooding frequency to some areas. Potential sites were identified by the Department of Environment and Heritage and included Pilby Creek to Chowilla Horseshoe, Pipeclay Billabong (near Pipeclay Creek weir) and Slaneys Billabong (near Slaneys Creek weir).

A management option of pumping water into the six larger wetland areas was investigated. The extents, volumes and pumping rates were determined for each wetland (Overton *et al.*, 2006). The wetlands included Coppermine waterhole, Werta Wert, Coombool swamp, Lake Limbra, Gum Flat and Lake Littra.

Watering of wetlands and floodplain areas by pumping, blocking and irrigation can achieve flooding regime thresholds but does not replace natural flood events. Natural floods connect these wetlands to the main channel for nutrient, fish larvae and seedling connectivity.

Management combinations

A number of the management scenarios described above can be considered as complementary. Groundwater lowering would be considered critical, operating in conjunction with the proposed environmental regulator to firstly combat the salt loads generated from the extra flooding and secondly to reduce salt accumulation rates and provide greater airspace within the soil profile.

A combination of weir raising and groundwater interception, a combination of groundwater interception and the proposed environmental regulator, and a combination of weir raising plus the channelling and wetland pumping scenarios were all considered. Results of these combinations are presented in table 5.8.

5.4.3 Management Considerations

Management scenarios for floodplain vegetation health can be categorised in relation to their impact on vegetation water availability. It is useful to consider the different sources of water for riparian vegetation. Table 5.12 reviews the main plant water sources and the management scenarios that could affect these.

Table 5.12 Riparian vegetation water sources and the potential impacts from management scenarios.

Source	Influenced by	Impacts on Vegetation Health	Management Scenarios
Soil water storage	Flooding	- Supply of freshwater to reduce matric potential (drought) - Leach out salt to reduce osmotic potential (salt accumulation)	- Increase flooding (recharge / leaching) - Increase flows - Weir raising - Manage structures
	Rainfall	- Supply of freshwater to reduce matric potential (drought) - Leach out salt to reduce osmotic potential (salt accumulation)	
	Salt accumulation	- Lower soil salinity to reduce osmotic potential - Lower groundwater to create airspace and reduce salt accumulation rates	- Lowering groundwater levels by lowering river regulation structures (weir lowering) - Lowering groundwater levels by groundwater interception pumping
Groundwater in the capillary fringe	Root zone / capillary fringe	- Fresher water extracted from top of groundwater	- Groundwater lowering (increase airspace)
	Freshwater lens	- Replenish freshwater lens in recharge areas	- Flooding (leaching) - Groundwater lowering (increase space) - Freshwater injection
	Recharge from creek edge	- Fresher water extracted from unsaturated water at the edge of the creek	- Weir raising to create higher river and anabranch levels - Increase small and medium floods (within channel flows) - Manage structures

	Flush zone around Lock 6	- Supply of fresher groundwater	- Weir raising to create higher river levels - Increase small and medium floods (within channel flows)

The major findings of the WAVES modelling were that:

- There is no benefit in lowering the water table to more than 5 metres below ground as groundwater discharge is close to zero at this depth;
- For sites with the water table less than 3 metres, water table lowering may provide slightly better benefits than the increased flooding from the MDBC Living Murray 1,500 GL/yr flooding scenario;
- For sites with the water table 3 to 4.5 metres deep, the relative benefits will be variable depending on soil type, groundwater salinity and elevation;
- For sites with water tables greater than 4.5 metres none of the scenarios will be of any great benefit;
- The greatest benefit came from the combination of water table lowering to 5 metres and increased flooding using the 1,500 GL/yr flooding scenario; and
- There is less risk in a management strategy of lowering groundwater as this method provides a longer time period where lack of floods will impact on the increasing soil salinisation as salinisation rate is a function of groundwater depth. In some cases increasing the flooding frequency will leach salt and change the balance of soil salinisation but success is dependent on the frequency of these floods. If a dry period occurs (such as the last five years) the decline in vegetation will be dramatic as seen in the major river red gum dieback in 2003 (MDBC, 2005).

Comparing management options simply on overall percentage of area of good health trees is limited and may not achieve the best outcome overall. An approach to assessing the impact of management scenarios on the Chowilla floodplain is to identify management units that could be separated on their environmental water requirements on the basis of their location in the landscape and the ability to supply water to these areas.

Figure 5.51 identifies biodiversity conservation areas based on an assessment by the South Australian Department of Environment and Heritage and the area of inundation from the operation of the proposed environmental regulator at 19.87 metres AHD with 5,000 ML/day in the river. Table 5.13 shows the areas of the different biodiversity conservation units, the area that would be inundated from the proposed environmental regulator operation and the percentage of that biodiversity unit inundated.

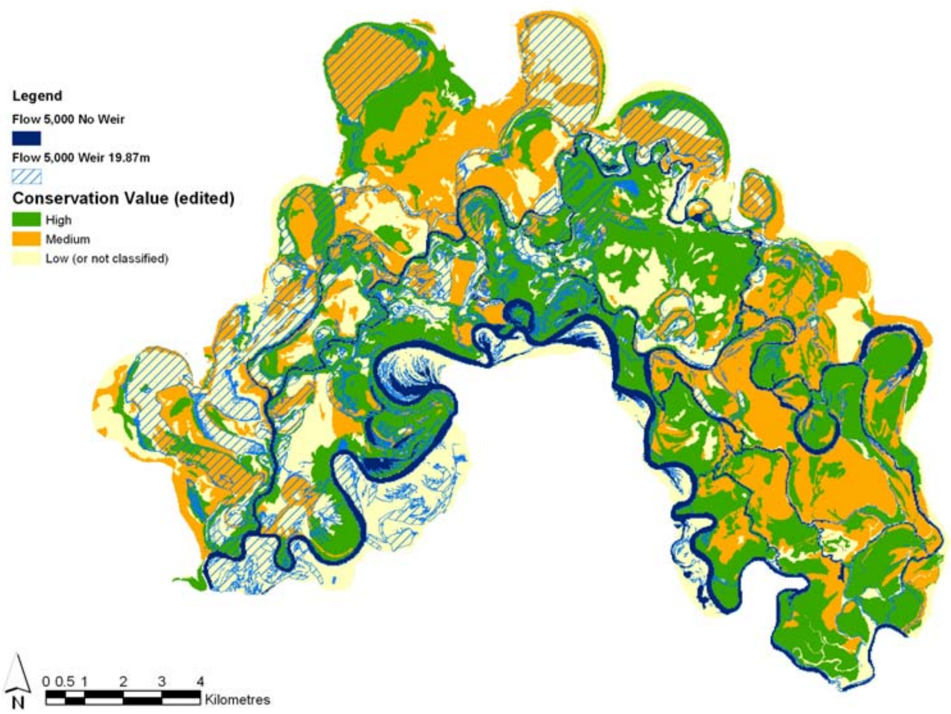

Figure 5.51 Biodiversity conservation areas on the Chowilla floodplain.

Table 5.13 shows that the management option of raising the weir improves the low conservation value areas to a greater extent than the high conservation areas.

Table 5.13 Biodiversity conservation values on the Chowilla floodplain and the area inundated by the proposed environmental regulator.

Conservation value	Area (ha)	Percentage	Inundated flow 5,000 weir 19.87m	Inundated percentage flooded with weir
High	7,183	42%	1,427	20%
Medium	5,875	34%	1,576	27%
Low	4,182	24%	1,590	38%
Total	17,239	100%	4,593	27% (average)

The Department of Environment and Heritage used the biodiversity conservation areas and combined other management target areas such as threatened species and areas of good condition. This created a target management map against which management scenarios could be tested. Figure 5.52 indicates the target management units and the area of inundation from the operation of the proposed environmental regulator at 19.87 metres AHD with 5,000 ML/day in the river.

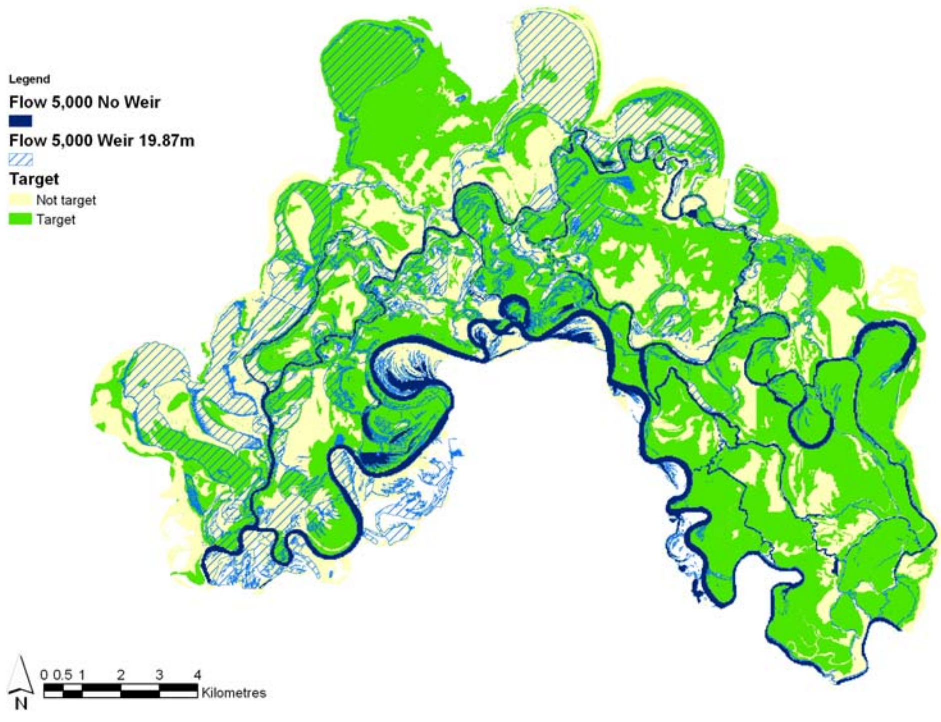

Figure 5.52 Target management areas on the Chowilla floodplain with areas inundated by a raised weir.

Another approach to assessing management options is to identify goals for lowering groundwater and increasing flooding and then compare management options by the degree to which they meet these environmental targets. Groundwater lowering has been shown to be a useful management option to control soil salinity and improve vegetation health. Figure 5.53 shows the draw down depth required to reach the critical groundwater depth of 5 metres to halt salt accumulation. Lowering groundwater reduces the rate of soil salinisation and benefits will be achieved in areas where the water table is drawn down to that depth. Reduction in soil salinisation rate does not improve vegetation, only slows its rate of decline. Areas that can be reduced in their rate of soil salinisation can be considered as 'flow or irrigation ready' and their health will improve with flooding or targeted irrigation.

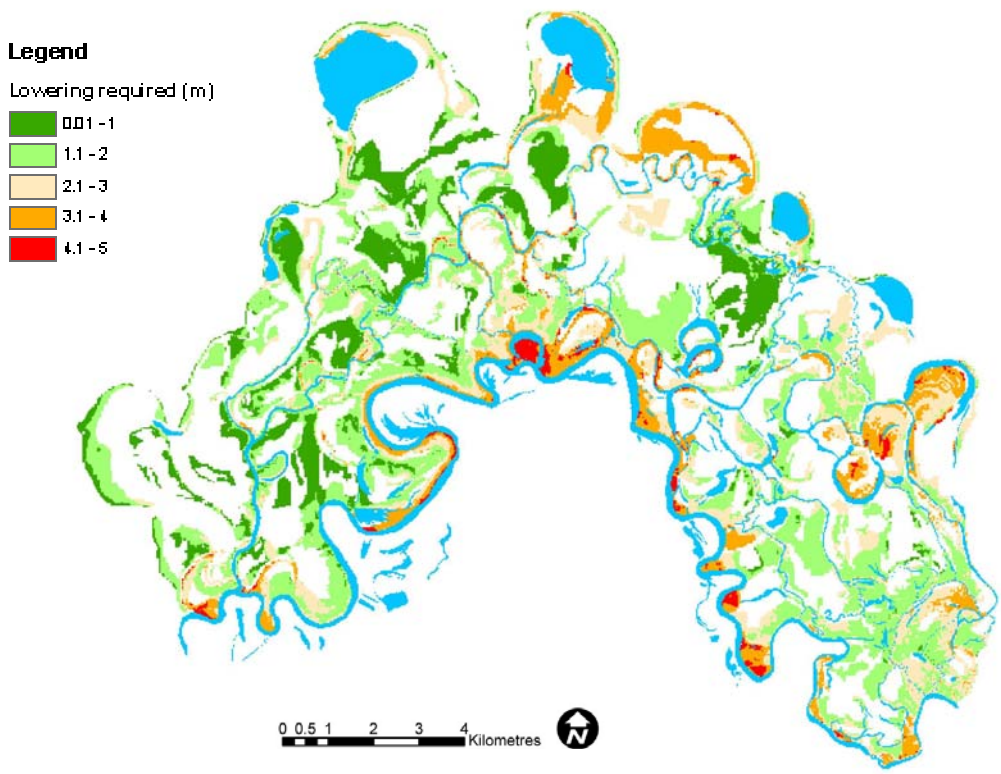

Figure 5.53 Draw down required to create a groundwater table of 5 metres.

The other main management option of increased flooding frequency will improve vegetation health, without the need for lowering groundwater, by changing the balance of leaching to discharge. The number of days required to flood an area in order to balance salt accumulation is shown in Figure 5.54. Increasing flooding without groundwater lowering is risky as it is reliant on the frequency of floods, these areas will quickly decline if a dry period occurs. The limitation of improved flow regimes as a management option, is that elevated floodplain areas require substantial amounts of water to inundate, which may never, or only very rarely, be available. Elevated floodplain areas which are salinised could be considered as sacrificial areas.

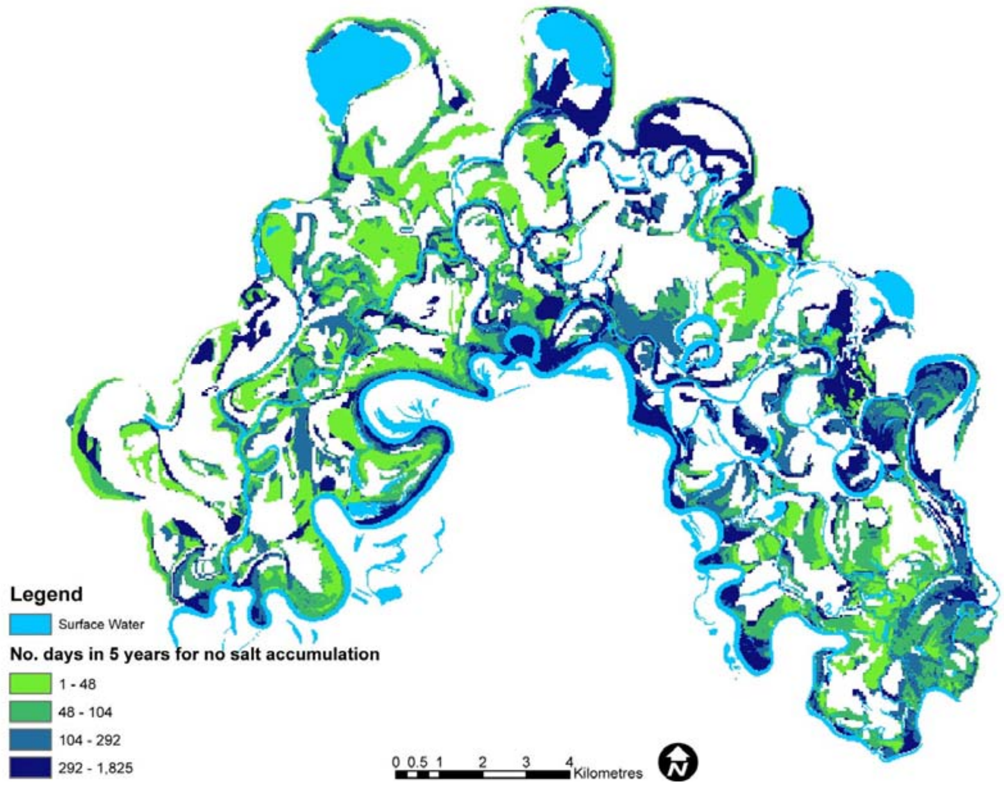

Figure 5.54 Number of flood inundated days during a five year period to halt salt accumulation.

The effect of water table lowering reduces the rate of soil salinisation but does not remove salt. Flow enhancement reduces salt but is susceptible to degradation should a dry period occur, as it has not addressed the rate of salinisation. The greatest benefit comes from a combination of the two scenarios. A management combination of lowering groundwater and increasing flooding frequency is likely to be required to conserve the majority of the Chowilla floodplain. Lowering groundwater is considered a safety option during dry periods but only makes the floodplain 'flow ready' to be improved with a flood.

The WINDS results are useful for prioritising different parts of the floodplain for management and combining the factors of:

- Current health;
- Potential risk of salinisation; and
- Capacity to respond to viable economic management changes.

These factors can be added to conservation significance to prepare a management priority plan for location planning of hydrological engineering works and salt interception schemes (Figure 5.55). Figure 5.55 is an example of how the WINDS modelling can be used to identify areas that can respond to groundwater lowering and improved flows within defined economic constraints.

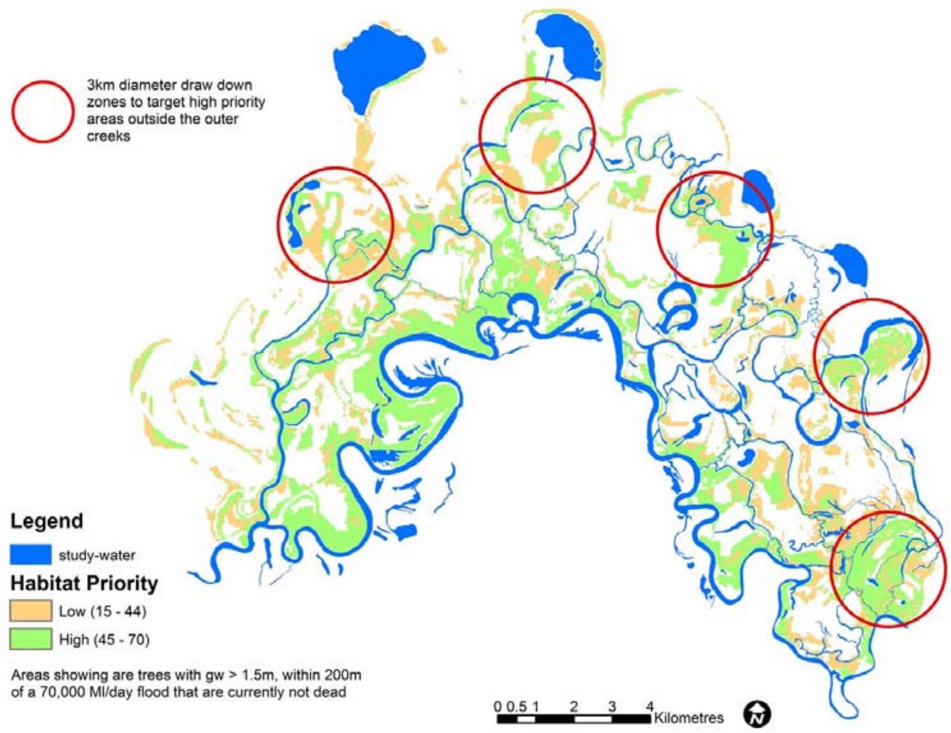

Figure 5.55 An example of the use of the DEH biodiversity rating in combination with the environmental risk modelling to prioritise areas for rehabilitation.

Further analysis of the model results can allow the identification of target areas for management scenarios. The areas of poor vegetation health on the floodplain can be categorised on the basis of groundwater pumping scenarios and flow enhancement scenarios on the basis of:

- Vegetation in areas that will respond well to economically viable groundwater pumping to reduce or halt salt accumulation processes. These are areas where a two metre draw down will benefit vegetation/habitat/environmental values the most;
- Areas that would require considerable flooding to reduce the salinity risk. These areas are those that can be made 'flow ready' with a draw down, but which are mostly dependent upon flow for significant benefit;
- Areas with deep groundwater that are in poor health due to a lack of flooding, which could be enhanced with environmental flow management; and
- Areas that would require groundwater lowering that would not be viable and considerable extra flooding. These areas will be difficult to restore, given their elevation and therefore flow opportunities. These areas may become candidates for sacrificial sites.

These areas can be depicted spatially and compared to conservation and social priority areas for management. Table 5.14 outlines the basis for the division of these areas. Modelling of the extent of inundation created by the proposed environmental regulator operation has been previously assessed by the percentage of floodplain covered. The effect on creeks and wetlands is greater, as these flow channels would then be connected even at low flows. The Chowilla floodplain has a series of ephemeral creeks and wetlands that provide channel flow, habitats for fish and aquatic vegetation, and supply freshwater to the fringing vegetation such as river red gum and black box communities.

Figure 5.56 shows the extent of the impact of the proposed environmental regulator on the ephemeral creeks and wetlands. Of the

2,016 hectares of ephemeral creeks and wetlands, 83% (1,676 hectares) could be inundated by the operation of the proposed environmental regulator under entitlement flow. This is based on the area of inundation and not on individual identified creeks or wetland bodies. Smaller control structures on these creeks and wetlands could be employed to control filling and drying cycles to best meet management objectives. The main effect of the proposed environmental regulator will be to allow the management of individual creeks and wetlands at entitlement flows. Flow control structures can have a detrimental effect on water quality. Fish passage is also of concern with flow control structures and fish passage is a critical factor in the design on any new control structure.

Table 5.14 Areas of the floodplain with poor vegetation health can be categorised on management scenarios.

	Flow enhancement possible (< 50,000 ML/day)	**Some flow enhancement possible (50,000 – 80,000 ML/day)**	**Limited flow enhancement possible (> 80,000 ML/day)**
Groundwater currently below 5 metres (> 5 metres)	environmental flows	environmental flows	Sacrificial
Groundwater lowering to significant (5m) depth is viable (2m) (3 – 5 metres)	Draw down (high priority) or Draw down (high) with environmental flows	Draw down (high) with environmental flows or Draw down (low) – Flow Ready	Draw down (low)-Flow ready
Some groundwater lowering can be achieved (1.5 to 3 metres)	Draw down (high) or environmental flows or Draw down (high) with environmental flows	Draw down (low) with environmental flows	Draw down (low) – Flow ready
Groundwater too shallow for viable draw down (0 – 1.5 metres)	Sacrificial or Draw down (low) with environmental flows	Sacrificial	Sacrificial

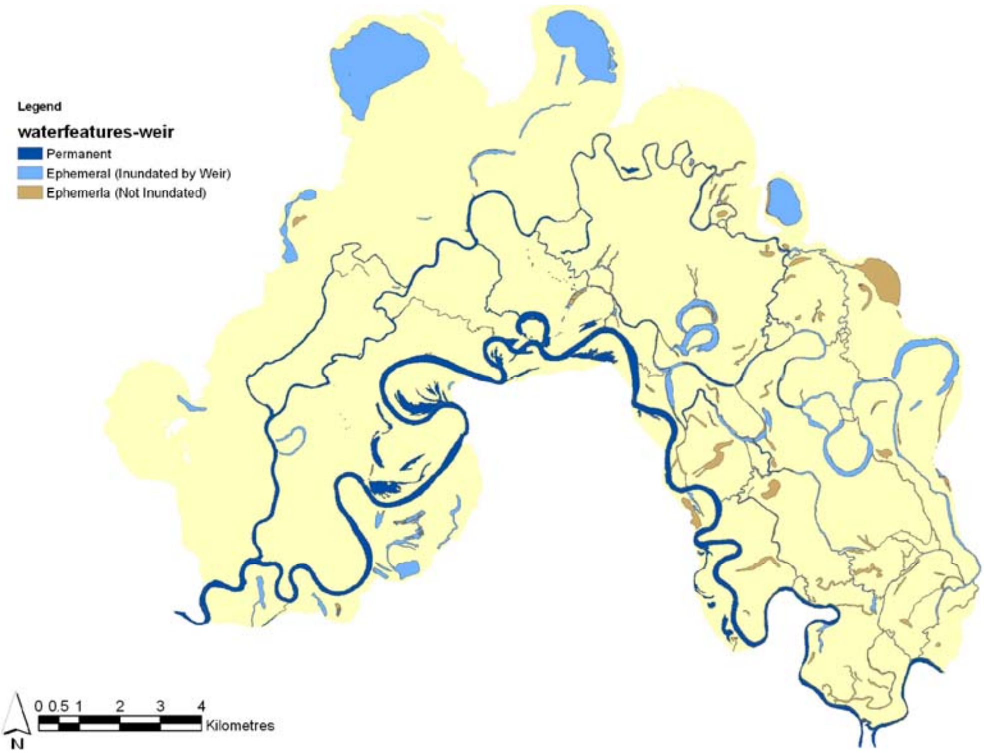

Figure 5.56 Impact on water features of the Chowilla floodplain from the operation of the proposed environmental regulator.

Tree health is relatively easy to observe in the field and to monitor using remote sensing. It is harder to assess the health condition of vegetation other than trees and therefore evaluate impacts of management options. The understorey vegetation was compared to modelled soil salinity from the WINDS model. Figures 5.57 to 5.59 show the distribution of a range of vegetation types on the Chowilla floodplain against modelled soil salinity. It can be estimated from these graphs that the floodplain previously had a natural salinity range of approximately 0 to 30 dS/m as defined by the range of natural occurrence of river red gums and salt bush (Figure 5.57).

In some areas, vegetation is being affected by an increase in soil salinity and species now occur in areas beyond their natural tolerances. Black box, for example, occur in areas of 34-42 dS/m and in these areas the health is poor (Figure 5.58). Other floodplain areas have species which tolerate higher salinities than 30 dS/m and can be considered as potentially invasive species (Figure 5.59).

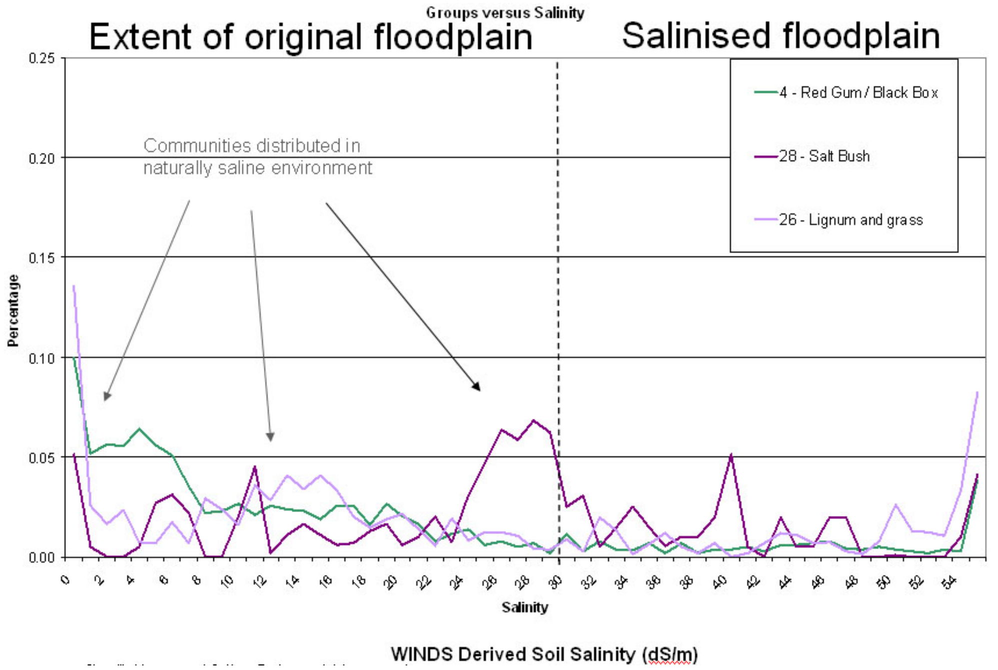

Figure 5.57 Soil salinity ranges for different vegetation types showing the presence of communities spread over a 'natural' salinity range likely to have occurred prior to river management.

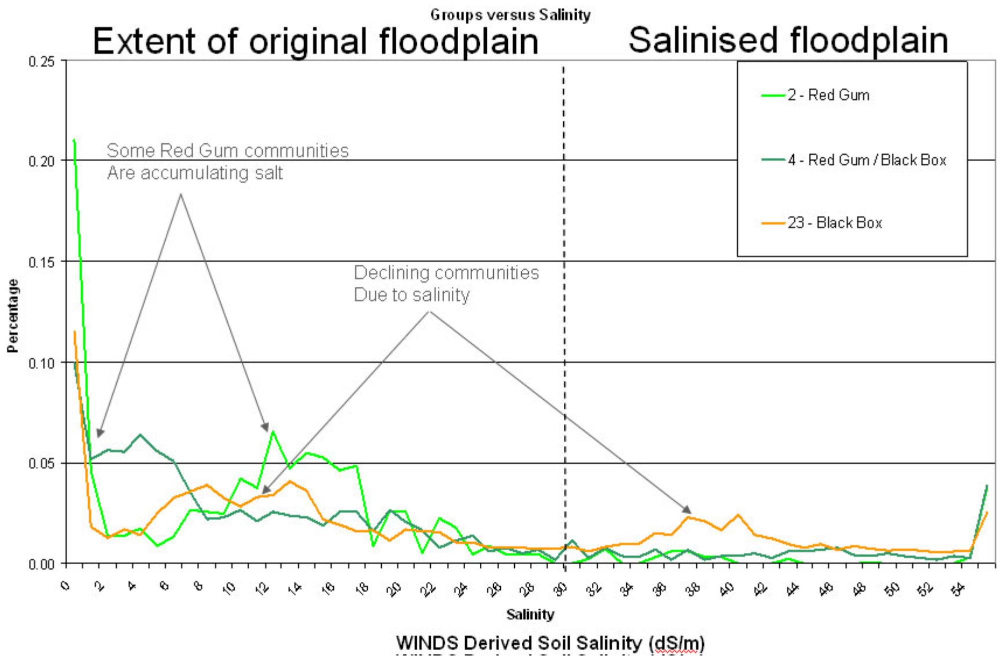

Figure 5.58 Soil salinity ranges for different vegetation types showing the presence of communities likely to be occurring in areas that are salinising.

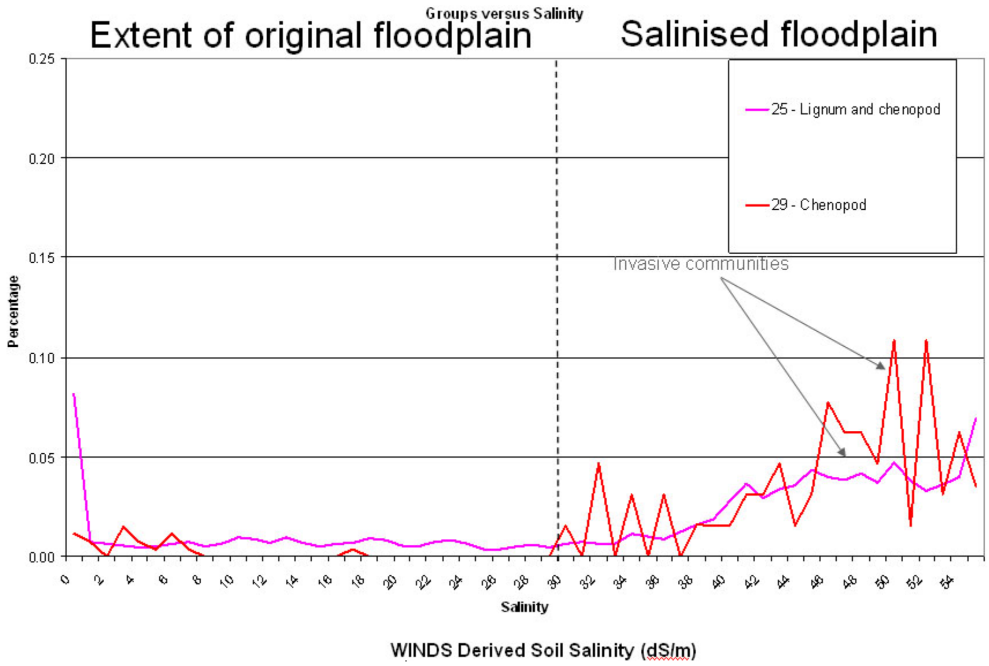

Figure 5.59 Soil salinity ranges for different vegetation types showing the presence of communities occurring in highly saline areas and therefore suggesting invasive species to the floodplain.

It is inevitable that the floodplain will change in its extent and spatial patterns as the volume of the water decreases, due to both climate change predictions and competing water users. As the volume of water decreases the frequency of flooding for any given flood magnitude will decrease. This has been discussed earlier as the active floodplain concept. Figure 5.60 shows the extent of the original 1 in 7 year floodplain (yellow) and the current extent of this flood frequency (blue).

An outcome of this is a transition of vegetation types on the floodplain as the flooding frequency boundaries shift to lower elevation. Figure 5.61 shows red gum regeneration within a wetland. This is not the traditional location of these trees, however this is now where the flooding frequency can support them. As the volume of environmental water reduces the floodplain will still have a range of flooding habitats but with smaller areas. This transition will result in dieback of pre-regulation areas of floodplain trees as new areas are established. This needs to be considered in setting management targets. For example the MDBC Living Murray set targets to maintain 70% of existing areas of red gum forests. Perhaps a more appropriate target is to set an extent of red gum but to allow these forests to transition to sustainable locations.

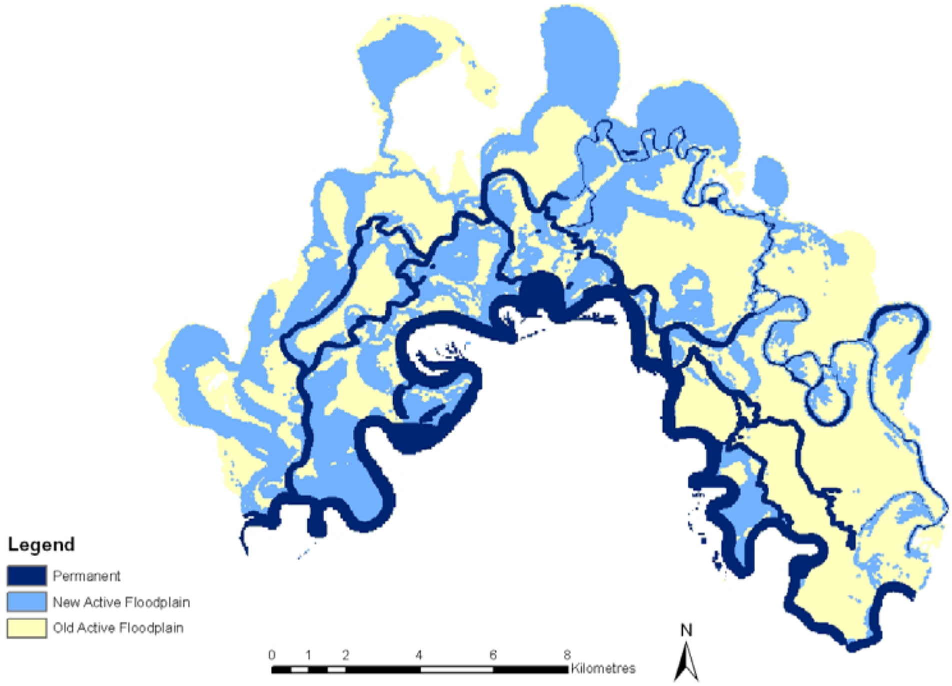

Figure 5.60 Difference between the old and the new 'active floodplain' for the Chowilla floodplain.

Figure 5.61 River red gum establishing in lower areas in the wetland as flood regimes alter.

5.5 CONCLUSION

For assessment of tree health modelling at a floodplain scale, a vegetation tree species map and tree health map was produced for the Chowilla floodplain. These maps were produced from combining existing vegetation mapping for South Australia and New South Wales with field data and remote sensing analysis.

One objective of this chapter was to develop a predictive model of the extent of floodplain inundation from surface water management scenarios, at the floodplain scale for the Chowilla floodplain, which includes new flow control structures. This objective was achieved by developing a pseudo-hydrodynamic model of floodplain inundation using existing creek level predictions.

An additional objective of the chapter was to produce a map of potential groundwater recharge and available water sources. This was achieved using soil hydraulic properties, changes in vegetation vigour following flood events identified by remote sensing and airborne electromagnetic imagery for the Chowilla floodplain. This mapping was used to provide input into a soil salinisation and tree health model.

The main objective of this chapter was to develop a predictive model of tree health for a semi-arid floodplain with changing management of flow and groundwater levels. This objective was achieved by the development of the WINDS model that allows changes in surface water flows and groundwater levels to be used to predict tree health. The model was applied to the Chowilla floodplain and predictions of tree health for 2003 were compared to field mapping. The results of the model were shown to have a 72% spatial matching.

The results from the WINDS model have shown that is useful for modelling floodplain vegetation health for a range of management scenarios including surface water and groundwater manipulation. It is therefore considered a useful tool for predicting impacts from future scenarios and to inform policy for floodplain protection and salinity mitigation.

Management scenarios have been modelled including lowering groundwater, a proposed new environmental regulator and enhanced flow regimes, and results compared to future 'do-nothing' scenarios. The effect of lowering the water table reduces the rate of soil salinisation but does not remove salt from the profile. Flow enhancement reduces salt but leaves the vegetation susceptible to degradation should a drought occur, as it has not addressed the rate of salinisation. The best benefit comes from a combination of the two scenarios. Despite the improvements in health predicted from groundwater lowering and flow enhancement, major improvements in vegetation will take many years to achieve under these management scenarios. The South Australian Government began construction of the $35 million 85 metre long environmental regulator through a decision process assisted by results from this research.

The WINDS model can be applied to any saline semi-arid floodplain once the initial base data is collected. The data required includes groundwater depth, groundwater salinity, recharge potential, tree salinity tolerance, river flow history and flooding patterns, rainfall and proximity to water features.

6 CONCLUSIONS

6.1 MODELLING FLOODPLAIN TREE HEALTH

This study has developed a number of models to predict floodplain tree health at regional and floodplain scales for management of both surface water and groundwater. Table 6.1 summarises the existing models and those developed in this study.

Table 6.1 Tools for assessing regional and floodplain scale surface and groundwater management scenarios. This study has developed a number of modelling tools (green) to add to existing models (orange). A future challenge is an integrated process model for tree health at regional scales (blue). The models are classified as either process based or habitat class based.

	Regional Scale	Floodplain Scale
Surface water flow impacts	PROCESS: River Murray Floodplain Inundation Model - RiM-FIM (4.3)	PROCESS: Existing hydrodynamic models
		PROCESS: RiM-FIM (5.2.2)
		PROCESS: Regulator model (5.2.2)
Groundwater and salinisation impacts	PROCESS: Floodplain Impacts Model - FIP (4.4)	PROCESS: Existing groundwater depth – MODFLOW models
		PROCESS: Groundwater salinity – WINDS model (5.3.6)
Floodplain tree health impacts	HABITAT: Flooding - drought Index (4.2.4)	HABITAT: Suitability model (5.3.2)
	HABITAT: Groundwater – FIP risk (4.3.3)	HABITAT: Groundwater - salinity risk (5.3.5)
	PROCESS: Integrated – future method or WINDS	PROCESS: Integrated - WINDS model (5.3.6)

A future goal is an integrated process model at regional scale. The Murray is of such importance that it would warrant the investment in data and modelling to use the floodplain scale model (WINDS) across the entire region. WINDS has already been used at several other floodplains within the region (Overton and Jolly, 2008; Overton *et al.*, 2008).

6.1.1 Regional Scale Management Scenarios

The goal of this study was to predict changes to floodplain inundation and groundwater depth with management scenarios and climate change, and the consequent impact on floodplain tree health from changes to soil water availability. The floodplain inundation and the floodplain impact from salinisation models developed have been calibrated against observed floodplain tree health and have shown that the approaches are suitable for providing a regional perspective of floodplain health and changes over time.

Policy and management at the regional scale is complex and has led to a separation of surface flow and groundwater issues. This has consequently led to conflict, as in the case of reducing salt loads to the river under the Basin Salinity Management Strategy and removing salt from the floodplains under the MDBC Living Murray program.

Regional environmental flow management scenarios considered were the increase in river flows and raising of weirs. The regional groundwater management scenarios considered included groundwater lowering from lowering weirs, pumping groundwater and reducing inflows into the floodplain from irrigation efficiencies, and increasing floods from enhanced flow scenarios and weir raising.

The 'active floodplain' concept put forward in this thesis identifies floodplain areas at risk from water stress as areas of less than a 1 in 7 year return period (now 70,000 ML/day) flood extent. This figure can also be used as the threshold flow value above which increased water from environmental flows is unlikely to be achievable given current water constraints. The 'active floodplain' concept is useful in defining the areas of the floodplain that are at risk from decreased flooding that would cause a loss of natural floodplain habitat. The 70,000 ML/day threshold is likely to be reducing as water extractions increase and further reduce the active floodplain area and the potential area of influence for environmental flows.

The RiM-FIM developed in Section 4.3 provided an initial model of vegetation health by comparing current with natural flood frequency, with poor health observed when floods occur less than one third of their original frequency. The floodplain impacts model (FIP) developed in Section 4.4 provided another model of vegetation health, where by health is dominated by salt accumulation from groundwater discharge.

River salinity levels were able to be modelled in the groundwater management model (FIP). Changing groundwater levels or groundwater inflows from irrigation can be assessed for their impact of river salinity. The impact on river salinity from surface water management options is no less important. The current flood risk model is not able to consider resulting impacts on river salinity from environmental flows releases or regulation infrastructure. This aspect should be considered in the future along with possible integration of the surface and groundwater management options.

6.1.2 Floodplain Scale Management Scenarios

The floodplain scale management scenarios that were considered in this thesis for the Chowilla floodplain, include groundwater lowering, enhanced flow regimes, new control structures, weir raising, channelling and wetland pumping. A number of management combinations were considered and were modelled for their impact on floodplain tree health and creek and wetland inundation.

The future health of the Chowilla floodplain is dependent on large increases in flow regimes and lowering of groundwater in most areas. No single management scenario was shown to meet the targets that have been set by MDBC for floodplain tree health. In many areas the flow regime required to reduce the high soil salinities from years of salt accumulation, is greater than that which had occurred under natural conditions before river regulation.

River red gum has been shown to be invading parts of the floodplain were decreased flow regime has caused a change in the vegetation type from *Phragmites australis* and other reeds to river red gum. This invasion is occurring all over the floodplain as the 'active floodplain' shrinks. This invasion can be seen as a natural response of the ecosystem to changing flow conditions. Bren (1992) points out that the question is not whether management can control these invasions but whether it should.

6.2 CRITIQUE OF THE METHODOLOGY AND FUTURE RESEARCH

The models developed in this study have been based on the dominant environmental factors affecting tree health, which include surface water and groundwater conditions. Other factors such as recruitment, disease and pest attack have been considered as secondary factors affecting the health of adult trees but which greatly effect the long term viability of the population.

The approach taken has been to develop habitat suitability models based on flow response relationships to provide the spatial aspect required for management at both regional and floodplain scales. The approach has then been to develop process-based models to increase the predictive capacity of management actions from incremental and integrated changes in environmental conditions. Other modelling approaches exist and could be adopted for the management scenarios being considered. There is a need for a comprehensive decision support system to manage the lower River Murray and this may require a suite of models.

There are a number of challenges to modelling ecosystem response including:

- A poor understanding of the water-dependent assets;
- A poor understanding of the ecosystem baseline condition and trajectory;
- The suitability of the monitoring data to assess 'health' and long term data on multiple species;
- The calibration of ecosystem response models and the error propagations;
- The use of models across multiple scales and their transferability to other areas;

- The ecosystem responds to multiple aspects of flow including the shape of hydrograph, water quality, turbidity and temperature;
- There are multiple factors affecting ecosystem response not just flow;
- That ecosystem response to changes in flow are not linear. For example, a 20% increase in environmental water does not imply a 20% increase in ecological health;
- That changes in the ecosystem condition/state will affect the outcomes of predictive models through feedback mechanisms which are not currently modelled;
- That there are few scientific studies on ecosystem response; and
- That future climate is uncertain.

A recent conference on ecosystem response modelling in the MDB (Saintilan and Overton, 2008) has identified that as part of the modelling of ecosystem responses there is a need for a range of scientific investigations. There is a need for improved inventory and health assessment of water-dependent ecosystems in the MDB and for their hydrology to be used as baselines to assess changes in ecosystem health. There is a lack of long-term monitoring projects that can demonstrate the link between changes in hydrology and changes in ecosystems. Ecosystem managers require explicit links between hydrology and ecology with rules that can guide the management of flows and water levels. There is a need for improved complexity of models to include multiple spatial and temporal scales. Understanding ecosystem changes requires an understanding not just of individual component responses to changes in hydrology but the whole ecosystem function. The cost of environmental flow releases needs to be considered in terms of the cost of the water and the value from the improved health of the ecosystems.

Further research on the regional scale investigations have been identified to include:

- Improved floodplain inundation modelling to incorporate new regulatory structures. This may require complex hydrodynamic modelling of the whole lower River Murray or improvements to the RiM-FIM model in areas where new regulation structures are being considered; and
- Incorporating flooding into the FIP model of groundwater and salinisation risk. The FIP model does not predict those areas where groundwater discharge from the shallow water table may be mitigated to some degree by flooding. It may be possible to do this in a probabilistic manner by including the compensating effects of flooding using the RiM-FIM and weighting the interrelated effects of elevation, flooding frequency and depth to groundwater. Alternatively, it may be possible to develop a soils map of the floodplain based on the vegetation maps, inferred floodplain elevations from the RiM-FIM, Landsat satellite imagery, high resolution aerial photography or field surveys. This would provide a means of identifying areas with lighter soil texture where floodplain salinisation is to some degree mitigated by the current flooding regime, and which would also benefit from improved inundation from environmental flows. It would also provide a means of identifying those areas of the floodplain that are likely to be recharge areas during flooding which is important for determining river salinity impacts of inundating floodplain areas using environmental flows.

Further research on floodplain groundwater and flooding interactions has been identified as:

- Further validation and updating of the vegetation health map from remote sensing, aerial photography and field work;
- Updating the groundwater depth surface, based on further analysis of vegetation health and aerial photography;

- Updating of the soil map to include sand deposits on inside of channel curves; and
- Converting the WINDS model from a moving five year average time period to a time period of actual events;
- Incorporating a full water balance model within the WINDS model. The previous modelling has determined that the major cause of discrepancy between actual vegetation health and modelled vegetation health from soil salinisation was the physical lack of available water. The addition of a recharge areas, rainfall and the drought factor to the WINDS model were highly simplified attempts at incorporation of the water content factor. Future work could incorporate this matric potential modelling with the osmotic potential modelling within the WINDS framework. The WINDS model considers the soil water availability from the plants ability to extract water from the soil given the salinity of that soil water. During drought conditions there may not be any physical water available for the plants regardless of the modelled salinity. River red gum trees on Chowilla have shown a decline during the summer of 2003 as a result of lack of water. The original WINDS model does not predict this die-back (although the improved WINDS model does) as the salinity of the soil water was probably not the dominating process that led to the die-back. The low rainfall and hot conditions of the previous two summers coupled with a lack of floods has probably caused a drying of the soil profile leading to drought stress of these trees; and
- Incorporation of impacts on vegetation communities other than tree communities.

6.3 CONCLUSIONS

The objectives of the thesis were to develop methods to assess environmental flow and groundwater management scenarios in semi-arid large rivers to manage floodplain tree health. The research achieved this by modelling floodplain tree health across the whole of the River Murray in South Australia and linked this to environmental flow management and land management decision making. To undertake this a number of key spatial information layers were required and the thesis has developed these at the required spatial scales appropriate to the decision making required, while identifying the minimum data requirements to allow the methodology to be applied quickly and economically in other areas.

Mapping of floodplain tree health in previous years has been linked with floodplain processes and changes in regional groundwater inflows and flooding frequencies. This mapping has shown clear evidence for irrigation impact on floodplain health, as well as providing a baseline for assessment of future vegetation health changes.

A floodplain inundation model (RiM-FIM) was developed for the lower River Murray (River Murray Floodplain Inundation Model - RiM-FIM) by combining remote sensing, GIS and hydraulic modelling. The RiM-FIM model was then used to demonstrate how it can assist in the management of environmental flows and the operation of flow control structures. Traditional river management has emphasised control of river flows to mitigate infrastructure damage but the move to environmental flow management has required a tool for identifying the extent of inundation from management scenarios. This is especially important for accountability in the use of this finite resource.

Floodplain inundation extents were monitored from Landsat satellite imagery for a range of flows, interpolated to model flood growth patterns and linked to a hydrological model of the river. The resulting model was analysed for a range of flows for any month of the year and weir configuration. It was independently tested using aerial photography of a flood event with an accuracy of approximately 15% underestimate of flood extent for that event. The results have supported the approach for determining flood extent over such a large area at a fraction of the cost of detailed elevation and hydrodynamic modelling, while improving on rating curves and hydrographs for analysis by including lateral and spatial elements. The GIS model allows prediction of impacts on infrastructure, wetlands and floodplain vegetation, allowing quantitative analysis of flood extent to be used as an input into the management decision process.

The RiM-FIM has recently been extended to the Hume Dam at Albury to cover the entire middle reaches of the River Murray, excluding headwaters and terminal lakes. This extended model has been used to assess environmental flow requirements from assessing risk of floodplain decline from a flood index that is derived from flooding history. In developing the RiM-FIM a pseudo-elevation dataset has been derived for the River Murray floodplain. This dataset has proven to be a useful resource for a number of floodplain modelling projects and was produced economically when compared to the capture of survey elevation data.

A floodplain impacts model (FIP) was developed for the lower River Murray to predict the effects of irrigation policy and river regulation by identifying areas of the floodplain at risk from saline seepage, soil salinisation and large salt inflows into the River Murray. The FIP model was then used to demonstrate the impacts of various groundwater management scenarios. A cross-sectional model of groundwater depth across the floodplain was implemented spatially on the entire of the River Murray floodplain in South Australia. The model has allowed prediction of the effects of irrigation on vegetation health for entire sections of the river for the first time. The model allows the effects of drowning through locking, irrigation and flooding to be separated spatially. The model has provided the basis for the development of irrigation zoning policy by identifying areas of the floodplain at risk from saline seepage, soil salinisation and large salt inflows into the River Murray.

A model of soil water availability (WINDS) was been developed for the Chowilla floodplain to model soil salinisation and water sources that impact on the health of the dominant floodplain tree species. This WINDS model was validated with current and historic vegetation health data. The WINDS model was then used to inform environmental flow and groundwater management scenarios on this regulated semi-arid floodplain. The resulting modelling provides spatial and temporal predictions on vegetation health and is able to inform flow strategies and groundwater lowering scenarios. A simple model of soil salinisation has been created using the major salt accumulation factors and used to infer vegetation health. The model is able to combine the effects of groundwater depth, groundwater salinity, flood frequency and soil type to determine vegetation tree health on a regional scale. It goes beyond the traditional finite element groundwater modelling, as it is specifically focused on vegetation health, and is able to accommodate more processes than just groundwater depth-salinity relationships. The model has been shown to successfully predict the spatial extent of vegetation tree health on the Chowilla floodplain with approximately 72% spatial correspondence with field recorded health. The methodology is particularly useful due to its limited parameter requirements compared with most vegetation health models and can be used in a temporally predictive manner. The model has been instrumental in the development of the Chowilla Environmental Flows Strategy Plan and the Groundwater Management Strategy Plan currently being developed.

This research has contributed to understanding of floodplain vegetation health processes by modelling the presence of freshwater lenses and predicting drought related die-back of riparian vegetation. Previous modelling has determined the major cause of discrepancy between actual vegetation health and modelled vegetation health from soil salinisation was the physical lack of available water. Conceptualisation of this soil water availability and freshwater lens storage of flood recharge advances the understanding of floodplain recharge processes. The application of airborne electromagnetic imagery to define recharge areas has advanced groundwater mapping in floodplains. The modelling of tree vegetation in other parts of the floodplain using the WINDS model has identified inaccuracies in the modelled groundwater depth layers near the River Murray where groundwater use by river red gum forests has lowered groundwater in this region below that previously estimated from bore interpolation. This has led to an improved groundwater interpolation method from sparse floodplain bores.

The relationship between vegetation species composition, soil salinity and water availability has been mapped for the first time for the floodplain. A predictive model has been produced for the distribution of vegetation communities based on water availability and soil salinity. It has provided new information on the patterns of distribution, and the tolerance and niche environments of understorey floodplain vegetation communities.

The research examined the link between spatial scales of data, environmental processes and decision making. The thesis presents three scales of vegetation health modelling that use a variety of spatial resolution data and address an array of management issues. Comparison of the three levels of modelling provides insight into priority data capture, monitoring requirements and requirements for management decisions. The three models presented include a region wide class model (RiM-FIM flood index), a simple process floodplain model (FIP) and a detailed temporally dynamic model for site specific locations (WINDS). The results show that increases in model complexity create increased precision of results at finer scales but provide little benefit for informing broader scale issues. Simpler models requiring fewer parameters are more economical to implement and provide similar results to more detailed models when scaled up to regional assessments. The thesis discusses which processes should be modelled at which scales to achieve the best management tools using limited resources.

The research provides tools and methods that can form part of the assessment of environmental flow management and groundwater management of a regulated semi-arid river floodplain by incorporating model outcomes at various scales. The health of riparian vegetation is the result of a complex integration of soil water availability from groundwater and surface flow provision. The modelling within the thesis has incorporated both aspects of water supply and in so doing has provided a spatial and temporal model to answer the fundamental questions of how much and how often when considering flow strategies and groundwater lowering scenarios. The approaches and modelling tools developed in this thesis for assessing environmental flow and groundwater management at floodplain and regional scales are applicable to other semi-arid rivers.

The research conducted for this thesis has been used in the Murray-Darling Guide to the Basin Plan (MDBA, 2010) for determining water requirements for a number of indicator key environmental assets in the River Murray using the RiM-FIM and the water requirements for the Chowilla indicator key environmental asset using the WINDS modelling.

The research is applicable to other semi-arid river systems such as those in the southern United States of America, Mexico, Spain, South Africa and other parts of Australia. The models developed can be used on any floodplain environment that has had changes to its flooding regime or its groundwater resources. These areas are highly degraded world wide and require improved management options to assist their sustainability into the future under changing climate and competing water users.

The thesis has improved the capacity of researchers and managers to predict outcomes from surface water and groundwater options in managing floodplain environments. This capacity is critical in light of overallocated floodplain environments world-wide.

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APPENDIX – SOFTWARE CODE FOR WINDS MODEL

Software code written in Visual Basic within ArcGIS

```
Private Sub UIButtonControl6_Click()
  Dim sLayerName As String
  Dim zwmax As Integer
  Dim zf As Integer
  Dim lowergw As Integer
  zwmax = 4
  zf = 1
  lowergw = 0

  ' Get the Map
  Dim pMxDoc As IMxDocument
  Set pMxDoc = ThisDocument
  Dim pMap As IMap
  Set pMap = pMxDoc.FocusMap

  ' Get the input rasters

  Dim pLayer As ILayer
  ' sLayerName = "c0zinit"
  sLayerName = "Co-2003"
  For I = 0 To pMap.LayerCount - 1
    If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
      Set pLayer = pMap.Layer(I)
      Exit For
    End If
  Next
  If Not TypeOf pLayer Is IRasterLayer Then
    Exit Sub
  End If
  Dim pRLayer As IRasterLayer
  Set pRLayer = pLayer
  Dim pInRaster As IRaster
  Set pInRaster = pRLayer.Raster

  Dim pLayer1 As ILayer
  sLayerName = "cg2"
  For I = 0 To pMap.LayerCount - 1
    If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
      Set pLayer1 = pMap.Layer(I)
      Exit For
    End If
  Next
  If Not TypeOf pLayer1 Is IRasterLayer Then
    Exit Sub
  End If
  Dim pRLayer1 As IRasterLayer
  Set pRLayer1 = pLayer1
  Dim pInRaster1 As IRaster
  Set pInRaster1 = pRLayer1.Raster

  Dim pLayer2 As ILayer
  sLayerName = "soils"
  For I = 0 To pMap.LayerCount - 1
    If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
      Set pLayer2 = pMap.Layer(I)
      Exit For
    End If
```

```
Next
If Not TypeOf pLayer2 Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer2 As IRasterLayer
Set pRLayer2 = pLayer2
Dim pInRaster2 As IRaster
Set pInRaster2 = pRLayer2.Raster

Dim pLayer3 As ILayer
sLayerName = "ts1y"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer3 = pMap.Layer(I)
    Exit For
  End If
Next
If Not TypeOf pLayer3 Is IRasterLayer Then
  Exit Sub
End If
Dim pRlayer3 As IRasterLayer
Set pRlayer3 = pLayer3
Dim pInRaster3 As IRaster
Set pInRaster3 = pRlayer3.Raster

Dim pLayer3b As ILayer
sLayerName = "ts2y"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer3b = pMap.Layer(I)
    Exit For
  End If
Next
If Not TypeOf pLayer3b Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer3b As IRasterLayer
Set pRLayer3b = pLayer3b
Dim pInRaster3b As IRaster
Set pInRaster3b = pRLayer3b.Raster

Dim pLayer3c As ILayer
sLayerName = "ts3y"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer3c = pMap.Layer(I)
    Exit For
  End If
Next
If Not TypeOf pLayer3c Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer3c As IRasterLayer
Set pRLayer3c = pLayer3c
Dim pInRaster3c As IRaster
Set pInRaster3c = pRLayer3c.Raster

Dim pLayer3d As ILayer
sLayerName = "ts4y"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer3d = pMap.Layer(I)
    Exit For
```

```
  End If
Next
If Not TypeOf pLayer3d Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer3d As IRasterLayer
Set pRLayer3d = pLayer3d
Dim pInRaster3d As IRaster
Set pInRaster3d = pRLayer3d.Raster

Dim pLayer3e As ILayer
sLayerName = "ts5y"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer3e = pMap.Layer(I)
    Exit For
  End If
Next
If Not TypeOf pLayer3e Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer3e As IRasterLayer
Set pRLayer3e = pLayer3e
Dim pInRaster3e As IRaster
Set pInRaster3e = pRLayer3e.Raster

Dim pLayer3f As ILayer
sLayerName = "ts6y"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer3f = pMap.Layer(I)
    Exit For
  End If
Next
If Not TypeOf pLayer3f Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer3f As IRasterLayer
Set pRLayer3f = pLayer3f
Dim pInRaster3f As IRaster
Set pInRaster3f = pRLayer3f.Raster

Dim pLayer3g As ILayer
sLayerName = "ts7y"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer3g = pMap.Layer(I)
    Exit For
  End If
Next
If Not TypeOf pLayer3g Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer3g As IRasterLayer
Set pRLayer3g = pLayer3g
Dim pInRaster3g As IRaster
Set pInRaster3g = pRLayer3g.Raster

Dim pLayer3h As ILayer
sLayerName = "ts8y"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer3h = pMap.Layer(I)
```

```
    Exit For
  End If
Next
If Not TypeOf pLayer3h Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer3h As IRasterLayer
Set pRLayer3h = pLayer3h
Dim pInRaster3h As IRaster
Set pInRaster3h = pRLayer3h.Raster

Dim pLayer3i As ILayer
sLayerName = "ts9y"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer3i = pMap.Layer(I)
    Exit For
  End If
Next
If Not TypeOf pLayer3i Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer3i As IRasterLayer
Set pRLayer3i = pLayer3g
Dim pInRaster3i As IRaster
Set pInRaster3i = pRLayer3i.Raster

Dim pLayer6 As ILayer
sLayerName = "clmin"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer6 = pMap.Layer(I)
    Exit For
  End If
Next
If Not TypeOf pLayer6 Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer6 As IRasterLayer
Set pRLayer6 = pLayer6
Dim pInRaster6 As IRaster
Set pInRaster6 = pRLayer6.Raster

Dim pLayer10 As ILayer
sLayerName = "zw3"
For I = 0 To pMap.LayerCount - 1
  If UCase(pMap.Layer(I).Name) = UCase(sLayerName) Then
    Set pLayer10 = pMap.Layer(I)
    Exit For
  End If
Next
If Not TypeOf pLayer10 Is IRasterLayer Then
  Exit Sub
End If
Dim pRLayer10 As IRasterLayer
Set pRLayer10 = pLayer10
Dim pInRaster10 As IRaster
Set pInRaster10 = pRLayer10.Raster


'------------------------------------------------------------------------------
'
```

```
'

 ' Create a RasterModel object
 Dim pRModel As IRasterModel
 Set pRModel = New RasterModel

 ' Create spatial analysis environment
 Dim pEnv As IRasterAnalysisEnvironment
 Set pEnv = pRModel

 ' Set output workspace
 Dim pWS As IWorkspace
 Dim pWSF As IWorkspaceFactory
 Set pWSF = New RasterWorkspaceFactory
 Set pWS = pWSF.OpenFromFile("n:\chowilla\temp", 0)
 Set pEnv.OutWorkspace = pWS

 ' set 2c to equal 4b and 2b to equal 3

 ' Set model, vbLf is used to separate equations
 pRModel.Script = " [Rcala] = con ( [Rsoils] == 1 , 0.0002 , con ( [Rsoils] == 3 , 0.0016 , con ( [Rsoils] == 4 , 0.0003 , con ( [Rsoils] == 5 , 0.0002 , con ( [Rsoils] == 6 , 0.001 , con ( [Rsoils] == 7 , 0.0003 , con ( [Rsoils] == 8 , 0.0005 , con ( [Rsoils] == 9 , 0.0009 , con ( [Rsoils] == 10 , 0.0008 , 0 ))))))))) " + vbLf + _
        " [Rcalp] = con ( [Rsoils] == 1 , -2.701 , con ( [Rsoils] == 3 , -2.507 , con ( [Rsoils] == 4 , -2.684 , con ( [Rsoils] == 5 , -2.701 , con ( [Rsoils] == 6 , -2.630 , con ( [Rsoils] == 7 , -2.684 , con ( [Rsoils] == 8 , -2.911 , con ( [Rsoils] == 9 , -2.271 , con ( [Rsoils] == 10 , -2.448 , 0 ))))))))) " + vbLf + _
        " [Rcalks] = con ( [Rsoils] == 1 , 0.006 , con ( [Rsoils] == 3 , 0.006 , con ( [Rsoils] == 4 , 0.006 , con ( [Rsoils] == 5 , 0.006 , con ( [Rsoils] == 6 , 0.006 , con ( [Rsoils] == 7 , 0.006 , con ( [Rsoils] == 8 , 0.5 , con ( [Rsoils] == 9 , 0.001 , con ( [Rsoils] == 10 , 0.002 , 0 ))))))))) " + vbLf + _
        " [Rcalphid] = con ( [Rsoils] == 1 , 0.1 , con ( [Rsoils] == 3 , 0.22 , con ( [Rsoils] == 4 , 0.1 , con ( [Rsoils] == 5 , 0.1 , con ( [Rsoils] == 6 , 0.1 , con ( [Rsoils] == 7 , 0.1 , con ( [Rsoils] == 8 , 0.1 , con ( [Rsoils] == 9 , 0.15 , con ( [Rsoils] == 10 , 0.22 , 0 ))))))))) " + vbLf + _
        " [Rcalphis] = con ( [Rsoils] == 1 , 0.36 , con ( [Rsoils] == 3 , 0.45 , con ( [Rsoils] == 4 , 0.36 , con ( [Rsoils] == 5 , 0.36 , con ( [Rsoils] == 6 , 0.36 , con ( [Rsoils] == 7 , 0.36 , con ( [Rsoils] == 8 , 0.4 , con ( [Rsoils] == 9 , 0.6 , con ( [Rsoils] == 10 , 0.45 , 0 ))))))))) " + vbLf + _
        " [Rq] = con ( [Rzw] > 1 , [Rcala] * pow ( ( [Rzw] + 2 ) , [Rcalp] ) , 0.001) " + vbLf + _
        " [Rcoz1] = (( [Rcg] / 4 ) * ( ( [Rq] * ( 1826 - [Rts3] ) / [Rcalphid] ) - ( [Rcalks] * [Rts3] / [Rcalphis] ))) + [Rco] " + vbLf + _
        " [Rcoz2] = (( [Rcg] / 4 ) * ( ( [Rq] * ( 1826 - [Rts2] ) / [Rcalphid] ) - ( [Rcalks] * [Rts2] / [Rcalphis] ))) + [Rcoz1] " + vbLf + _
        " [Rcoz3] = (( [Rcg] / 4 ) * ( ( [Rq] * ( 1826 - [Rts1] ) / [Rcalphid] ) - ( [Rcalks] * [Rts1] / [Rcalphis] ))) + [Rcoz2] " + vbLf + _
        " [Rcoz1c] = con ( [Rcoz1] > 0 , con ( [Rcoz1] > 55 , 55 , [Rcoz1] ) , 0 ) " + vbLf + _
        " [Rcoz2c] = con ( [Rcoz2] > 0 , con ( [Rcoz2] > 55 , 55 , [Rcoz2] ) , 0 ) " + vbLf + _
        " [Rcoz3c] = con ( [Rcoz3] > 0 , con ( [Rcoz3] > 55 , 55 , [Rcoz3] ) , 0 ) " + vbLf + _
        " [Rxw1] = ( 1 - ( [Rcoz1c] / 55 ) ) * 0.15 " + vbLf + _
        " [Rxw2] = ( 1 - ( [Rcoz2c] / 55 ) ) * 0.30 " + vbLf + _
        " [Rxw3] = ( 1 - ( [Rcoz3c] / 55 ) ) * 0.55 " + vbLf + _
        " [Rwinds] = con ( [Rclmin] > 0 , [Rxw1] + [Rxw2] + [Rxw3] , 2 ) "


 ' Bind to raster
 pRModel.BindRaster pInRaster, "Rco"
 pRModel.BindRaster pInRaster1, "Rcg"
 pRModel.BindRaster pInRaster3, "Rts1" '1826
 pRModel.BindRaster pInRaster3b, "Rts2" '1826
 pRModel.BindRaster pInRaster3c, "Rts3" '1826
 pRModel.BindRaster pInRaster3d, "Rts4" '1827
```

```
 pRModel.BindRaster pInRaster3e, "Rts5" '1826
 pRModel.BindRaster pInRaster3f, "Rts6" '1826
 pRModel.BindRaster pInRaster3g, "Rts7" '1826
 pRModel.BindRaster pInRaster3g, "Rts8" '1827
 pRModel.BindRaster pInRaster3g, "Rts9" '2331
 pRModel.BindRaster pInRaster6, "Rclmin"
 pRModel.BindRaster pInRaster10, "Rzw"
 pRModel.BindRaster pInRaster2, "Rsoils"

 ' Run the model

 pRModel.Execute

'--------------------------------------------------------------------------

  ' Get outputs
   Dim pRasterRwinds As IRaster
 Set pRasterRwinds = pRModel.BoundRaster("Rwinds")

 ' Unbind raster
 pRModel.UnbindSymbol "Rco"
 pRModel.UnbindSymbol "Rcg"
 pRModel.UnbindSymbol "Rts1"
 pRModel.UnbindSymbol "Rts2"
 pRModel.UnbindSymbol "Rts3"
 pRModel.UnbindSymbol "Rts4"
 pRModel.UnbindSymbol "Rts5"
 pRModel.UnbindSymbol "Rts6"
 pRModel.UnbindSymbol "Rts7"
 pRModel.UnbindSymbol "Rts8"
 pRModel.UnbindSymbol "Rts9"
 pRModel.UnbindSymbol "Rclmin"
 pRModel.UnbindSymbol "Rzw"
 pRModel.UnbindSymbol "Rq"
 pRModel.UnbindSymbol "Rcala"
 pRModel.UnbindSymbol "Rcalp"
 pRModel.UnbindSymbol "Rcalks"
 pRModel.UnbindSymbol "Rcalphid"
 pRModel.UnbindSymbol "Rcalphis"
 pRModel.UnbindSymbol "Rcoz1"
 pRModel.UnbindSymbol "Rcoz2"
 pRModel.UnbindSymbol "Rcoz3"
 pRModel.UnbindSymbol "Rcoz1c"
 pRModel.UnbindSymbol "Rcoz2c"
 pRModel.UnbindSymbol "Rcoz3c"
 pRModel.UnbindSymbol "Rxw1"
 pRModel.UnbindSymbol "Rxw2"
 pRModel.UnbindSymbol "Rxw3"
 pRModel.UnbindSymbol "Rwinds"

'-----------------------------------------------------------------------

' Add it into ArcMap
 Set pRLayer = New RasterLayer
 pRLayer.CreateFromRaster pRasterRwinds
 pMap.AddLayer pRLayer
 pMxDoc.FocusMap.ClearSelection
EndSub
```