EFFICIENT WATER ALLOCATION IN A HETEROGENEOUS CATCHMENT SETTING

Lisa Yu-Ting Lee

A thesis submitted in fulfillment of the requirements for the degree of Doctor of Philosophy

Agricultural and Resource Economics
Faculty of Agriculture, Food and Natural Resources
The University of Sydney

2007

CERTIFICATION OF ORIGINALITY

I hereby certify that the substance of the material used in this study has not been submitted already for any degree and is not currently being submitted for any other degree and that to the best of my knowledge any help received in preparing this thesis, and all reference material used, have been acknowledged.

Lisa Yu-Ting Lee

2007

ABSTRACT

Efficient Water Allocation in a Heterogeneous Catchment Setting

The problem of water scarcity has become one of the most controversial topics in Australia over the past decades, with particular focus being the 'sustainable' allocation of water between extractive and environmental purposes. Geographical factors are defining the extreme variability in climate and water supply in Australia and, in the past, this was used as a rationale for the construction of large irrigation projects to deliver water to rural, urban, and industrial users. During this 'expansionary' phase of Australia's water use sector, the cost of augmenting supply was relatively low and environmental considerations were secondary to the development imperative. As a result, water resources became over-allocated for extractive uses spurred on by consistent underpricing of water, which indicated a failure to reflect the true cost of water supply. As Australia's water economy entered a 'mature' phase, it was no longer possible to increase supply cheaply as the most easily accessible water resources had already been captured. This was followed by widespread environmental degradation manifested in the Murray-Darling Basin, the nation's largest river basin which hosts much of Australia's agricultural production. Consequently, the focus shifted towards demand management, leading to a myriad of regulation aimed at increasing the allocative efficiency of scarce water resources. Towards this end, substantial government funding was injected into the various initiatives throughout the water reform process.

Despite the on-going government activities in the area of water reform, the understanding of the actual economic impact and environmental outcomes of various water policies in practice remains limited. In the absence of such understanding, the effectiveness of various government water initiatives is ambiguous and inevitably compromised.

The present study addresses this knowledge gap by establishing a method for evaluating the economic and environmental outcomes of environmentally-oriented polices that affect irrigated industries in a catchment. The method is based on an integrated biophysical and economic modelling approach, which enables spatial relationships to be

captured accurately allowing a more realistic analysis. Information generated from a computer based biophysical simulation model form the basis of an economic optimisation model with constraints pertaining to environmental targets and water supply limits. The economic model consists of a linear programming and dynamic programming component, and involves the optimisation of resource use from a catchment manager's perspective, seeking to achieve efficient resource use but at the same time conform to given environmental objectives. This two-stage modelling process was required to determine the optimal intra-seasonal and inter-seasonal water allocation, given various catchment environmental targets. The interdisciplinary approach enables the economic and ecological outcomes of the catchment management policies to be simulated and assessed at a spatially explicit scale, due to the link to Geographical Information Systems (GIS) in the biophysical model.

The overall objective was to create a decision-making framework that could be used to determine the least-cost means of meeting environmental targets and resource constraints. The solutions to the analysis are directly applicable to the case study, the Mooki catchment in northern New South Wales (NSW), but with an adaptable framework that can be applied to other catchments. Specific objectives include an evaluation of the possibility of using alternative irrigation systems, as well as an evaluation of the benefits that can be realised by establishing water market, in the light of environmentally-oriented catchment policies for the case study. The economic cost of achieving environmental targets pertaining to environmental flow requirements and salinity reduction, in the form of end-of-valley salinity targets, was explicitly calculated through the economic model.

While salinity targets have been set for NSW catchments, the practicality of such targets is in question, given the substantial reductions in water allocation to irrigation activities, which is one of the key contributors to deep-drainage. An additional objective in this study was therefore to investigate the value of having deep drainage targets. A further consideration is the effect of "external agents" in the form of government plans to buy-back entitlements from irrigation districts, or the possibility of significant water rights purchases from mining industries. The implications of external water market entrants on the regional agricultural industry were examined.

Some conclusions and recommendations drawn from the results of this thesis are as follows:

- Alternative irrigation systems, including pivot and drip irrigation, are beneficial to irrigators in the Mooki basin, improving their water use efficiency and productivity. Pivot irrigation systems were shown to be the optimal system for most of the catchment, while drip irrigation systems are less economically viable due to the high cost of investment. Significantly, the viability of these irrigation systems is reliant on the security of water supply. It has been demonstrated that where groundwater is used in conjunction with pivot or drip, profit is consistently higher compared to where surface water is used. This relates to the uncertainty of river flow in an ephemeral system, which result in irregular irrigation water availability and, consequently, lower crop yields. To encourage investment in water efficient technologies, it is important there are ample and secure water supplies. Considering the recent cuts in groundwater entitlements in the Mooki basin, and the prospect of future reductions in both surface and ground water rights, irrigators in the region may be reluctant to make the investment. This is especially the case where the capital requirement for water efficient technologies is substantial. It reiterates the importance of secure water rights and clear policy implications for future supplies.
- It was found that the initial area-based water licensing led to an inefficient distribution of water amongst irrigators, and that a fully functional water market would enhance basin profitability since water is shifted to higher value uses in the downstream-most region of the Mooki. This leads to an efficient outcome, as irrigation areas contract and leaves more land available for conservation purposes. The presence of a water market also augments the value of irrigation technologies, leading to a shift away from tradition furrow irrigation towards pivot irrigation systems. In this light, it would be more effective for government

funding to be used in promoting water trade than subsidising the cost of irrigation technologies.

- The opportunity costs of meeting environmental flow and salinity reduction targets are also reduced where water efficient technologies and water trading are utilised. However, where these environmental targets are stringent, the economic burden will be substantial even if water trading or irrigation technologies are used. Where a significant reallocation of water for environmental flows or reduction in salinity is envisaged, the resulting opportunity costs should ideally be justified by the environmental benefits that are generated.
- A dual-instrument, simultaneously managing water use and deep drainage through separate instruments, is unnecessary. Surface water caps alone provide sufficient conservation signals to reduce unproductive water losses to deep drainage. If drainage is not at a critical level (i.e. not excessively contributing to salinity) then it is more efficient to impose caps on surface water, which have the added benefit of increased environmental flows. Furthermore, (surface) water caps do not affect groundwater use, which have recently been cut to 'sustainable' extraction rates. This is preferable to imposing additional salinity caps which will cause groundwater use to fall below sustainable extraction levels, and are likely to generate excessive administrative costs. Also, given the difficulty and cost associated with deep drainage measurement, the economic cost for setting a 'wrong' target will also be high.
- The economic impact of an external agent competing for water in the regional water market is not expected to be significant. The volume of water that is demanded by a hypothetical coal mine in the catchment represent a relatively small portion of surface water supply available to the irrigators. The effect on the regional agricultural industry is also fairly small compared to the value that the water represents to the coalmine. While the general conclusion is that there is a net benefit to permitting external buyers to enter the water market, other indirect effects such as employment or environmental impacts have not been factored into the analysis. These considerations would form a valuable extension in future research.

This thesis represents one of few bio-economic studies that can provide spatially-referenced solutions in an Australian context, through the combined use of a GIS-integrated biophysical and economic model. The ultimate intention is to demonstrate the value of developing a streamlined, interdisciplinary framework that utilises the power of GIS, to enhance the efficiency of natural resource management and lead to a socially optimal outcome.

ACKNOWLEDGEMENTS

This work was funded by the Commonwealth Government Australian Postgraduate Awards scholarship, the Faculty of Agriculture, Food and Natural Resources top-up scholarship, and the Cotton Catchment Communities Cooperative Research Centre top-up scholarship. I would like to express my gratitude for the generosity of these institutions which have made this thesis possible.

I would like to express sincere thanks to my supervisor, Dr Tihomir Ancev, for his untiringly enthusiasm in my research from the day he has arrived at the faculty. Thank you for your on-going support and encouragement, and seeing me through my undergraduate and postgraduate years. You have been a tremendous influence in my development as a researcher and I am endlessly grateful for your generosity.

I would also like to extend my appreciation to my associate supervisor, Dr Willem Vervoort, who has been exceedingly helpful and energetic throughout the course of this research. This thesis would not have been possible without your invaluable input and attention to detail, and I thank you for your devotion.

Many thanks to Dr Michael Harris and Dr Ross Drynan, who have provided significant academic guidance for my project and have been generous with their help and advice. I would also like to acknowledge Associate Professor Fredoun Ahmadi-Esfahani, who has persuaded me to undertake further research at the beginning. I am greatly indebted to you all.

Finally, I would like to take this opportunity to thank my friends and family, who have been there through my ups and downs. Thank you to my colleagues in the Wallace Basement, who have been extremely supportive and made this experience so enjoyable. I am very grateful for the friendships we have forged. Above all, thank you to my mother, my father, my sister, and Gaston, who have constantly been there to help me through tough times. I thank you wholeheartedly for your kind words of encouragement and unwavering faith in me. This thesis is dedicated to you.

Lisa Yu-Ting Lee
Sydney, August 2007.

TABLE OF CONTENTS

Chapter, Headings, Sub-Headings **Page**

LIST OF TABLES

LIST OF FIGURES

LIST OF INSETS

Chapter 1.

INTRODUCTION

Vigorous debate surrounding the efficient use of water resources has been taking place in Australia in recent years, stimulated by the prolonged drought conditions and increasing focus on environmental value of water. At the centre of the debate over water management is the Murray-Darling Basin (MDB), regarded as Australia's food bowl that produces over 41% of Australia's agricultural output. The importance of the basin is highlighted by the fact that the majority of all water use in Australia is consumed in the MDB, with almost 90% of the system's water diverted for extractive uses (CRCIF 2005; DAFF 2006). Numerous water policies have been implemented since the 1994 Council of Australian Governments (COAG) reform, in recognition of inefficient water allocation between extractive and non-extractive uses. However, there is great uncertainty surrounding the various strategies to improve irrigation efficiency from both ecological and economic standpoints. The aim of this research is to propose a method that can be used to determine the economic impact of various environmental policies, and the way that these policies can be achieved at least cost. The method is showcased on a case-specific basis using spatially explicit information.

The problem of the misallocation of water resources can be largely attributed to the long period during which water was underpriced in Australia. This was driven by the early expansionary phase, where irrigation development was subsidised which resulted in irrigation water never being priced at its full cost (Randall 1981a; Davidson 1969). Consequently, excessive water diversion has led to severe degradation of the riverine environment, as well as raising groundwater tables, resulting in secondary salinity and salt scalds on agricultural land. While it is important to limit the environmental degradation resulting from the inefficient allocation of water resources, it is also important that proposed measures towards better water management generate maximum net social benefits. This may be achieved by shifting inefficiently used water resources from irrigation to where it has greater value, which may be in other sectors of the economy or in the environment. Water markets have been relied upon to achieve such an efficient reallocation, and have been introduced in most catchments in New South Wales (NSW). This was done in conjunction with Water Sharing Plans, which stipulate a set of extraction rules and minimum environmental flow requirements. In addition, the Murray-Darling Basin Commission (MDBC) has implemented a Salinity Management Strategy to control salt loads entering the Murray-Darling from various parts of the Basin. The Federal government has also recently announced a National Water Plan to buy-back entitlements to provide for environmental flows, as well as to support investment in water efficient technologies on-farm.

However, while there is much discussion in the literature pertaining to the efficient allocation of water on a catchment scale (Heady and Hall 1968; Beare et al. 1998; Bernardo et al. 1987; Zilberman et al. 1991; Tsur and Dinar 1995; Bjornlund 2003b; Freebairn 2005; Heaney and Beare 2001; and others), empirical work to assess the cost of various environmental targets and to determine the least-cost means of achieving these objectives seem to be lacking. There is considerable pressure for irrigation industries to improve water use efficiency and improve their environmental record, and yet the effects of the various measures in this direction on the economic performance of the affected industries, are unclear. To meet these challenges irrigators need to make optimal decisions about crop production, source of irrigation water, land allocation, and in particular the use of water efficient irrigation technologies which require substantial capital investment. Furthermore, the effects of the functioning water market in relation to the abovementioned aspects have yet to be fully understood in an empirical setting. The increasing competition that irrigators are facing from other industries, most notably the mining industry (Strang 2006) has also become an additional challenge in recent times.

This research therefore aims to develop an empirical method that can be used to determine the optimal choices that should be made at a farm and catchment level in relation to a number of control variables (crop choice, water use, irrigation systems, water trade, etc.), so that the economic objectives (e.g. profit maximisation) are achieved, but at the same time the outcomes are conforming to the environmental imperatives of the community. Particular environmental concerns addressed in this research are in relation to environmental flow provisions and salinity targets. The research approaches the problem by using an interdisciplinary framework, including a Geographical Information System (GIS) based biophysical model such that the results are directly applicable to the study area but flexible enough to be transferred to other catchments.

This integrated framework is used to evaluate a number of research questions pertaining to the effectiveness of various policies targeted towards improving environmental outcomes in a catchment with a substantial irrigated agriculture industry, as well as the impact of increased competition for water from other uses and sectors. As part of the analysis, the value of a fully-functioning water market within a case study irrigation district will be investigated, with the key research questions being the impact of water trading on the regional economy and its role in achieving environmental targets at the least-cost.

Another facet to explore is the value of using policy instruments designed to control salinity in conjunction with policy instruments designed to control the quantity of water diverted from the river systems. While salinity standards have been set for NSW catchments as part of the MDB Salinity Management Strategy, the practicality of a widespread salinity capping scheme is questionable, given the difficulty and cost associated with measuring diffuse sources of salinity across a large landscape. Due to the conjunctive nature of water use and deep drainage (water lost to the soil profile in irrigation, contributing to salinity), there is also the possibility of duplicating administrative costs with little net benefit to environmental objectives. There is the need to assess the capacity of policies that target deep drainage to achieve salinity mitigation, and thereby evaluate the need for a separate policy instrument to manage deep drainage occurrence. An objective of this research is to look at the economic viability of introducing a separate target to contend with salinity, through the imposition of a salinity, or deep-drainage cap.

The impact of future reduced water allocations may be confounded by increasing competition for water. Substantial competition for the basin's irrigation industry may arise from the entry of external buyers into the regional water market, which has significant implications for the regional agriculture. This comes amidst discussions of government buy-backs of entitlements for environmental flows, as part of the Commonwealth Plan for National Water Security (Howard 2007), and the possibility of an expansion of coal mining, a water intensive industry, in the northern region of NSW. This has instigated considerable community debate pertaining to environmental concerns as well as competition for infrastructure and water, given plans to substantially reduce surface and groundwater entitlements over the next five to ten years. The potential effect of the presence of an external water buyer in a catchment, personified by a hypothetical coal mine, on optimal water allocation in the catchment where a functioning water market exist is another research question addressed in this thesis.

1.1 RESEARCH OBJECTIVES

In this research, the overall aim is to develop a decision-making framework that can be used to determine the production pattern in the basin which allows efficient irrigators to maintain profitability, but simultaneously conform to water sustainability targets and environmental objectives of the community. The framework involves a combined economic and GIS-based hydrological modelling of production activities on a catchment level, enabling a spatially explicit analysis of production and resource use. Modelling at a catchment scale allows for an analysis that readily incorporates the social values of water, allowing the optimal spatial distribution of resources and agricultural activities to be determined. Specific objectives of this research are:

1. To apply the GIS-based decision-making framework to the case study basin, and thereby construct a resource management framework that is transferable to other catchments;
2. To determine optimal spatial choice of alternative irrigation systems in a catchment, in the face of tightening environmental targets;

3. To evaluate the improvement in allocative efficiency brought about by a fully-functioning water market in the case study region;
4. To determine the value of dual policy instruments (deep drainage target and environmental water flow target) to control environmental outcomes in a catchment. Deep drainage has serious environmental consequences in terms of increased groundwater and soil salinity, and potential water logging. While it is important to reduce deep drainage, it is important to evaluate the efficacy of creating a separate instrument to manage its occurrence;
5. To assess the potential impact of a significant industrial water user participating in the regional water market, on the regional agricultural industry. This can correspond to either Government buy-back of entitlements or to an entry of a water-intensive mining industry in the catchment.

1.2 RESEARCH APPROACH

The research approach in this thesis has been to conduct a spatially explicit analysis of optimal decisions in spatially referenced irrigation enterprises. This is in contrast to the representative farm approach, which has been extensively used in economic analyses (Aluwihare et al. 2005; CARE 2003; Letcher and Jakeman 2002; and others). The representative farm approach, which typically utilises average values for parameters of interest (e.g. deep drainage, crop yield, irrigation, etc.) in effect analyses inherently heterogeneous landscapes by imposing assumptions of homogeneity. While this approach reduces the complexity involved and can usefully shed light on some important phenomena, it can result in misleading conclusions which lead to blanket policies that are costly and ineffective, because they do not take into account the differentiated nature of catchments.

In contrast to this, the parameters of interest in this thesis are treated on a site-specific basis and at a high level of spatial detail. This is achieved through the use of a GIS based biophysical computerised simulation model. The model was applied on the Mooki River catchment, because of data availability and biophysical modelling expertise.

Nevertheless, the theoretical and methodological approaches are flexible enough to be implemented, with minor adjustments, to other catchments or regions.

A summary of the steps undertaken in this thesis is as follows:

1. An overview of the current water situation and reform process in Australia is conducted, with more detailed inquiry into the New South Wales (NSW) water policies, in particular those relating to the case study catchment. The aim is to determine the changes pertaining to water allocations that affect irrigators in the region;
2. Simulations are run in the hydrologic-biophysical model, the Soil and Water Assessment Tool (SWAT), which was set up with parameters specific to the catchment. The SWAT uses historic climate data including rainfall and river flow to simulate agronomic and hydrologic variables, depending on the production activities specified. Simulations were run under various scenarios with different production parameters, and the output from the simulations are used as data input for the economic model;
3. An economic model was developed based on the data from SWAT, and was set up as a dynamic optimisation model through a finite time horizon. For each time period, a separate optimisation model was solved in the form of a linear programming model integrated with an intra-regional water trade model. The periods were linked by resource constraints and limits on the volume of extraction in each period. This is based on the current water reform policies for the Mooki Basin. Three different treatments were analysed, each with different levels of choice variables available, including alternative irrigation systems (AIS) and water trading.
4. Alternative scenarios were run within each treatment, with parameterised environmental flow requirements, salinity caps, water market prices, and the possibility of an external agent entering the regional water market. Resource allocation under each scenario was driven by the optimisation objective in the economic model, given the constraints on resources.

5. Inferences are drawn from the results of the various scenarios and treatments, and recommendations made regarding water resource management towards the socially optimal outcome. The recommendations are made in line with the economic efficiency criteria to shed light on the efficacy of various environmental policies, and how the policy objectives can be achieved at the least-cost.

1.3 OUTLINE OF THESIS

This thesis is comprised of nine chapters. Chapter 1 introduced the reader to the research question and the approach taken. Chapter 2 provides a review of the current situation with water allocation and use in Australia, and some historical notes on the evolution of the problem. In Chapter 3, the institutional arrangement for water management in the Australian context is presented, in order to introduce the various environmental policies relevant to this thesis. Chapter 4 provides a review of literature pertaining to water economics, which highlights the complexity involved in managing this important natural resource. Chapter 5 outlines the economic theory to resource allocation, which underpins the operation of the economic model used in this thesis. In Chapter 6, a description of the case study basin, the Mooki, and the various characteristics specific to this basin are presented. Chapter 7 presents the specific economic model used for this thesis, as well as the integrated approach and utilised data. The results from the model simulations are discussed in Chapter 8. The ultimate Chapter 9 summarises the research findings and provides some conclusions, policy implications, limitations and directions for further research work.

Chapter 2.

BACKGROUND OF WATER USE IN AUSTRALIA

The water supply situation in Australia underpins the distributional problems at the crux of the nation's water management debate. A review of the geographical features that dictate the aridity of Australia's landscape, and the ensuing problems of water supply, is provided in this chapter. The transition between the developmental and mature phase of Australia's water economy is also reviewed, with reference to the range of government initiatives aimed at rectifying the resource allocation problem throughout this period.

2.1 STATE OF WATER RESOURCES IN AUSTRALIA

Australia is regarded as the driest inhabited continent on earth, with the lowest runoff to precipitation ratio compared to other continents (Haisman 2005). However, on a per capita basis Australia is no drier than other countries, with large areas of well-watered land. In this light, water scarcity can be attributed to the extreme variability in water resources both spatially and temporally. More than one-third of Australia produces no surface runoff at all, with 60% of runoff occurring north of the Tropic of Capricorn and the rest being concentrated in Tasmania (Pigram 1986). The continent's spatial variability in rainfall is reflected by the fact that floods and droughts are the most frequently occurring natural disasters. The distribution of freshwater runoff in Australia is illustrated in Figure 2.1.

A number of factors contribute to Australia's aridity. Firstly, the continent is characterised by a flat landscape, the most distinctive topographic feature being the Great Dividing Range along the eastern and south-eastern coastline. Orographic lifting of moist air over the mountain ranges results in greater precipitation along the coast but leaves arid conditions inland. Secondly, evaporation is very high; on average 87% of all moisture is lost to evaporation compared to 60% occurring in North America and Europe. Although the rate of evaporation is variable across the landscape, only a small proportion of rainfall becomes surface runoff, much of which flows out to sea. The moisture is evaporated or transpired by vegetation, with the excess seeping underground (AWRC 1975).

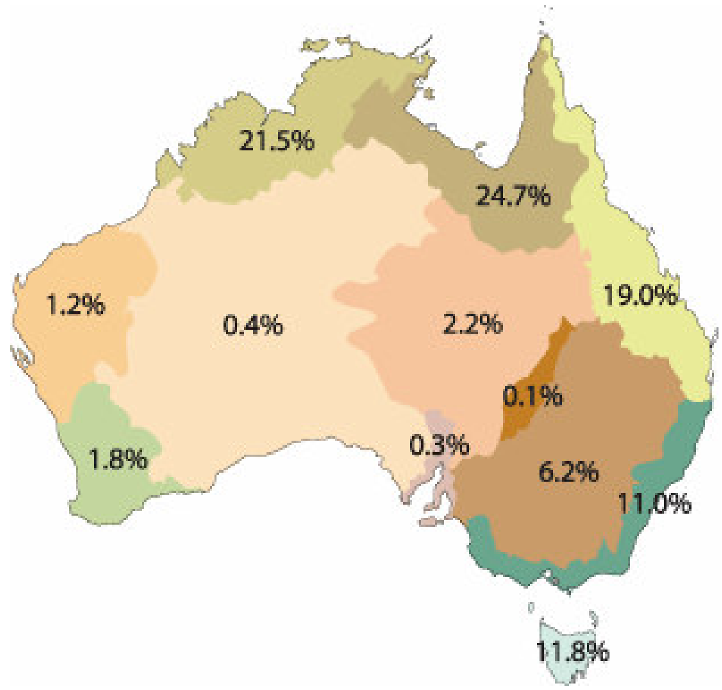

Figure 2.1: Percentage runoff distribution from each drainage division (source: NLWRA 2001).

Droughts and floods are common events in Australia. The continent is situated at the latitudes of the subtropical high pressure belt, which leads to generally drier climates. In addition, Australia is affected by the El Nino climatic events which cause weather fluctuations in the Pacific Ocean, at times inducing droughts in eastern and northern Australia (BOM 2007). One of the most severe El Nino events occurred in 1991-1995, and caused an estimated cost of $5 billion to the economy. Severe droughts could also be unrelated to the El Nino. Examples of this include the "Federation drought" from 1895 to 1903, during which stock numbers fell by more than 40%. Flooding is thought to follow droughts, due to the heavy rain accompanying the breakdown of El Nino. Flooding also occurs in the tropics during monsoon seasons, and commonly along the east coast and in the south induced by pressure systems. It is considered Australia's most costly natural disaster, inflicting $400 million a year in damages (BOM 2007).

Due to this climatic variability, a number of large storage dams have been constructed to secure water for human consumptive demand. Major irrigation expansion occurred in the

1950's and 1960's, with large irrigation projects funded by the federal government and operated by public agencies. The intention was to 'drought-proof' agriculture, which was based on the notion that the value of irrigated production and regional development justifies the cost of public investment in irrigation infrastructure. This notion however was not supported with rigorous economic assessments (Pigram 1986). To date, Australia has 447 dams with a capacity of 79 million ML of water, out of which around 20 million ML being consumed by agriculture, industrial and urban uses each year (NLWRA 2000).

The most water intensive industry is irrigated agriculture, which occupies less than 1% of total agricultural area in Australia but consumes 75% of all surface water diverted. This primarily occurs in the Murray Darling Basin (MDB) (DAFF 2006; CRCIF 2005). Most water is used for irrigation crops and pastures, making up 10,085GL or 90.5% of water used by agricultural establishments. The following table illustrates the agricultural water use by state in 2004-05, ranked by the level of consumption (Table 2-1). It can be seen that NSW ranks as the highest agricultural water consumer in Australia.

Table 2-1: Agricultural water use in Australia, by State 2004-05 (source: Trewin 2006).

	Agricultural establishments	Irrigation	Other agricultural uses	Total
	no.	ML	ML	ML
NSW(a)	40 162	3 716 557	259 551	3 976 108
Vic.	32 357	2 363 764	206 456	2 570 219
Qld	27 132	2 613 404	251 486	2 864 889
SA	14 111	877 818	^127 010	1 004 828
WA	11 915	267 098	162 274	429 372
Tas.	3 877	231 758	23 690	255 448
NT	380	14 198	31 440	45 638
Aust.	129 934	10 084 596	1 061 906	11 146 502

^ estimate has a relative standard error of 10% to less than 25% and should be used with caution

(a) Includes ACT.

Groundwater resources are also an important water source. Major groundwater basins underlie 60% of the continent, with the Great Artesian Basin extending over 22% of

Australia making it one of the largest aquifers in the world (AWRC 1975; DAFF 2006). The distribution of available groundwater resources are shown in Figure 2.2. There are confined and unconfined aquifers, in total providing a sustainable extractive yield of 25,780 GL (NLWRA 2001). Confined aquifers (artesian water) are overlain by an impermeable layer and held under pressure, such that water flows freely from the bore once it is tapped. Unconfined aquifers, on the other hand, do not have confining strata and requires pumping (AWRC 1975). Groundwater makes up 14% as a source of irrigation water for agriculture in Australia, however its use has grown dramatically in recent years as surface flows have become fully committed (Pigram 1986).

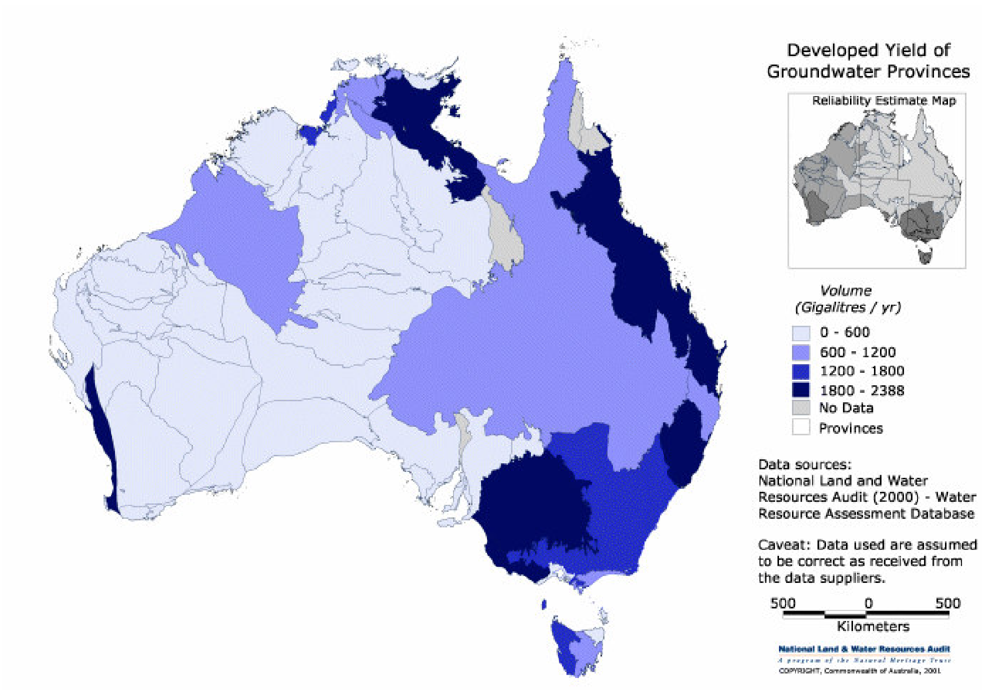

Figure 2.2: Sustainable yield of groundwater provinces (source: NLWRA 2001).

The MDB is the most developed agricultural area in Australia and covers 1million km^2, or 14% of Australia (AWRC 1975). It consists of two major tributaries; the River Murray begins in the Snowy Mountains of NSW which ends in South Australia, and the Darling River extends north through inland NSW which ends across the Queensland border (Figure 2.3).

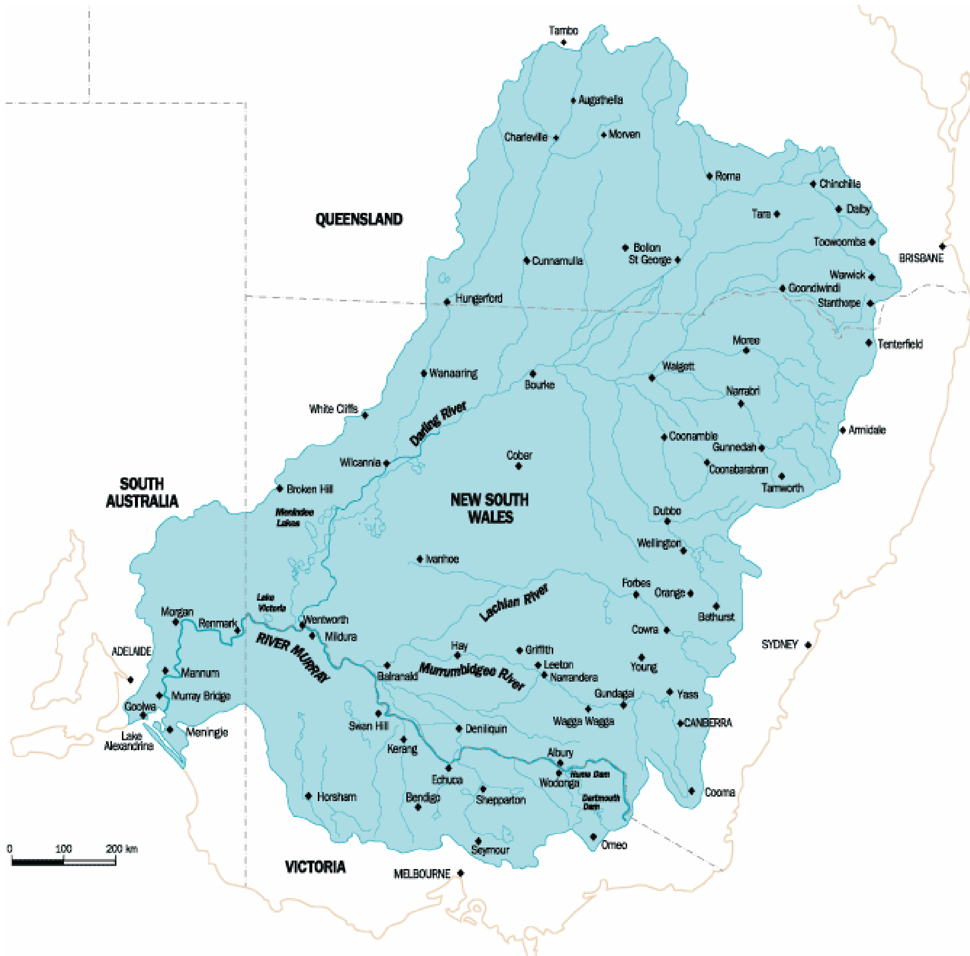

Figure 2.3: Murray-Darling Basin.

Prior to agricultural development, estimated average discharge from the mouth of the Murray was 12,900GL/yr, with flows ranging from 1,600 to 54,200GL/yr (CRCIF 2005). The bulk of in-river flows have been diverted with the expansion of irrigation schemes, with the volume of water diverted for agriculture over the last four decades increasing to over 12,000GL/yr (Figure 2.4). This severely affected flows at the mouth of the Murray, which now range from 0 to 27,464GL/yr (CRCIF 2005). While irrigated agriculture in the MDB contributes to more than 41% of Australia's agricultural profit, it has been argued that the resources would have been better spent on expanding dryland production or other agricultural enterprises, where there is comparative advantage (DAFF 2006; Davidson 1969).

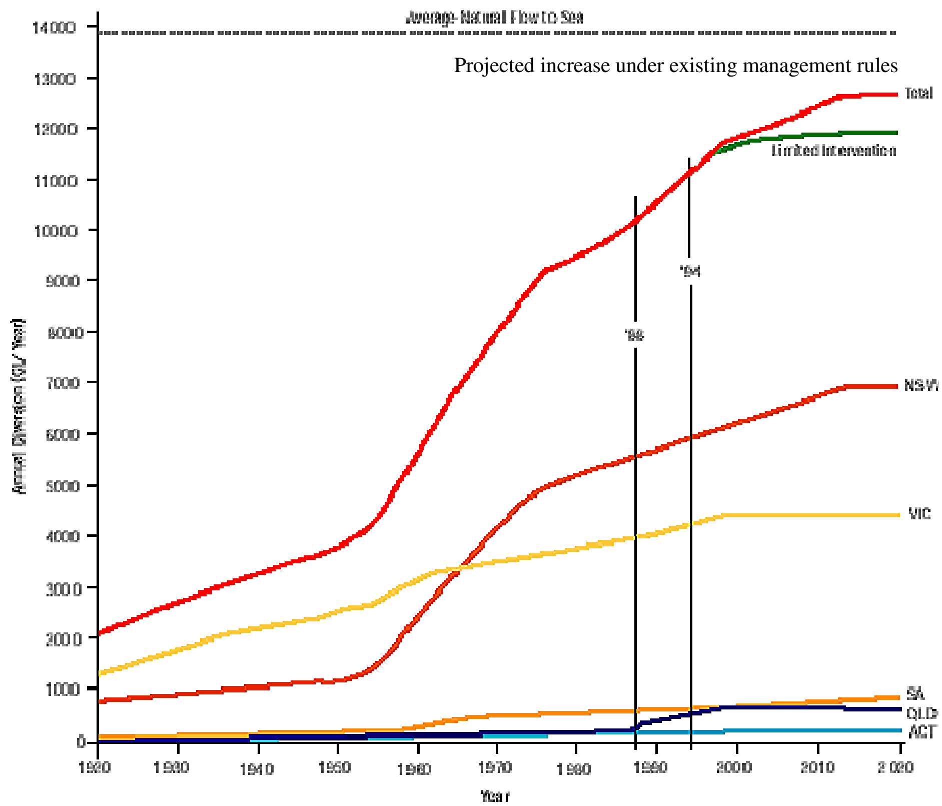

Figure 2.4: Growth in water use in MDB since 1920 (source: MDBC 1995).

2.2 WATER QUALITY AND SALINITY

Due to the variability of surface runoff and discharge, water quality is also extremely variable, especially given the high salinity of Australia's soil relative to other continents. Changes in water quality over time have been largely induced by landuse changes, which has been most pronounced after the introduction of European farming techniques. This includes the replacement of deep rooted vegetation with shallow rooted grasses and annual crops, which disrupts the natural water and salt balance, and the contamination of surface and groundwater from chemical and fertiliser use (MDBC 2001). The intensity of irrigated agriculture in the Murray-Darling Basin (MDB) is thus generally linked to the myriad of environmental problems present in the basin, including land degradation, river salinity, land salinity, water quality problems, and loss of biodiversity (MDBC 2001). The decline in river flow has seen a reduction of 60% of tidal areas which form the habitat for migratory birds, and an influx of exotic fish species, including European Carp, as a result of the decline in water and habitat quality (Oliver 2007; Wong et al. 2007).

The greatest economic consequence of environmental damages is dryland salinity, which is estimated to cost $247 million per year in the MDB (MDBC 2001).

Salinity is caused by natural salts in the soil being brought to the surface by rising underground water-tables. Australia's climate and historical geomorphic processes have led to a naturally high salt load in the soils, as a consequence of relatively saline surface waters (NLWRA 2001). There are two categories of salinity: primary salinity and secondary salinity. Primary salinity refers to the naturally occurring salts stored in the soil or groundwater that is slowly leached down below the root zone or is carried out of the system. Secondary salinity refers to the human-induced mobilisation of salts through land use changes, largely through irrigation and land clearing (Figure 2.5 and Figure 2.6). Dryland salinity is caused by the replacement of native deep-rooted vegetation, perennial trees, shrubs and grasses with annual crops and pastures that use less water, leading to a rise in the water-table. Irrigation induced salinity occurs through the application of irrigation water which percolates through the soil profile recharging groundwater, raising the water-table. Despite the marked differences between these two types, the hydrological process of both types of secondary salinity is fundamentally the same (DEH 2007).

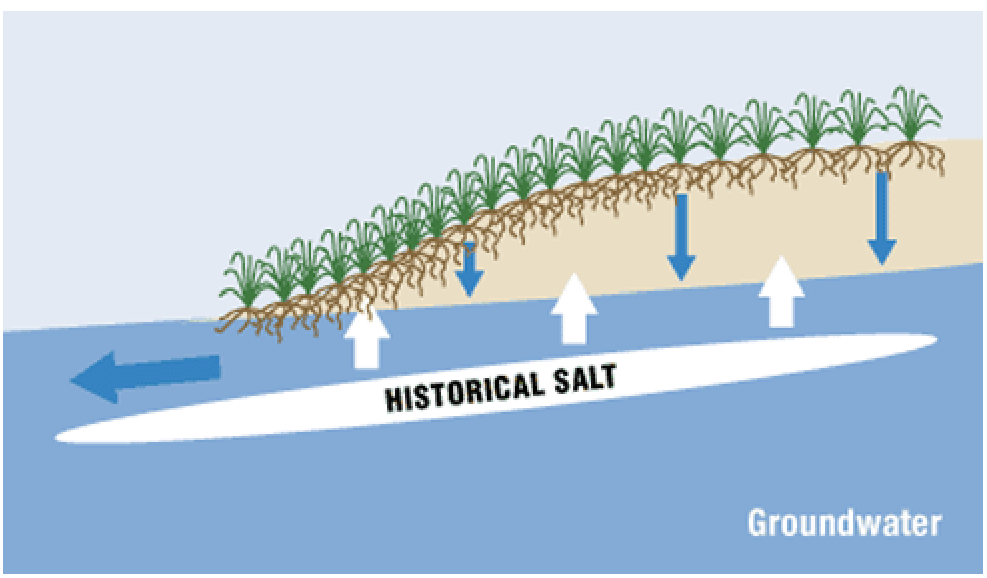

Figure 2.5: Dryland salinity caused by the replacement of deep-rooted natives with annual crops (source: NAPSWQ 2001).

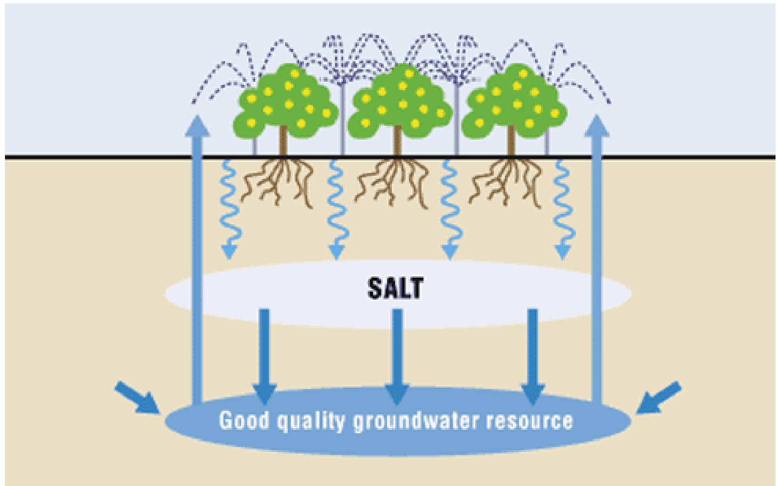

Figure 2.6: Irrigation salinity caused by excessive application of irrigation water (source: NAPSWQ 2001).

Salinity occurrence through time is closely linked to the hydrological functioning of groundwater systems, in particular the way it responds to changing recharge and how it is distributed. It is understood that increase in groundwater tables is a result of high rates of deep drainage (water loss below the root-zone) associated with current farming systems, contributing to groundwater recharge and secondary salinity (Asseng et al. 2003). However, there remains a significant information gap regarding the way the underground hydrological system reacts to altered recharge, due to the expense and non-transferability of studies. Often the response to landuse changes is very slow, taking between 10 to 10,000 years, and once the changes have taken place it takes a long time to achieve a new equilibrium (Jolly et al. 2001). Local flow systems, which have smaller capacity to store additional recharge, is thought to respond more rapidly to landuse changes, while regional flow systems have a large capacity and takes longer time to respond (DEH 2007).

The consequences of increased salinity include vegetation damage, dieback, water logging, saline waters and infrastructure damage. It is estimated that dryland salinity affects 5% of agricultural land, or 2.5 million hectares (NRM 2004). Salinity impact in the form of land and water degradation is thought to cost up to $3 billion per year, and a decline in bird varieties of 50%. Urban salinity damage in buildings and roads is also substantial. In the Namoi and Gwydir region in NSW, dryland salinity costs the households and businesses approximately $11 million per year, and a further $6 million to agricultural producers in terms of infrastructure and lost income (Wilson and Ivey 2001). In South Australia more than $6 million is spent on building maintenance alone (NRM 2004).

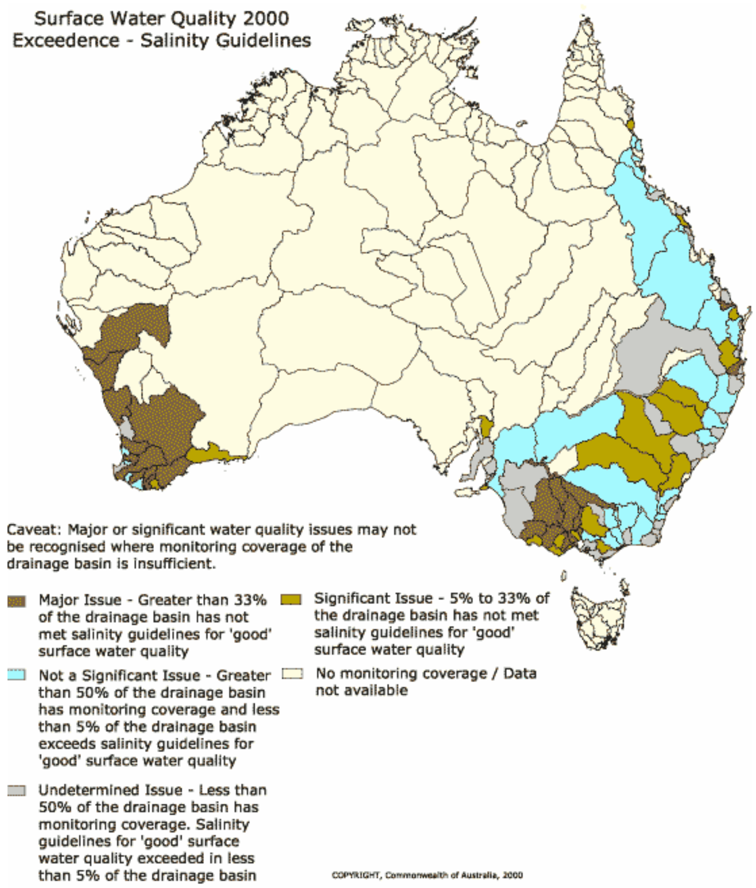

Figure 2.7: Surface water quality guidelines exceedance – salinity (source: NLWRA 2001).

The areas in Australia where salinity presents a problem are shown in Figure 2.8. The salinity afflicted areas are mostly in basins along the South-West Coast, South-East coast and southern MDB. However, some areas which show no signs of salinity problems may have insufficient monitoring coverage to indicate otherwise.

Salinity is measured through the electrical conductivity (EC) of the water. EC is related to the concentration of dissolved salt in the water, which allows electrical currents to flow through; the higher the salt concentration, the higher the EC reading. The level of EC in rivers also depends on the flow rates, where high flows dilute salt concentrations, giving a lower EC reading. However, the actual salt load that is exported can still be high in waters with a low EC, as salt loads are calculated as a multiple of EC measurement and the flow. Salinity can therefore be within stipulated standards while carrying significantly high amounts of salt loads downstream. Furthermore, projections of salinity risk based on historic river salinity can be of limited usefulness in predicting future salinity values due to the non-linear nature of salinity, as well as changes in climatic and production patterns (NLWRA 2001).

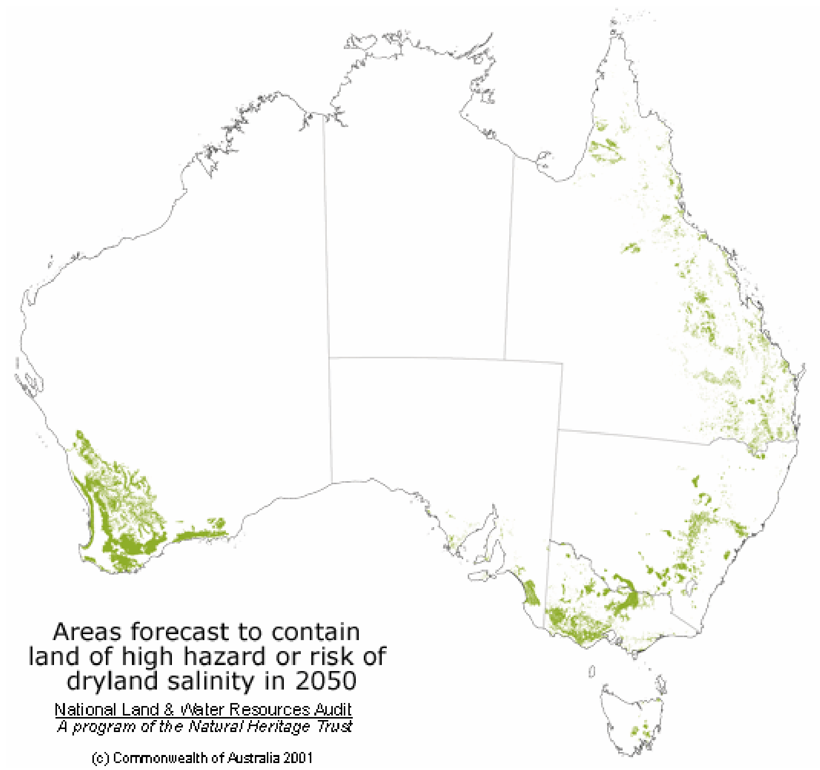

Figure 2.8: Forecast salinity risk areas in 2050 (source: NLWRA 2001).

Nevertheless, salinity risk maps are the best indicators available to determine areas where salinity will become a problem. It is predicted that almost all river basins in the MDB will have increased salinity in the next 50 to 100 years, and a number of national programs have been put in place to manage the salinity issue in Australia. These will be discussed at greater length in Chapter 3.

2.3 HISTORY OF WATER WORKS IN AUSTRALIA

The problem of water use can be attributed to the history of allocation and management of the resource. A first keystone development in Australian water management was the Irrigation Act 1886, formulated based on Alfred Deakin's report in 1884 (which investigated administrative arrangements of the irrigation industry in western America). This report subsequently formed the basis for Australian water laws and the push for the 'nationalisation' of water. This led to the overhaul of the 'prior appropriation' doctrine which gave water rights to individuals on a first-come-first-serve basis, and full ownership of water was handed over to the states. The objective was to productively utilise as much water on the largest possible area (Smith 1998), which reflects the sentiment that water left in the river has zero value. This also marked the beginning of heavy government involvement in water infrastructures in Australia.

By Federation in 1901, demand for irrigation grew with the development of agriculture, and more irrigation projects were commissioned with generous provisions made by the states and later by the Commonwealth. State governments vigorously encouraged the construction of large dams (Smith 1998). The imperative for development during this 'expansionary phase' of the water economy meant that economic considerations were sub-ordinate to the objective of rural expansion (Godden 1997). Large construction works were perceived as relief activity to generate employment, and it was during the period post World War II and the worldwide depression that several large dam projects have been commissioned. The first irrigation project in NSW was the Burrinjuck Dam that was completed in 1912, followed by the Hume Dam in 1936 (Smith 1998). The justification for these heavily subsidised projects was the long term returns generated from increased productivity and population growth (Davidson 1969).

The extent of government subsidisation essentially meant that the irrigation industry would not exist had irrigators been charged the full cost of dam constructions. These include the largest projects of its time, the Snowy River Scheme and the Ord River Dam, constructed between 1949 and 1974. These projects were subject to little or no economic assessments, and continued even where the economic and technical infeasibility had been known, due to ulterior motives including national prestige and political expedience. The cost of construction and operation had never been recouped, much like other schemes in this era, and irrigators only had to pay maintenance costs (Godden 1997).

The outcome of this ‘developmental phase’ was a substantial public debt on a number of these water infrastructures, which also deterred governments from financing further investments into its upkeep (Quiggin 2001). It was not until the 1970s that emphasis shifted to the management of water, as Australia’s water economy shifted from its developmental to mature phase (Freebairn 2005). Furthermore, increasing environmental awareness meant that large-scale dam constructions are no longer favourable, with the damming of Lake Pedder in 1974 driving the green movement in Australia. The economic infeasibility of these dams was formally recognised with the 1992 Industry Commission report, that ‘much of the past public investment in irrigation would not have proceeded had irrigators been required to meet full costs, including capital costs’ (Industry Commission 1992 p. 85). The focus has since shifted towards the efficient management of currently developed water supplies to meet the ever growing demand, with the cost of dam operation being addressed through the cost-recovery process.

2.4 MATURE WATER ECONOMY

A mature water economy refers to the intensifying competition for water between users and uses, amid increasing social costs of securing more water supplies (Randall 1981a). For Australia, this comes as the cheapest and easiest sites for harnessing water resources were fully exploited during the national development phase, ending in the 1980s (Smith 1998). During the expansionary phase, the welfare cost of subsidies to water use was relatively small due to the lower social costs of expanded water use. However, as the water economy shifts into a mature phase, the management of water resources is complicated by policies inherited from the expansionary phase (Quiggin 2001). Most of the water resources in Australia have now become over-committed, with the paradigm now being the efficient reallocation of developed water supplies rather than the capturing of new water sources (Randall 1981a). The beginning of the mature phase can be regarded as the time the intergovernmental Murray-Darling Basin Agreement and the

Cap was introduced (Quiggin 2001). This involves the reallocation of water away from low-value uses to where it has higher value. The formation of water markets and pricing policies are crux to this reallocation, to facilitate the process of increasing water use efficiency.

Problems pertaining to water scarcity and allocation derive from early government policies to provide water to all users regardless of the cost, helped by the doctrine that 'water left in the river is wasted' as no value has been ascribed to its environmental significance. Cheap and heavily subsidised water had been supplied to urban and agricultural users through a string of dam constructions paid for largely by the public sector. This subsequently led to an overinvestment in irrigated cropping, accompanied by an underinvestment in water efficient technologies, and a gradual shift from dryland to irrigation farming including rice, cotton, and horticulture (Godden 1997; Arthington 1996).

The over-allocation of water entitlements has led to a situation of inefficiency, where the volume of water extractions far exceeds the level at which economic efficiency would be achieved. Such level would occur where the true opportunity cost of using water equates at the margin with the price of water. Where there are unpriced adverse or beneficial effects, the social cost diverge from resource and/or opportunity cost, which underpins the problem of water allocations because its price has not been cost-reflective. This is an artefact of initial water licence allocations which were originally area-based; such that licence holders were confined only by the area irrigated and the marginal cost to users was close to zero (Freebairn 2005). Hence, no explicit value had been tied to each unit of water; only a nominal licence cost and annual water fee which only captured a small percentage of operating and maintenance costs for distributing water from the main storages (Godden 1997). By the 1960s most licences were converted to volume-based licences to limit extractions due to escalating demands. However, the level of surface and groundwater extraction had persisted at unsustainable levels, leading to substantial environmental degradation in the river systems (PC 2003).

It was soon recognised that an allocation needs to be set aside for the environment as a legitimate user, and some reduction in the level of extractions is necessary. A significant turning point was the 1994 COAG statement, which proposed that the ownership of land and water be separated and that environmental needs for water be recognised (COAG 1994). The COAG agreements in effect reinforced the preference for price-based and market-based solutions to environmental problems (Quiggin 2001), which proposed for the capping of extraction levels; clawing back entitlements through government buy-back; and through cost-recovery, by incorporating the true opportunity costs into water charges. Extensions to the reforms in 2002 and 2004 highlighted the need for secure water rights to encourage water trade and a cost-sharing framework for reductions in entitlement (Freebairn 2005). Some irrigator groups in cotton regions have foreshadowed such problems arising from over-allocation of irrigation entitlements as early as the 1970s, and called for more sustainable development of water resources. In the Namoi Valley, this led to a voluntary embargo by cotton irrigators on issuing further groundwater licences in 1983 (Hamparsum, 2004). This was in contrast to the proposal by the NSW Water Resources Commission in 1983, to "mine the [groundwater] resource over 30 years, not allowing for recharge" in the Namoi Valley (Commonwealth of Australia 2003, p.p. 15014). It was also claimed that irrigators were told by authorities to activate their irrigation licences or they would be lost in conversion to volumetric licences because it was done according to history of use (Anthony, 2003), although there has been no official record of such advice (Jobling, 2000). Nevertheless, it appears the opportunity to acquire low-value entitlements at relatively low cost, has been lost.

2.5 THE CAP AND THE WATER MARKET

The low price paid for water means that there has been a lack of incentive to utilise water in a manner that corresponds to its true opportunity cost. This has led to an over-extraction of water and underinvestment in water efficient technologies (Zilberman et al. 1991). To remedy the situation of excessive wateruse, the Cap was introduced in 1995, following the completion of a water audit of diversions from rivers within the MDB. This audit had shown water diversions averaging at 10,800GL/yr and increased by 8% over the preceding six years. The Cap varies from state to state. For NSW and Victoria, the

Cap is the "volume of water that would have been diverted under 1993/94 levels of development" (MDBC 2003 p. 4). This volume also varies within the state, depending on the catchment in question. However, NSW has been lagging in the process of defining the Cap for some valleys, specifically the Border Rivers in northern NSW (MDBC 2007b). South Australia has been the most progressive with implementation of its Cap, having not issued any new allocations since the early 1960s. It was also the first state to sever water property rights from land in 1997 and allowed for licences to be temporarily traded (MDBC 2003).

The purpose of imposing a Cap on the level of diversions for consumptive uses was to limit the over-extraction of water and provide water for environmental services (MDBC 2003). The capping effectively increased surface water scarcity, resulting in an increase in demand for irrigation water and the need for an effective water market to be developed (Heaney et al. 2002). The market that is established has enabled trade to occur and an explicit price to be placed on water. Irrigators have the option to purchase water where allocations are insufficient to meet crop requirements; alternatively they can receive revenue from the sale of water allocations. This should reveal the true value of water, such that its opportunity cost is reflected in the market prices and encourages efficient use, (Randall 1981a). A properly functioning water market is thus a conduit to efficient allocation of scarce water resources, because it allows water to be diverted from lower value uses to higher value uses.

However, the price of water remains difficult to estimate due to administrative impediments and the existence of a relatively 'thin' market, whereby there are few participants in the market to create competitive conditions of trade. Trade in permanent water entitlements has been less than 1% of diversions in 2001-02, with less than 1% of all trade occurring inter-regionally (Heaney et al. 2004). There has been a general apprehension towards the water market, drawing from concerns pertaining to community decline, threat of foreign ownership, and a perceived loss of subsidies (Randall 1981a). Crase and Jackson (1998) (in Crase et al. 2000) reported that only 2% of farmers in the Murray region would consider permanently selling their water entitlements. Gaffney

(1997) also reports the lack of motivation for sellers to participate in trade as a significant hindrance, with the only participating irrigators being those intending to exit the industry or facing financial debt. The perception that demand will grow perpetually is another reason irrigators withhold from trade. This appears to be the sentiment of many irrigators in the Namoi, with the expectation that if the entitlement is sold then they will need to buy it back in future at higher prices. The preference is therefore to invest in water efficient technologies rather than buying more water to augment supply (Morgan 2005, pers. comm.). Skepticism is also expressed, based on 'philosophical' grounds that water should not be traded like a commodity (Norrie 2006, pers. comm.). This view reflects the lack of understanding and acceptance of water trading, even on a temporary basis.

Complexities including the interconnectivity between surface and groundwater, and the trade of water into salinity-prone regions exacerbating environmental problems, have further prevented the full benefit from trade to be realised (Young and McColl 2005). This led to the conjecture that the costs involved with creating perfectly defined water access right to be greater than the potential benefits that would be derived from trade (Beare et al. 2001). Bennett (2005b) also cautions against the promotion of one institutional arrangement over another, e.g. market-based approach over 'command and control', without consideration for the transaction costs involved and its distribution amongst affected parties. The benefit that market instruments could provide should be assessed on a case-specific basis to determine its virtue.

2.6 COST RECOVERY

The pricing mechanism has an important role in achieving efficient resource use, by providing coordinating market signals for its distribution (Randall 1981b). However, the price of water charged to irrigation licence holders has for a long time reflected a relatively small portion of the state's costs of running the irrigation infrastructure and had little account to the future availability of the resource. Government policies in the past have perceived water security as a priority, and many schemes were put in place to ensure water is delivered to meet demand. Regional development was emphasised as a justification for irrigation schemes that were economically unjustifiable (Davison 1969).

In turn, the provision of water has imposed significant costs on the authorities managing the infrastructure, since the price of water never reflected the true cost of supply. There was also public concern over the impacts underpricing is having on the environment, since environmental costs have generally been overlooked in decisions to allocate water for irrigation (Godden 1997). The economic rationale to water pricing for public irrigation schemes in Australia has been to recover only annual operating costs, which are relatively low (Godden 1997). Current cost-recovery processes, whereby higher water charges are being implemented for regulated river systems, aim to correct the over-exploitation of river systems and the inefficient use of scarce water resources. By increasing the private costs of using water for production, the external cost of over-extraction is minimised if prices charged reflect more closely to the true social cost of water extraction (IPART 2004). It is also important for further investment to rebuild run down infrastructure, which the government has been reluctant to do because of its history of unsuccessful investments in water infrastructure (Smith 2000).

Cost-recovery plans for water services have been recently implemented in NSW to address these concerns. In the cost-recovery process, the level of licence fees and water charges to all users are revised upwards each year to recoup current costs incurred by regulators (DLWC 2001). This is in accordance with the COAG 1994 Water Reform Framework to stem natural resource degradation caused by the under-pricing of bulk water services (COAG 1994). A medium term price path was proposed in a submission by the Department of Land and Water Conservation (now Department of Natural Resources), whereby the fixed charge and usage charge for irrigation water was set to rise by 20% each year from July 2001 to June 2004 (DLWC 2001). It was envisaged to allow an 86% level of cost recovery of the expense of running water services, compared to 54% under earlier pricing arrangements. Some valleys only required only small increases in regulated water charges to achieve full cost recovery, while other valleys require very large increases in both regulated and unregulated water charges (IPART 2001).

The pricing structure as set out by the NSW Independent Pricing and Regulatory Tribunal (IPART) is such that bulk water prices are in the form of a two-part tariff (Table 2-2).

This major change was made in 1997/1998, with a fixed charge on licence entitlement and a variable charge on water use. The two-part structure is intended to provide financial viability for the operation of dams and also to provide a conservation signal to users (IPART 2004). However, this price structure had only applied for regulated rivers, not unregulated systems (rivers without an up-stream head dam to control downstream releases of water), where irrigators were still charged on a per-hectare basis. In unregulated systems, a two-part tariff does not apply until area-based licences have been converted to volumetric licences (DIPNR 2004a). This had not occurred for many unregulated systems until Water Sharing Plans were gazetted in 2004, with the original pricing system in place until proper metering and monitoring is implemented to allow accurate determination of water extraction (DIPNR 2004a; Hudson 2005, pers. comm.).

Table 2-2: Cost recovery level for river systems in NSW (source: IPART 2001 p. 40).

<table>
<tr><th></th><th>Regulated Water</th><th>Unregulated Water</th><th>Ground Water</th></tr>
<tr><td>Border</td><td>83%</td><td>26%</td><td rowspan="2">Barwon region</td></tr>
<tr><td>Gwydir</td><td>87%</td><td>53%</td></tr>
<tr><td>Namoi</td><td>81%</td><td>28%</td><td rowspan="2">22%</td></tr>
<tr><td>Peel</td><td>44%</td><td>Included in Namoi</td></tr>
<tr><td>Lachlan</td><td>83%</td><td>17%</td><td rowspan="2">Central West
21%</td></tr>
<tr><td>Macquarie</td><td>116%</td><td>43%</td></tr>
<tr><td>Far West</td><td>No regulated rivers</td><td>20%</td><td>21%</td></tr>
<tr><td>Murray</td><td>82%</td><td>20%</td><td>34%</td></tr>
<tr><td>Murrumbidgee</td><td>91%</td><td>43%</td><td>17%</td></tr>
<tr><td>North Coast</td><td>7%</td><td>13%</td><td>16%</td></tr>
<tr><td>Hunter</td><td>36%</td><td>19%</td><td>15%</td></tr>
<tr><td>South Coast</td><td>24%</td><td>13%</td><td>6%</td></tr>
<tr><td>Total</td><td>81%</td><td>19%</td><td>20%</td></tr>
</table>

2.7 WATER ENTITLEMENTS AND SEASONAL ALLOCATIONS IN NSW

The water entitlement in NSW can be grouped according to its supply reliabilities. The highest priority is given to local and major urban water utilities, followed by high security entitlement holders, with general security licence holders allocated a share component only after other higher priority users' requirements, or 'fixed commitments', have been met first. These include environmental provisions and high security supply commitments. High security licence holders are guaranteed to receive full allocation in all but the worst drought years, and two years worth of water supply is reserved in dams to meet the demands of high security users. Each licence type is subject to a different set of prices, reflecting the degree of water security. The assurance of water supply is reflected in higher fixed and variable usage charges for high security licence holders than general security users (PC 2003).

Around 90% of NSW entitlements are general security, while perennial crops are mostly under high security licences (PC 2004). The volume of water available for extraction is announced at the start of each season by the water authority, known as the Available Water Determination (AWD), which is then apportioned to licence holders within the water district. AWD announcements can be made throughout the year for general security licences if more water becomes available (DIPNR 2004c). All NSW water access licences are kept in NSW Department of Lands Water Access Licence (WAL) Register, and Department of Natural Resources (formerly DIPNR) forwards any changes to access licences, such as annual AWD credits, to the Department of Lands for updating (DIPNR 2004c). The seasonal allocation given to each licence holder is a percentage of the Share Component specified in the entitlement (Figure 2.9).

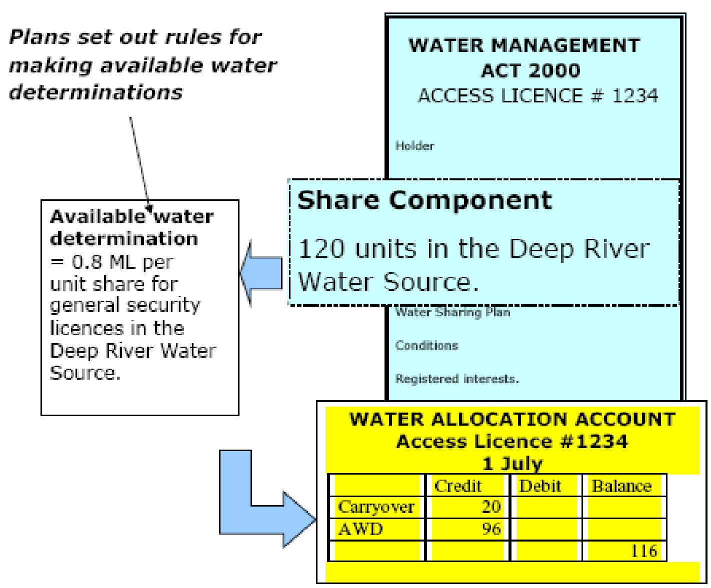

Figure 2.9: Allocation of a season's water to licence holders' accounts (source: DIPNR 2004c).

The AWD credits can be accumulated and used across seasons, as part of the carry-over rules. These rules have been introduced so that irrigators can carry-over part of the unused seasonal allocation into the following year, providing an incentive to be water efficient. The maximum volume that can be carried over from one water-year to the next is 100% of the share component, with an allowable 200% extraction in any one year and a maximum of 300% accessed over three rolling years (DIPNR 2004b). That is, in any one year the maximum extraction permitted is the two-year water allocation, whereas the extraction limit cannot exceed allocations received over a rolling-three years. Irrigators could then choose to bank some water for the following season or use more in the current production. The objective of introducing carry over rules was to provide greater flexibility and an incentive for conservation, rather than to exhaust all allocations within one year.

2.8 SUMMARY OF CHAPTER 2

The evolution of Australia's water economy was presented in this chapter, which was shown to be linked to geographical variability in water resources in Australia. The environmental problems that have arisen from the over-extraction of water resources, particularly salinity, were discussed. This has led to a number of government initiatives which aimed at creating a sustainable level of resource use and were discussed in brief. The next chapter will discuss the trend in water management in Australia in detail, and how it has been implemented through the various institutional arrangements.

Chapter 3.

INSTITUTIONAL ASPECTS

In this chapter, an appraisal of the various intergovernmental arrangements in Australia's water economy is conducted. The objective is to provide an understanding of how water management principles have been implemented in practice, and the range of catchment policies that affect irrigators in Australia and in NSW.

3.1 RIVER MURRAY WATERS AGREEMENT AND THE MURRAY-DARLING BASIN AGREEMENT

The earliest intergovernmental agreement in Australia was the River Murray Waters Agreement (RMWA) in 1915. The agreement was made between the Commonwealth, New South Wales, South Australia and Victoria, and marks the beginning of intergovernmental cooperation over water resources. The issues were initially related to securing a share of water supplies, with the stimulus being the severe drought from 1895 to 1902 which had prompted the colonies and States to forge an agreement to manage the Murray River. This Agreement was especially significant for South Australia, because it guaranteed the State a minimum flow. Various amendments to the RWMA were made over the 70 years of its operation, although these were only minor changes relating to the construction of dams and weirs (MDBC 2006a).

Over time, environmental damage in the basin became more noticeable. However, it was not until 1984 that a more concerted effort was made to addresses the resource and environmental problems that were increasingly prevalent in the basin. It was eventually recognised that problems with rising water and land salinisation, amongst other natural resource issues in the basin, extended across borders and collaboration between the jurisdictions was required to sufficiently manage these problems. The outcome was the Murray-Darling Basin Agreement, signed by the governments of NSW, Victoria, and South Australia in 1992 and legally enforceable under the Murray-Darling Basin Act 1993. By 1998, Queensland and the ACT both became signatories to the agreement. The

Agreement also saw the creation of the Murray-Darling Basin Commission and the Community Advisory Committee (CAC), responsible for the coordination of planning and development of policy pertaining to the sustainable and equitable use of the Basin's resources (MDBC 2005b). The transition of the water economy from the developmental phase to the mature phase is reflected by the change in focus in water management doctrine during this period. The objective was no longer to use as much water as possible, but to achieve sustainable use between human use and environmental protection through a coordinated approach. The MDB Agreement is enforceable by legislation, and was signed in the same year when the Council of Australian Governments (COAG) was formed in 1992 (COAG 2005a). The formation of the COAG led to a succession of water reforms in Australia, with the environment playing an increasingly significant role at each stage.

3.2 COUNCIL OF AUSTRALIAN GOVERNMENTS AND THE MURRAY-DARLING BASIN COMMISSION MINISTERIAL COUNCIL

The Council of Australian Governments (COAG) comprises the Prime Minister, State Premiers, Territory Chief Ministers and the President of the Australian Local Government Association. Its primary function is the development and execution of policy reforms, significantly the implementation of the National Competition Policy (NCP) and water reform. Commonwealth-State Ministerial Councils are formed under COAG as facilitating bodies for joint action between Contracting Governments to develop, coordinate, and monitor policy reform under consideration by the Council. Over 40 Ministerial Councils have been established under COAG, including the Murray-Darling River Ministerial Council which was created as part of the Murray-Darling Basin Agreement (COAG 2005b). These bodies are responsible for the implementation of the Murray-Darling Basin Initiative (MDBC 2001). The Commission is the executive arm of the Ministerial Council, and is an autonomous body separate from government organizations or departments.

The Commission develops and implements policies by working cooperatively with the partner governments, committees, and community groups, with the aim to devise integrated management of the Murray-Darling Basin. The Community Advisory Council (CAC), also created under the Agreement, comprise of people with varying areas of expertise and with networks throughout the Basin. Its role is to provide the Council with advice from a community viewpoint on natural resource management issues, and to enhance the adoption of management strategies. The CAC has had an active role in the establishment of salinity management programs, integrated catchment management, the establishment of the Cap on diversions, and environmental flows (MDBC 2001).

The Murray-Darling Basin Agreement, enforceable by the Murray-Darling Basin Act, contains a section specifying the requirement for States to implement a Cap on diversions. Each Contracting Government is bound by the Murray-Darling Basin Agreement to comply with annual diversion targets set out by the Ministerial Council (MDBC 2004). To hasten the progress of the Agreement, the implementation of the Cap, as a part of greater water reform, has been tied in with the NCP which provide financial incentives to motivate its implementation. The NCP payments were the first of a string of Federal funding aimed at promoting full water reform in the years to come.

3.3 COUNCIL OF AUSTRALIAN GOVERNMENTS WATER REFORM AND THE NATIONAL COMPETITION POLICY

The water reform process has been underway since early 1990s, as part of a wider micro-economic reform agenda towards liberalised national competition. It was recognised during this phase that water extraction for consumptive uses in the Murray-Darling Basin is exceeding sustainable levels. Water reform subsequently became a key issue in the 1994 Council of Australian Governments (COAG) meeting during which the Contracting Governments agreed to a Water Reform Framework for water resources, to implement a "strategic framework to achieve an efficient and sustainable water industry" (COAG 1994). This agreement was based on the Working Group on Water Resources Policy commissioned in 1993 which concluded that the water industry had significant inefficiencies and was unsustainable. The Working Group's recommendations for the reform process significantly advocated the separation of water property rights from land title, allocation of water to the environment as a legitimate user, putting in place necessary institutional arrangements to facilitate water trade, and the need for full-cost recovery of water delivery costs and assets (Smith 2000). The key accomplishment from the Water Reform Framework has been the incorporation of environmental considerations in water use decisions. Water management plans have been developed by governments to provide for ecological flows, and an embargo placed on new water allocations from stressed water bodies.

The process of water reform is stimulated by the endorsement of the National Competition Policy (NCP) by COAG in 1995. NCP payments were tied into the framework to accelerate the rate of water industry reform and to facilitate structural adjustment in the process, not just for water but for structural reforms in other industries. This agreement evolved from the governments' microeconomic reform imperative over the past decade which aimed to reduce barriers to free trade in all areas of the economy. The reform agendas were based on the 1992 Hilmer inquiry into a National Competition Policy for Australia, which provided recommendations for the direction of reform. Significantly, the report advised the implementation of competitive neutrality principles to reduce the competitive advantage inherent to publicly owned enterprises and to revise legislation inhibiting competition in key industries. With respect to water, the reform was to allow for more sustainable water use through efficient pricing, which reflects the true cost of water delivery and thereby improves conservation signals to users (NCC 1998).

An Implementation Agreement was drawn up under the NCP, where the Commonwealth Government made available NCP payments to each State and Territory on condition that satisfactory progress was made in the direction of recommended reforms agreed to in the NCP. Each government reports annually to the National Competition Council (NCC) on progress made with implementing the agreed reforms. An assessment of each State and Territory's progress is made by the NCC prior to 1 July of the years 1997, 1999, and 2000 to determine whether the jurisdictions had met conditions upon which payments were to be made (NCC 1998).

However, despite the NCP payments, progress since have varied between the states. While the intergovernmental Murray-Darling Basin Agreement and 1994 COAG Agreement has set the foundation for a basin-wide perspective on water management, disputes remain between upstream and downstream states. There has been regular blame-shifting and under-resourcing by contracting governments, which has hindered the implementation of full water reform (Howard 2007). Additional funding from the Federal government have since been made out to the States, and further agreements to address the impediments to full water reform have been initiated. This includes the National Water Initiative in 2004, essentially an enhancement of its predecessor the 1994 COAG Agreement, and was a keystone event in the water reform history.

3.4 NATIONAL WATER INITIATIVE

At the 2002 COAG meetings, it was agreed that an update of the 1994 Water Reform Framework was needed to overcome obstacles in the reform process, in particular for clearly defined property rights in order to mediate the uncertainty for water users (Freebairn 2005). The crux of the problem was the determination of environmental flow requirements and allocations, stemming from the lack of clearly specified water property rights between extractive users and ecological services. A report from the Chief Executive Officers' Group on Water was presented in the December 2002 COAG meeting, suggesting national principles on water allocation and entitlements. The principles outlined in the report, and after extensive consultation with key stakeholders, subsequently provided the basis for the National Water Initiative (NWI) (NSW Farmers' Association 2003). The purpose of the NWI was to establish a nationally compatible system of water access entitlements, efficient water markets, institutional arrangement for the recovery and management of water for the environment, improved accounting and best practice water pricing, and urban water issues. The key objectives of the NWI include:

- The encouragement of permanent trading and more cost-effective water recovery to achieve environmental outcomes;

- To improve public access to information, more compatible registry arrangements, monitoring, reporting and accounting of water use.;
- More comprehensive and transparent water planning;
- To address over-allocated systems as quickly as possible (COAG 2004).

The NWI was endorsed at the June 2004 COAG meeting, and is regarded as a landmark agreement in which all stakeholders agreed to a framework to meet the objectives of both consumptive and environmental use (Borthwick 2006). The Natural Resource Management Ministerial Council oversees the implementation of NWI objectives by each State and Territory, although the assessment of progress made with NWI is undertaken by the National Water Commission (NWC), created at the June 2004 meeting with the endorsement of the NWI. The 2005 assessment of compliance with water reform commitments under the National Competition Policy were also carried out by the NWC (COAG 2004).

Another key responsibility of the NWC is the allocation of the Australian Water Fund (AWF) set up by the Prime Minister in December 2004, for which further funding was made available by the Federal government. A commitment of $2 billion over five years was to be allocated amongst three programs under the umbrella of AWF: Water Smart Australia, Raising National Water Standards, and Australian Water Fund Communities. Funds allocated to activities under the Water Smart Australia programme are for encouraging the adoption of 'smart technologies' and better management of water use at the farm level. The Raising National Water Standards Program invests funds to increase the ability to monitor and measure water use, and the Community Water Grants program provides grants to communities to promote efficient water use. These programs are jointly administered by the Department of the Environment and Water Resources (formerly Department of Environmental and Heritage), and the Agriculture, Fisheries and Forestry departments (DPMC 2006). Further to the NWI agreement was the development of the Living Murray Initiative, to target and prioritise the ecological assets in the MDB that are to be protected. The contracting governments to COAG have pledged significant funding towards achieving these targets, although it appears that progress has been lagging.

A common theme that has emerged throughout the course of these agreements is the lack of clear, measurable objectives associated with the funding that have been injected at different stages of the water reform process. This is a trend that is also observed in other intergovernmental strategies carried out by the Murray-Darling Basin Commission.

3.5 MURRAY-DARLING BASIN COMMISSION: LIVING MURRAY INITIATIVE AND FIRST STEP AGREEMENT

The Murray-Darling Basin Commission (MDBC) created The Living Murray program in mid 2002 in response to the degradation of the River Murray system. Part of the initiative was the First Step program, developed in 2003 and identifies six significant forests and lakes in the Murray Darling Basin – the six significant ecological assets – to be protected in the five years following the creation of The Living Murray initiative (MDBC 2005b). The implementation of The Living Murray First Step is enforceable through the Murray-Darling Basin Water Agreement, endorsed as part of the National Water Initiative (NWI) signed at the June 2004 COAG meeting. The contracting governments agreed to commit $500 million over five years, commencing 2004-2005, for the implementation of actions specified in the Living Murray First Step, including the recovery of (on average) 500GL of water per year (DPMC 2006). This volume is thought to provide one-third of the water the river needs (Sexton 2006). Complementary to this funding, a total of $150 million has also been set aside by the MDBC in 2002 to be spent over eight-years (2003-2011) on the Environmental Works and Measures Program, targeting six significant environmental assets in the Murray-Darling Basin (MDBC 2005b). Although the funding is substantial, the figures appear to be chosen out of expediency rather than well justified on economic grounds. It was soon apparent that the initiatives lack clearly defined deliverables to rationalise the sums allocated, and inevitably led to more criticism regarding the governments' slow progress.

The Commonwealth's perceived solution was to make a one-off payment of $500 million to the MDBC to hasten the process. This sum would be spent over five years from 2006 to accelerate the implementation of Living Murray Initiatives agreed by the MDBC in 2003 (COAG 2006). Almost $250 million of the lump sum payment will be used for capital works and improvements in infrastructure, including upgrading weirs, enclosing canals, and building fish ways. A further expenditure is to increase flows directly by purchasing entitlements off irrigators who have achieved water savings (Sexton 2006). Altogether, the agreements amount to a significant sum, however it remains short of defined, measurable outcomes associated with each initiative.

Little has been achieved since the Living Murray program commenced in 2003, and only a few water-saving projects have been undertaken (Sexton 2006). The funding committed to the NWI and Living Murray Initiative has exceeded $1 billion since 2004 without corresponding accomplishments that reflect this expenditure. Projects under the Living Murray are mostly still under development or investigation, and the recovery of an average 500GL per year is not expected to be realised till 2009. The projects that have been implemented to date altogether provide just 270GL, through 'market-based' solutions, buying back water entitlements from irrigators, or infrastructure projects such as improved piping (MDBC 2007a). The implementation of the Cap has also been delayed in three of the contracting states, some of which are not expected to be finalised till 2008 (MDBC 2007b). Furthermore, it remains uncertain what the economic implications of the water reallocation targets are for irrigators, and whether the revenue allocated towards this end are justified or insufficient.

3.6 NATIONAL PLAN FOR WATER SECURITY

In January 2007, the Federal Government announced yet another initiative to 'fix' the water problem of the nation. The strategies outlined in the National Water Plan were in accordance with the objectives outlined in the NWI, specifically to address over-allocation, modernise irrigation, to improve water information availability and to create a transparent water management regime. This involves an allocation of $10 billion over 10 years, with the majority of the expenditure dedicated to advancing work carried out under the Living Murray Initiative and the Australian Water Fund. The breakdown of the funding is as follows:

- $6 billion for improving water use efficiency on-farm and off-farm, with $1.5 billion allocated towards subsidising the cost of water efficient technologies and the rest used to improving delivery system efficiency and monitoring;
- $3 billion for structural adjustment, to retire unviable or inefficient irrigation areas and to buy-back water entitlements;
- $600 million to reconfigure the governance of the MDB, by transferring all jurisdictional power over the system from the State Governments to a single minister under the Commonwealth Government;
- $480 million dedicated to improving the accuracy and availability of water data. This involves the creation of comprehensive, transparent water accounts and metering on a national scale. These functions will be carried out by the Bureau of Meteorology.

This is a substantial amount of funding that has both positive and arguably counterproductive features. A small, but important, element of the plan is the gathering of water information. It requires that irrigators "share all of their existing water data assets with the Bureau [of Meteorology] and transfer all new data to the Bureau as it is collected" (Howard 2007, p. 17). Under current arrangements, even where water metering is in place, water extraction data is considered confidential information and is not publicly disclosed (Hudson, pers. comm. 2005). The new arrangement would require irrigators to disclose such information to a government department, allowing for better water management through accurate monitoring of water use on a national basis. This is envisaged to significantly improve the ability to manage Australia's water resources.

On the other hand, provisions in the Plan to improve on-farm water use efficiency, in the form of a subsidy for modern irrigation technologies, may conflict with the objective to retire inefficient irrigation areas. The financial assistance for water efficient technologies would allow less efficient irrigators to remain in the industry, and to use the water savings to expand irrigated production (Ancev and Vervoort 2007). In this sense, the value of such properties become inflated and unnecessarily increases the cost of structural adjustment or the buy-back of water entitlements.

It can be seen that sustainable water use and environmental protection has gradually become the focus of discussion throughout the course of water reforms over the last two decades. While generous amounts of government funding has been put towards achieving ‘sustainability’ during this period, the ‘efficient’ allocation of water between extractive and ecological uses has remained a point of contention. In particular there is significant uncertainty regarding the value ascribed to the environment that is required to justify the cost of water reallocation.

3.7 BASIN SALINITY MANAGEMENT STRATEGY

Other existing intergovernmental agreements surround the management of water quality, significantly salinity. One such agreement is the Basin Salinity Management Strategy (BSMS), an initiative by the Murray-Darling Basin Ministerial Council that began in 2001. The salinity management framework set out in the BSMS is associated with the National Action Plan for Salinity and Water Quality, endorsed by the COAG in 2000. The National Action Plan identified nine priority regions in the Basin that are at risk of salinity, and are targeted by strategies set out in the BSMS. One key feature is the end-of-valley salinity targets for each tributary catchment and a Basin target at Morgan in South Australia. The objective is to maintain the salinity at Morgan at less than 800[1] EC units for 95% of the time, with the overarching aim to halt environmental degradation and for safe human use (MDBC 2006b). The end-of-valley targets set for catchments in the MDB were determined based on this objective. Essentially, the salinity level will be capped to limit the salt load entering the basin, in particular to protect the assets at risk of salinity that have been identified in the National Action Plan. The projects to achieve end-of-valley targets in each tributary include a mix of land management, engineering and river flow options, the choice of which depend on the State and catchment community priorities.

[1] 800 EC units is the recommended safety limit for human consumption.

For the Namoi River Basin, where the case study area of the Mooki catchment is situated, the mean salt load target is 127,600t/yr and a mean salinity level of 440μS/cm (MDBC 2005a). This makes Namoi the third highest contributor to the salt load received in the MDB, carried down-stream into the Barwon-Darling system which flows into the Darling River. In fact, the three largest contributors of salt load to the MDB are all located in NSW (Lachlan, Murrumbidgee and Namoi valleys). There is a strong justification to manage the salt load and salinity levels in these catchments in NSW.

3.8 WATER ACTS IN NSW

State legislation relating to water management varies between states. In NSW, water resources are regulated by the Water Management Act (WMA) 2000, which replaced its predecessors Water Act 1912 and the Water Administration Act 1986. The intention of water law was originally for administrative and legal arrangements over the use of water, which is subject to steadily increasing competing demands (Smith 2000). Licences were originally area based, however in 1981 licences were redefined to limit the volume that licence holders could access due to escalating demands, with volumetric limits implemented first in the River Murray and soon after in other systems (PC 2003). However, the level of surface and groundwater extraction continued to escalate to unsustainable levels in much of NSW, and led to substantial environmental degradation in the river systems. Following the 1994 COAG agreement, significant water reform took place in NSW including setting sustainable access limits as part of the WMA. This subsequently led to the current definition and access rules for water licences in NSW (Smith 2000).

The WMA stipulated the creation of a registry for the administration of water licences, which detail the share and extraction components, and the expiry date of ownership (see Section 2.7). The registry is managed by the NSW Department of Lands for the Department of Natural Resources (DNR) and is responsible for licence administration, renewals and transfers (temporary and permanent) (PC 2003). These access licences are linked to the Water Sharing Plans (WSP) developed by representatives of DNR (DIPNR 2004b).

The rules under the WSP are as follows. WSPs are a requirement of the WMA, and were created for the sustainable use and sharing of water resources in both regulated and unregulated river systems. The most significant aspect of the WSP is the criterion for a share of water to be allocated for the environment, comprising of environmental health water, supplementary environmental water, and adaptive environmental water (ACIL Consulting 2002). 'Environmental health water' is the volume of water committed to fundamental ecosystem health and may not be diverted for other purposes. Supplementary environmental water is committed for specific ecological services at specific times and circumstances, and may be taken for other purposes at other times. Adaptive environmental water is intended for specific environmental purposes of private entitlement holders (PC 2003). The rules for water sharing are established as commence-to-pump (CTP) rules, which specify the minimum in-stream flow the river must reach before irrigators could commence pumping (Aluwihare et al. 2005). The minimum flow accounts for the environmental health water, for this thesis is referred to as environmental flows. DNR is the body responsible for the assessment and implementation of the WSPs. Many water districts have been converted from the Water Act 1912 to the new Water Management Act 2000, which occurs once the WSP for the valley has been gazetted and commenced (DIPNR 2004a). However, up until 2004, plans for many river systems were still being formulated and assessed (Hudson 2005, pers. comm.), so valleys under area-based licences were still charged flat rates. Once the WSP is implemented, all irrigation licences are converted to volume-based licences and subject to a two-part tariff, in line with cost-recovery principles. There is also a differentiation between the cost of supply to a regulated system (rivers with an upstream head dam to control flows) and an unregulated system (rivers without an upstream head dam). In NSW, the prices set to recover the cost of dam operations and water services are set by the Independent Pricing and Regulatory Tribunal.

3.9 THE INDEPENDENT PRICING AND REGULATORY TRIBUNAL AND THE COST RECOVERY OF NSW WATER SERVICES

The Independent Pricing and Regulatory Tribunal (IPART) of NSW is the body responsible for setting maximum prices for water services provided by the State Water Corporation and the Water Administration Ministerial Corporation (WAMC), the current bulk water supply authorities for NSW. IPART has been responsible for regulating bulk water prices since 1996, setting prices to reflect the costs of water supply consistent with Council of Australian Government (COAG) water reform framework. Prior to 2001, the Tribunal set the water prices for the former Department of Land and Water Conservation (DLWC), which had been the sole manager of NSW bulk water services. The Department has since been restructured, replaced by two separate bodies State Water Corporation and the Department of Infrastructure, Planning and Natural Resources (now Department of Natural Resources – DNR). This restructuring was necessitated by the cost-recovery process, where a more transparent cost base for the bulk water services of the former DLWC was needed to determine efficient water price setting (IPART 2004).

State Water is governed by the State Water Corporation Act 2004, and has the primary role of managing all of the NSW bulk water delivery functions on regulated rivers, outside of areas operated by urban water authorities including the Sydney Catchment Authority, Sydney Water Corporation, and the Hunter Water Corporation. State Water releases flows from its dams into rivers to be accessed by its customers, including irrigation corporations, country town water supply authorities, farms, mines, and electricity generators. The delivery of environmental flows is also part of State Water's operations (SW 2005).

The Water Administration Ministerial Corporation (WAMC), created under the former Water Administration Act 1986 (*WAA 1986*), is the statutory entity through which the designated NSW government Minister delivers its water resource management functions, including the approval of projects and other administrative matters. It is through the WAMC that water management policies are designated to DNR (PC 2003).

IPART sets separate maximum prices for State Water and DNR's water resource management activities in supplying bulk water to users from regulated, unregulated and groundwater sources. Submissions made by State Water and DNR detailing pricing proposals form the basis on which IPART sets prices for cost-recovery. The proposals detail the projected costs incurred by each authority in supplying water and providing water supply facilities, and must be justified with comprehensive financial information (IPART 2004). An independent consultant (ACIL Consulting carried out the 2001 determination) is then commissioned to assess the efficiency of State Water and DNR's projected operating and capital expenditures.

The pricing structure for services is a two-part tariff, comprising of a fixed-charge based on volume entitlement and a usage charge per megalitre extraction. The fixed-charge component allows for some revenue stability, while the usage charge is consistent with user-pays and efficiency objectives. This two-part tariff configuration applies to both State Water and DNR pricing structure. The total charge set for regulated rivers is the sum of costs for the two authorities' water services, while unregulated and ground water systems only incur DNR service charges because there are no dam operation costs. Irrigators in unregulated systems are charged a flat-fee until metering is in place, in which time a two-part tariff is charged (IPART 2004).

The latest price determination was made in August 2006 for prices applicable up to June 2010. In one of the earlier determinations in 2001, the Tribunal had increased charges for regulated rivers by up to CPI+15% per year, and unregulated rivers up to CPI+20%. In the following IPART determinations, it was estimated the price level for most regulated valleys the prices are close to full cost recovery level. In the 2005 determination, prices in valleys which achieved full cost recovery are adjusted to the CPI to maintain current prices in real terms (IPART 2005). An example of the structure of prices set for water systems is shown in Table 3-1, which applies to the Namoi Valley in NSW (IPART 2006 p.15 and 135).

Table 3-1: IPART price determinations for Namoi (source: IPART 2005 p. 17).

Institutions	Regulated			Unregulated	
	Entitlement Charge		Usage Charge	Entitlement Charge	Usage Charge
	High security	General			
SWC Entitlement Charge ($/ML)	8.04	5.36	6.42	-	-
WAMC Entitlement Charges ($/ML)	2.62	1.75	2.09	2.30	1.53
Total $/ML	10.66	7.11	8.51	2.30	1.53

Current pricing determinations are designed to recover relevant water resource management costs and for demand management, rather than to address environmental externalities. There has been no pricing provision to encapsulate the environmental costs of withdrawing water from the water sources; the only costs relating to environmental protection are State Water operations including the installation of fish ladders, mitigating thermal pollution, and releasing environmental flows (IPART 2005). While the two-tiered system has the effect of sending water conservation signals, bulk water charges make up a relatively low proportion of consumers' bills. Unless prices are significantly increased, water consumption is unlikely to fall to environmentally sustainable levels pursued in the COAG Water Reform and National Water Initiative 2004. However, a high usage charge for water may devalue water entitlements, since the value of ownership over rights to extract water from the flow of benefit from its use (Goesch 2001).

3.10 SUMMARY OF CHAPTER 3

In this chapter, the suite of institutional arrangements in Australia's water economy has been presented. The transition between the 'expansionary phase' and the 'mature phase' of the water economy is evidenced by the increasing cooperation between States for the sustainable management of water resources, beginning in the 1990s. This was followed by a series of intergovernmental agreements, with growing focus on the environmental aspects of water. However, progress to implement the strategies outlined by the agreements has been slow, and arguably inadequate. The cost associated with achieving various environmental targets, for example in the National Water Initiative and Living Murray Initiative, do not appear to be well justified on economic or ecological grounds. The revenue allocation to achieve the targets and the target itself also seem to be chosen out of convenience rather than based on rigorous evaluation. The determination of an 'efficient' target and the best way to achieve it, however, is difficult. The implications of various water policies and the complexity of water management are discussed further in the following chapter, in which a review of literature on the different aspects of water management is presented.

Chapter 4.

LITERATURE REVIEW

In previous chapters, the evolution of Australia's water economy and its institutional arrangements were analysed. This forms the foundation for this chapter, which reviews literature relating to problems faced in water management. The intention of this chapter is to present the current state of the art in analysing the complexity of moving towards efficient reallocation of water between competing uses.

4.1 CATCHMENT MODELLING – AN INTEGRATED APPROACH

A general trend in water (and environmental) management has been the shift towards interdisciplinary treatment of the problems (Bjornlund 2003a). This shift in doctrine evolved from greater appreciation of the hydrological, ecological, economic and social aspects of water resources, such that a more 'sustainable' use pattern may be established, and to achieve this efficiently via economic instruments (Rolfe 2005). It is therefore appropriate that the interdisciplinary approach is increasingly common in natural resource management, involving a mix of physical, social, economic and ecological perspectives (Jakeman et al. 2005). In economics literature, the amalgamation of economic and biophysical information may be classed as bio-economic modelling, which links environmental outcomes with economic performance. These models are useful for predicting the costs associated with various environmental settings, providing information for more efficient resource management options to be forged (Bennett 2005a).

However, there appears to be a lack of such efficient solutions being implemented. This is evidenced by the relatively ad hoc environmental policies, whereby the targets and revenue allocation for achieving the target seems to be chosen out of expediency. Two significant examples include the Living Murray First Step Agreement to spend $500 million to provide 500GL of water over five years (MDBC 2007c); and the $10 billion

National Water Plan over 10 years, one-third of which will be spent to buy-back entitlements (Howard 2007). Efficient targets should be set such that the last unit of water allocated to the environment equals its marginal value in its next best extractive use. Current arrangements most likely deviate from such efficient allocations (Freebairn 2003), which may be an artefact of the high transaction costs associated with acquiring information regarding the trade-off between extractive and non-extractive uses.

Methods such as non-market valuation help bridge the information gap regarding efficient targets, by eliciting the willingness to pay (WTP) for a bundle of environmental goods. The WTP obtained provides a measure of the social benefit derived from a certain level of environmental protection, and can be used to determine the trade-off between alternative water uses. However, the process is expensive and results are non-transferable, partly due to the inconsistent way data are collected, and the scarceness of available studies (Rosenberger and Loomis 2003). Bio-economic models present a relatively less costly way of estimating the trade-off between different uses of resources (extractive and non-extractive), although the focus has generally been on cost assessment (Bennett 2005a). Ideally, the efficient allocation of resources could be determined through a combination of non-market valuation and bio-economic methods. One notable case, which has incorporated choice-modelling data into bio-economic modelling is Mallawaarachchi and Quiggin (2001), however this seems to be an isolated case.

A number of models have been developed for examining the optimal allocation of irrigation water, through the combined use of biophysical simulation and mathematical programming (e.g., Grismer and Gates 1991; Gretton and Salma 1997; Caswell et al. 1990; Heaney et al. 2001; Powell et al. 2003; Aluwihare et al. 2005; Ancev et al. 2004; Letcher and Jakeman 2002; and others). While there is a range of such 'bio-economic' studies, to the best of the authors' knowledge, most of these have been based on representative farms and often estimate costs on a farm-basis, or rely on representative equations for spatial factors and physical relationships which are fairly inconsistent across different studies. In addition, certain hydrological and biophysical links are excluded perhaps for ease of assessment or for focusing on specific cause-effect relationships.

There is now a range of advanced computer models available which can capture the biophysical relationships in an environment. The advance of computer technology has contributed to the improvement in the study of these spatial relationships, especially with the development of Geographical Information Systems (GIS). However, the use of GIS in resource economics research appears to be limited, particularly within the framework of optimisation economics. In the US, there has been significant drive to stimulate economic research incorporating remotely referenced information since 1992, with funding from US EPA spurring the development of integrated ecological and economic analysis, for the Patuxent catchment in Maryland (Ancev and Odeh 2005; Voinov 1997). The greatest applications of GIS information in economics, however, seems to be in hedonic pricing and spatial econometrics, in quantifying the significance of spatial dependence between e.g. socio-economic variables within regions (e.g. Bateman et al. 2002; Bockstael 1996; Doss and Taff 1996; Clapp et al. 1997).

The use of GIS based biophysical models in Australian economic analysis is also limited, but growing. For example, Nordblom et al. (2007) uses GIS layers of landuse and soil type to generate biomass production using the plant growth model named APSIM; Mallawaarachchi and Quiggin (2001) incorporate GIS in the bio-economic simulations for cane growth, also using APSIM; Bennetton et al. (1998) predict the spread of fire across a terrain using GIS layers, feeding into a benefit-cost analysis of fire fighting in Victoria. There is also good potential for its use in benefit transfer for drawing comparisons between attributes at target sites and existing studies (Troy and Wilson 2006; Ancev and Odeh 2005).

However, a particular area of neglect in these biophysical models has been in the realm of groundwater hydrology. Although models like APSIM are versatile in its ability to “plug in / pull out” sub-modules as required (McCown et al. 1996), the model does not appear to have strong hydrological components linking soil type to groundwater hydraulics. The hydrological model most adopted in NSW is the IQQM, which was developed by the Department of Infrastructure, Planning and Natural Resources and has been used as the basis of Water Sharing Plans (WSP). It estimates the movement of water throughout a river system, based on information on inflows, extractive demands, soil information, and water management rules such as the WSP (Hameed and O'Neill 2005). However, this model also lacks any links between groundwater and surface water systems, as well as crop effects associated with climate (Letcher and Jakeman 2002). One of few economic studies to incorporate a comprehensive geo-hydrologic component is the catchment modelling approach presented in Hatchett et al. (1991). Generally, this model has a similar functioning to a more widely used model, the Soil and Water Assessment Tool (SWAT), which disaggregates land parcels according to soil characteristics of the catchment with links to surface and groundwater hydraulics.

In the present study, the Soil and Water Assessment Tool (SWAT) hydrological model was used to model the Mooki catchment. This geographic information systems (GIS) based model was created by the USDA Agricultural Research Service and has been widely applied to catchment management problems (Neitsch, et al. 2001). The model has been used to map large catchments with spatially varying physical characteristics. These include processes such as water movement, sediment movement, crop growth, nutrient cycling etc., using input data of weather, soil properties, topography, vegetation and land management practices in the watershed. While the model is data intensive, it provides a high degree of spatial detail because of the model's conjunctive use of GIS. An additional advantage is its geo-hydrological component, which links the surface and groundwater systems with landuse and soil profiles. The model does not rely on regression equations to describe the input and output variables; instead it uses specific physical processes to predict impact of land management practices in large, complex watersheds (Neitsch et al. 2001).

The SWAT model has been adopted in a handful of economic studies. A catchment management framework which incorporates the use of spatially referenced data from SWAT is presented in Ancev et al. (2006), to generate optimal solutions to resource use on a site-specific level. Tanaka and Wu (2004) used SWAT to simulate the changes in crop production associated with various nitrogen reduction targets, and derived the costs associated with pollution targets. In Whittaker et al. (2007), the SWAT model is integrated with a generic algorithm to iteratively determine optimal solutions and trade-offs between alternative conservation practices for a watershed. This allows for resource changes from upstream use to feedback into downstream decisions. However, such an approach is laborious and computationally expensive, requiring the use of clusters of 'slave' processors linked through a generic algorithm to reduce the time for convergence. The evolution of GIS-integrated optimisers in Australia may have been hindered by the limited programming expertise and computational requirements for such algorithms, especially those with complex feedback mechanisms.

Nevertheless, the use of a GIS-based economic optimisation framework allows for more accurate assessment of the potential outcome of catchment policies, although perhaps not at insignificant cost and complexity. The treatment of water related issues at a catchment level, based on spatially explicit data from GIS allows the optimal distribution of agricultural activities to be determined, as well as the optimal distribution of water resources. Furthermore, the use of average parameters (e.g. deep drainage coefficients, crop yield, irrigation etc) to analyse inherently heterogeneous landscapes can result in blanket policies that may cause significant inefficiencies (e.g. Greenville and MacAulay 2006; Varela-Ortega et al. 1998). Using a GIS-based biophysical simulation model, the parameters in this thesis are treated on a site-specific basis and at a high level of spatial detail, which allows for a more precise analysis of the social-economic values of water on a catchment scale. It also leaves open the prospect of incorporating non-market valuation into this analysis as future work.

4.2 MARKET-BASED INSTRUMENTS

A number of instruments have been used in natural resource management, including taxes, standards, and, in particular, cap-and-trade which is thought to be more politically favourable (Weber 1999). This decentralised means of resource allocation is preferred because it is postulated to achieve better outcomes by eliminating the transaction costs incurred through centralised agencies (Tsur and Dinar 1995). The water market has been promoted as part of the water reform to achieve efficient solutions, by allowing water to reallocate to its true value (under perfectly competitive conditions). An efficient water market is also thought to reduce the opportunity cost of environmental flows significantly, by allowing environmental water to be sourced from the least profitable irrigation activities (Heaney et al. 2002). However, it appears that water markets in both developing and developed countries are far from the theorised ideal, in which they are able to increase allocative efficiency and to reduce the burden of improving environmental quality. There are often institutional, political, and physical barriers that prohibit the full functioning of water trade and a competitive market. In an imperfectly developed market, potential traders are faced with large transaction costs when entering the water market, including the gathering of market information, finding potential trading partners, legally effecting transfers etc (Carey et al. 2002). While Australian water markets have some formal trading arrangements (e.g. trading platforms including Water Move and Water Exchange Australia), there remain impediments such as poorly defined property rights, uncertainty in supply, infrastructural impediments, excessive transaction and transfer costs, and speculative hoarding behaviour. These factors have effectively deterred irrigators from participating in trade, precluding efficient water markets and limiting the opportunity to source environmental flows at low cost (Crase et al. 2000).

Furthermore, there are often administration fees and commissions to process transactions by water authorities. Many irrigation authorities also actively prohibit inter-valley trading to guard against stranded assets (Goesch and Beare 2004). These factors confine water to be transferred in local markets, which have relatively lower transaction costs because there are less legal and bureaucratic restrictions compared to inter-regional trading (Easter et al. 1998). In addition, transfers are more efficient since there are less transmission losses where water does not need to travel long distances (Carey et al. 2002). Local trade therefore has some advantages over inter-regional trade.

The management of diffuse source pollution through market-based instruments also exhibits significant market failure, due to the considerable transaction cost required to accurately measure its occurrence. Deep drainage is a diffuse source pollution associated with water use and carries salts through the soil profile, contributing to salinity. Some deep drainage is required to carry salts out of the soil profile to avoid salt build-up, which should be at the natural rate provided by rainfall. Too much drainage, e.g. from irrigation, can cause groundwater levels to rise and increase salinity risk (Silburn and Montgomery 2005).

Caswell (1991) highlighted the detrimental effect of polluted drainage waters – which become return flows – as a potential problem to downstream irrigation. However, the cost for the upstream producer to minimise deep drainage may be greater than the benefits to the growers downstream. The 'efficient' level of deep drainage should depend on the cost of reducing deep drainage and whether the quality of return flows has an effect on crop growth. On the other hand, while deep drainage is generally regarded as detrimental to agricultural production, it can also provide benefits to downstream users. Heaney and Beare (2001) found that the improvement in irrigation efficiency upstream has implications for the volume and quality of water downstream, transmitted through the reduction of the level of return flows that contribute to surface and ground water supply. In this way, deep drainage can have both positive and negative effects on downstream water users. If water quality has negligible effects on crop growth, then the upstream user should only reduce deep drainage to the extent where downstream supply is not affected.

It is considered optimal to create exclusive property rights over resources in order to contend with market failures inherent to 'common-pool-resources', with one instrument corresponding to one objective (Tinbergen 1950). However, because water use and deep drainage are linked, it may not be efficient to have separate instruments for their control. Weinberg et al. (1993) examines the extent to which water market in its own right could contend with water quality (salinity) problems. This study shows that introducing water trade can lead to an overall reduction in water use and by associated deep drainage, although this is not necessarily the least-cost solution that can be achieved. Nevertheless, the difference in benefit between a targeted deep drainage reduction policy compared to a water reduction policy, is only of a small magnitude; the cost of acquiring accurate information regarding discharge points in order to implement drainage policies may involve transaction costs that outweigh any difference in efficiency between an optimal set of input taxes and a water market. This finding is reiterated in Legras and Lifran (2006). The authors model irrigation-induced salinity under different market designs, in the form of a water diversion cap, and two decoupled markets for diversion caps at the basin level and recharge caps at the zone level. The findings imply that a catchment-level water diversion market would be more efficient in managing coupled externalities. Caswell et al. (1990) also suggested that using water-pricing policies might be more effective than deep drainage pricing to induce changes in irrigation practices. This conclusion was drawn from the finding that the effectiveness of drainage pricing on technology switch is low relative to water pricing. This is simply because the volume of water applied is higher than the volume of drainage generated in irrigation.

The potential to use a cap-and-trade scheme as a means to effectively manage recharge, is examined in Whitten et al. (2005), which studies the use of tradeable recharge (deep drainage) credits in Coleambally Irrigation Area. It was found that introducing a recharge cap-and-trade scheme would provide a relatively small gain, and hence small farm scale benefits. One significant obstacle identified by the authors is the diffuse nature of recharge, which is different to other cap-and-trade models where the pollutant is point-sourced and measurable. It is difficult to link the drainage occurrence to one distinct origin due to the spatial and temporal variation in its occurrence. The conclusion was that usual cap-and-trade models are not likely to generate sufficient benefits to drive its full adoption, in addition the expectation that the full gains from trade will not be realised in practice. Another alternative is point/non-point source credit trading, which has been implemented in some catchments in the US (Horan 2001). Under this system, point source polluters are able to purchase additional credits from non-point polluters at a given conversion rate. However, the trading ratios, for which non-point source credits are converted to point-source pollution, is subject to significant uncertainty. This is because the rate at which non-point loadings are reduced is regarded as an imperfect substitute for point-source loading reductions. Furthermore, the market requires that non-point and point-source polluters coexist in a basin for it to be effective.

While salinity standards have been set for NSW catchments, as part of the Murray-Darling Basin Salinity Management Strategy, the practicality of a widespread salinity-capping scheme is therefore in question. The diffuse nature of deep drainage makes it costly and difficult to quantify, not to mention the expense of instruments used to measure deep drainage, e.g. lysimeters (Triantafilis et al. 2003). The time lag between when deep drainage becomes recharge to the shallow groundwater table also adds to the complexity in accurately measuring its incidence. While there are now separate and tradeable water entitlements to promote efficient allocation, the use of market-based deep drainage instruments has not been widespread. Proposed deep drainage reduction policies have ranged from voluntary best management practices (e.g. on-farm storage lining) to district-level drainage (salinity) restrictions (e.g. Basin Salinity Management Strategy). A number of papers recognised deep drainage occurrence as being contingent on technology and soil quality (Ancev et al. 2004; Caswell et al. 1990; Khanna et al. 2000). These authors advocate the use of pollution taxes and the adoption of efficient technology as the best means to provide conservation incentives for polluting inputs (deep drainage). However, the appropriate salinity control measure is likely to vary from basin to basin, depending if salinity (deep drainage) is a persistent problem in the region.

4.3 HYDROLOGICAL ISSUES – RETURN FLOWS

Some literature cautions against relying on modern irrigation systems as the solution to conserving water in river basins altogether. Scheierling et al. (2004) and Huffaker and Whittlesey (2000) suggests that promoting water efficient technologies without regard for overall basin hydrology may lead to adverse outcomes for the basin due to the contribution return flows make to downstream water supply. Scheierling et al. (2004) stresses that unconsumed water becomes return flows that contribute to downstream water supply, so reducing deep drainage through improvements in irrigation efficiency would only result in a fall in irrigation water for downstream users. By this reasoning, policies encouraging the use of efficient irrigation technologies would have limited effect on generating real increases in surface water. However, it is intuitive that in-stream flows should increase due to a reduction in direct water extractions, thereby compensating for the fall in return flows. In addition, reducing return flows, and hence deep drainage, would mean improved water quality in terms of diminished salt content and salinity.

It is also true that if the irrigator becomes more water efficient, and in the absence of water market, the water conserved would simply be used to expand production if the producer has idle land. The outcome would be a net reduction in basin water supply since the same volume is being extracted but less becomes return flow due to improved irrigation efficiency. Huffaker and Whittlesey (2000) contend with this effect of reduced downstream supplies by factoring in the opportunity cost of improved upstream irrigation efficiency. The net benefit of improving irrigation efficiency at a location upstream is measured against the foregone benefit of decreased irrigation return flow downstream, and the desirability of any improvement in upstream efficiency falls as more water become return flows. In essence, both Scheierling et al. (2004) and Huffaker and Whittlesey (2000) suggest that without considering the hydrological relationship in the basin, policies to improve irrigation efficiency may lead to unexpected adverse outcomes.

Water trade and improvements in water use efficiency have similar adverse effects of reducing return flows, which impact on the quality and volume of water used downstream. It has been suggested that a system of property rights for return flows is required to internalise these effects (Heaney and Beare 2001). If return flows are from irrigation areas with relatively low underlying groundwater salt concentration it can provide dilution flows downstream. Water trade or improving irrigation efficiency may reduce these positive externalities because it deprives some areas of beneficial return flows (Heaney and Beare 2001). On the other hand, deep drainage has also been considered as pollution, for example in Caswell et al., (1990). If the recharge is high and ground water salinity is high, return flows would contain high salt concentrations and increase salinity risk. Under such circumstances, if deep drainage is reduced, the amount of saline recharge transported to the river system is also reduced.

Improving upstream irrigation efficiency is thus beneficial only if downstream groundwater salinity is high and groundwater response times are short; defined as "the time it takes for a change in recharge to be reflected in a change in saline discharge" (Heaney and Beare 2001). The aquifer in the Namoi, where the present research is focused, is thought to be responsive to changes in deep drainage (Karen Ivkovic 2005, pers. comm.), meaning the recharge rates are quite high. The recharge time estimated for the Mooki is 17 days (Vervoort 2005, pers. comm.), so the response time is short and water quality changes are realised quickly. Excluding the beneficial properties of return flows in the economic analysis is therefore appropriate for the Mooki case study, since the quality of return flows may generate greater negative than positive externalities by augmenting salinity problems. This is discussed in detail in Chapter 6.

Furthermore, the reduction in return flows on downstream supply should not be an issue if there is the opportunity to substitute this volume from the water market, where it exists. Rather than relying on return flows, the irrigator could purchase water directly from the market, to compensate for the reduced volume. Under these circumstances, the decision to invest in water efficient technologies would not be influenced by considerations of reduced return flows; each irrigator would select the system that maximises their profit. Downstream irrigators who are affected would purchase water allocations from the market if it is profitable to do so. Those downstream irrigators who find it unprofitable would forego the reduced volume and adjust production accordingly. This would allow return flows to be valued explicitly, rather than implicitly through its value in production.

4.4 WATER PRICING AND WATER USE EFFICIENCY

The pricing of water and whether it can improve water use efficiency is a particularly contentious subject. The intention of the cost-recovery process was to better reflect the true cost of supply of water, ideally equating the marginal environmental cost and marginal economic cost. However, significant increases in water prices have implications for equity in access, since the supply of cheap, high quality water has been viewed as a basic right (Godden 1997). It is also politically unfavourable, and users have been able to exert political influence to preclude increases in irrigation water prices (Easter et al. 1998). Some authors also cite the relatively large price increases required to reduce demand, which would also affect low income groups more than wealthier groups (Renwick and Green 2000).

The social implications of water price increases may have contributed to government's reluctance to use pricing policies to manage water demand. Government intervention has typically been through quantitative instruments regulating the level of water use, for example through a Cap on water diversions. In NSW, this Cap is written into Water Sharing Plans as a minimum in-river flow requirement, in recognition of the environment as a legitimate user of water. However, the Cap may not reflect the efficient allocation, since the opportunity cost of production of the water capped from irrigation may not justify the environmental benefits generated. This is due to the relatively ad hoc nature of caps (Freebairn 2003). Therefore, whether price-based or quantity-based water demand management is the most effective means of achieving efficiency targets is debatable.

The use of modern irrigation systems has been advocated as an effective means of achieving water conservation while maintaining profitability. The importance of alternative irrigation systems in achieving water conservation targets has been highlighted in several papers, including Ancev et al. (2004), Bernardo et al. (1987), Caswell et al. (1990), and Varela-Ortega et al. (1998). Bernardo et al. (1987) demonstrate that even with little flexibility in terms of irrigation system choice the investment is worthwhile even in the short run. This finding is also supported by Caswell et al. (1990), and further examines the effect of using water pricing as a catalyst to investing in efficient irrigation technologies. The central argument in Caswell et al. (1990) seems to be that technology choice is dependent only on water price or subsidy and tax combinations, which is enough to spur the switch from one technology to another. If output prices are high then drip irrigation prevails as the most favourable option. However, Varela-Ortega et al. (1998) argues that farmers' choice in technology is not necessarily just price dependent but depends on the agronomic and structural limitations, financial constraints, and to a lesser extent on water prices. This is a conclusion that is coherent to Foley and Raine (2001), who suggested the change in irrigation systems is not necessarily out of economic consideration but simply because of convenience when replacing systems. Additionally, most farmers have preferred sprinkler irrigation to drip systems mainly due to the difficulty in maintaining drip systems (Murray 2004).

A general consensus seems to be that imposing a uniform water pricing system for a water district to meet desired water efficiency objectives is likely to result in considerable economic losses. While higher water prices can spur greater adoption of water-saving technologies, policies must be region specific because a blanket pricing scheme would merely induce small changes in water use but cause significant income loss. Older water districts, where technological efficiency (T.E.) is lower and water demand is more elastic, have different responses to more modern irrigation districts, where T.E. is higher and water demand is less elastic. These two areas have different potentials for water savings as a result of the level of T.E., characterized by the level of water demand elasticity. As a result, an across-the-board increase in water price may only result in net welfare losses and little water savings if localized, heterogeneous conditions are not taken into account when formulating policies for a region (Caswell et al. 1990; Varela-Ortega et al. 1998).

The current water pricing authority in NSW appears to be well aware of such welfare impacts. Cox and Warner (2007) discuss the importance of limiting bulk water charges primarily to recover the operational cost of delivering water, and with a secondary objective of demand management. Conservation signals can be provided through the market price for water would be eroded if water delivery charge were high, since it reduces the flow of benefits received from its ownership. In the presence of water market opportunities, there is a natural incentive to invest in water efficient technologies up to the point where the benefit from water savings equates the capital costs (Ancev and Vervoort 2007).

4.5 EXTERNAL WATER TRADING AND STRANDED ASSETS

A pressing issue is the prospect of external buyers entering a regional water trading system in the near future, which has implications for the cropping industry and may open the possibility of stranded assets. This comes amidst discussions of substantial government buy-backs of entitlements for environmental flows. The Commonwealth

Water Plan announced early 2007 budgeted $10 billion towards managing the MDB water resources, $3 billion of which is allocated towards directly buying-back water entitlements (Howard 2007). There is also the possibility of an expansion of coal reserve mining, a water intensive industry, in the northern region of NSW. The Gunnedah shire overlays one of the largest underground coal reserves in NSW, which stretches 500km from Wollongong and Narrabri, and is 150km wide (Gunnedah Shire 2006). A five-year exploration licence has been granted in early 2006, for exploration and development of coal reserves in parts of the Mooki valley. This has instigated significant community debate pertaining to environmental concerns as well as competition for infrastructure and water, given current plans to substantially reduce surface and groundwater entitlements over the next five to ten years (Strang 2006).

However, unfettered water trading between competing uses is thought to achieve a most efficient outcome, enabling water to trade to its highest value use or sectors (Tsur and Dinar 1995; Carey et al. 2002; Heaney et al. 2002; Goesch and Beare 2004, and others). Some authors advocate for rural-urban trade (Weinberg et al. 1993; Dwyer et al. 2005). This is perceived as an efficient way to improve on the current water allocation system, which has been relatively fixed even though demand has risen more in the industry and urban sectors compared to agriculture (Rolfe 2005).

The benefit of encouraging greater trade between regions is illustrated in Weinberg et al. (1993). They estimated the rural water market price to be less than half the price in urban markets in their Californian case study. This presents an opportunity for useful water transfer between the urban and rural sectors. Rural-urban trade was also examined in an Australian context and estimated net gains to trade were greatest when trade is unfettered between irrigators in south-east Australia and also to townships in interconnected hydrological systems (Dwyer et al. 2005). The greatest benefit could therefore be achieved if transaction costs are reduced to increase the probability of an efficient equilibrium, and encouraging greater trading to external agents outside of the irrigation district or industry (Carey et al. 2002).

However, allowing the entry of an external buyer into the regional water market increases competition for surface and ground water resources. This is in light of significant cuts in groundwater entitlements in the Namoi catchment that are expected over the decade. Resources will become increasingly scarce, as it is likely there would be some reallocation of resources from agriculture to mining or the environment. It is important that the impacts of external agents competing with irrigators for water are assessed, to avoid significant social costs on rural economies dependent on irrigation industries.

A commonly cited reason for restricting trade is because as irrigators leave an irrigation district, a higher cost is imposed on those remaining in the system (Goesch 2001). This is the problem of stranded assets, whereby higher delivery charges are imposed on remaining irrigators as more water property rights are sold outside the irrigation district. In extreme cases the irrigation authority is left with large fixed costs and no customers, so external trading is prohibited by some catchment authorities due to the risk of stranded assets. This has implications for the Commonwealth Water Plan, in which the purchase of environmental flows form a significant portion of planned expenditure. In order to minimise the socio-economic impact of purchases, the government can either enter markets across regions, or by paying an exit fee to prevent high capital costs being imposed on the remaining irrigators in low value areas. Alternatively, a structural adjustment package could be provided to those regions significantly affected by the withdrawal of water (Goesch and Heaney 2003). These considerations again highlight the importance of site-specific analyses, in order to evaluate the economic impacts that arise from government intervention.

4.6 CATCHMENT STUDIES OF THE NAMOI

Key studies pertaining to water reform in the Namoi river valley include Ancev et al. (2004), Aluwihare et al., (2005), CARE (2003), and Letcher and Jakeman (2003). Both Ancev et al. (2004) and Aluwihare et al., (2005) model the profit maximising allocation of irrigation water under limited water supply for multi-crop producers for the Mooki River sub-catchment. Ancev et al., (2004) look at the basin wide allocation of water, while Aluwihare et al., (2005) conducts a farm-level model to analyse the socio-economic costs of proposed Water Sharing Plans (WSP). The novel component of the model in Aluwihare et al., (2005) is that it integrates the existence of on-farm storages, which is an important feature on farms in unregulated rives, and the analysis of uncertainty in production. However, water losses to deep drainage and groundwater hydrology were not considered, although much literature points to these as significant components that need to be incorporated. Nor did it consider the use of alternative, water efficient, irrigation systems and the potential gain in the presence of a water market.

Letcher and Jakeman (2003) analyse the impact of water policies at a basin level, by aggregating the basin into large portions of homogenous regions, according to the area measured by stream gauges in the catchment. Its ‘integrated assessment’ framework involves a hydrological model that links upstream water extractions to downstream water supply, which feeds into an agricultural production model. The study uses a sophisticated model that focuses on the management of ‘off-allocation’ water – water that spills from the dams – in the Namoi Valley. Its main shortfall is the lack of a groundwater component, and does not consider salinity impacts or changes in environmental policy. The value of water efficient technologies and the water market were also not evaluated as adjustment options if water resources become scarcer.

CARE (2003) conducted a study of the groundwater WSP on the Namoi catchment, based on representative farms for each Zone in the Namoi. The likely impact of the WSP on these farms was assessed over a 20-year period, using an input-output (IO) analysis to assess the region-wide impact. The data used regarding farmer’s production activities and assumptions on economic parameters were very detailed. However, the use of input-output tables has been more commonly employed for analysing the effect of changes in demand for outputs, rather than to measure impacts of changes in resource availability. Furthermore, IO analyses tend to overestimate regional impacts due to the erroneous application of the “value-added” approach to estimate shadow prices of (for example) water, without subtracting the opportunity cost of non-water inputs (Young 2005). It is considered a poor substitute for a full economic analysis, although it is regarded as a cheaper and less controversial option (Bennett 2005a).

Also taking a catchment level approach, Ancev et al. (2004) presents a combined hydrological-economic model to examine the optimal basin water allocation and deep drainage occurrence. The modelling framework proposed in Ancev et al. (2004) involves the use of SWAT, a GIS-based biophysical model, to provide site-specific results. Constraints pertaining to deep drainage were also formulated to examine the effect of policies targeting the level of deep drainage that occur in the catchment. A similar method of analysing the effect of pollution targets is also analysed in Tanaka and Wu (2004). SWAT was used to simulate the changes in crop production associated with various nitrogen reduction targets, for which corresponding marginal profit loss curves were derived. This process is similar to Ancev et al. (2004), but instead of deep drainage targets, nitrogen targets have been set in Tanaka and Wu (2004). Based on this methodology, marginal profit/loss curves could be obtained for any desired environmental constraint, including environmental flows. These two studies form the basis of the modelling framework adopted in this thesis.

4.7 SUMMARY OF CHAPTER 4

In this chapter, a literature review of the published work in the field of water management and the physical complexities involved has been presented. From this review, the issues relating to the case study are highlighted for the purposes of building an appropriate model for this thesis. Using an integrated biophysical and economic modelling approach, simulations from the biophysical model, SWAT, will form the basis of an economic optimisation model in this thesis. Rather than modelling a representative farm, site-specific, spatially referenced information of the case study of the Mooki basin in the Namoi valley is used. Using this approach, the model provides realistic results that are directly applicable to the basin. It also has an advantage over modelling a representative farm because of the heterogeneous nature of catchments. This is because blanket policies devised for a representative farm may be ineffective or lead to significant social costs, e.g. large income losses for small changes in resource use (Varela-Ortega et al. 1998). The optimisation process is conducted from a catchment perspective constrained by environmental targets and water supply limits, which would allow more insight into the impacts of various environmental policies and how irrigators could adjust to them at the least-cost.

In the following chapter, the specific model used in this thesis is presented. The model specifications, in terms of the linear programming and dynamic programming components, are examined in detail. This includes a discussion of the theoretical solutions to water allocation based on this modelling framework, which are in accordance with economic efficiency.

Chapter 5.

ANALYTICAL FRAMEWORK

In this chapter, the economic model used to determine efficient resource allocation for the case study, the Mooki basin, is presented. The optimisation process involves finding the resource distribution that maximises social economic welfare from a catchment manager's perspective, given resource constraints and environmental objectives. The solution provides profit maximising production mix for various water availabilities, in the incidence of government policy changes or increased competition for water.

5.1 SOCIAL OPTIMISATION v PRIVATE OPTIMISATION

There are two ways to model basin water allocation. One way is to determine optimal water allocation from a catchment manager's perspective, and the other is from an individual irrigator's perspective. The optimisation process from both viewpoints is similar, since each would have a common objective to maximise net benefit from production subject to resource constraints. The main difference is that a catchment planner would have the aim of maximise social wellbeing, an integral part of which involves maximising profit to farmers. This is done by distributing water according to its value at the margin across different users. From a private perspective, the opportunity cost of resource use is confined to the farm-level. The water scarcity rent and externalities which have not been priced, in the form of salinity and return flows, are not internalised in the private irrigator's water allocation decisions. As a result, a model simulating individual profit maximising objectives may generate results that deviate from what is socially desirable, due to the discrepancy in the opportunity costs included in the objective. There would be undeniable benefits from an approach that examines behaviour of individual producers, since results from such an analysis could be used to predict the effect of policy. However there remains the need for a policy direction towards socially optimal outcomes, which would not be the case when the problem is only examined from the perspective of individual producers. It is therefore useful to model from a social perspective, in order to provide policy direction as to the least-cost means of achieving environmental targets and resource constraints at a basin-level. An optimisation model from a perspective of catchment manager is conceptualised below.

5.2 MODELLING COMPONENTS

The irrigation water management at the catchment level will be addressed through an integrated biophysical and economic modelling. The first component is the biophysical model, Soil and Water Assessment Tool (SWAT). This is a basin scale, physically-based hydrologic model that uses Geographic Information Systems (GIS) data to perform parameter estimation and geographical analysis. The agronomic and hydrological outputs form the input for an economic optimisation model, which maximises net catchment profit given resource constraints and environmental targets. This involves intra-seasonal static optimisation, through a linear programming (LP) model, and inter-seasonal dynamic optimisation of groundwater use. This two-stage decision making process ensures that net social benefit is maximised across the planning horizon, and enables the value of inter-temporal trade-offs of groundwater resources to be integrated into production decisions. Surface water allocations cannot be carried over and they have to be used within one season, due to the ephemeral nature of the Mooki. A planning horizon of ten years is considered. This is because the cut-back in future groundwater allocations, according to the groundwater Water Sharing Plan, is to be phased in over ten years. An additional reason is that amortization period of ten years was assumed for capital investments in alternative irrigation technology.

The economic impact of increased competition for water resources is also evaluated, by simulating the effect of an external water user in the regional water market. An empirically derived demand for water for a large coalmining company is used for this analysis.

5.3 MODEL SPECIFICATIONS

The driving force behind the economic modelling is profit maximisation by agents that use scarce resources. This is done from a catchment manager's perspective, for which profit maximising forms an important component in attaining maximum social welfare. The objective is to find the profit maximising distribution of water for the Mooki Basin subject to water availability and deep drainage (DD) constraints, given choices in crops grown, source of irrigation water, irrigation area, irrigation systems, and the opportunity to trade water. A benevolent catchment manager who could hypothetically exert control over these choices would therefore make optimal decisions with respect to these choice variables in such a way that maximises the net social benefits from agricultural activities, but at the same time takes into account resulting environmental impacts. The environmental impacts are predominantly caused by extractive water use and by DD, resulting in increased groundwater and soil salinity, and potential for water logging. Had the groundwater salt concentration of the Mooki been low, there would also be the externality of reduced return flows from improved irrigation efficiency. These effects enter into the catchment manager's decision problem, such that resources are distributed in a way that results with the greatest social benefit in the long-run. The following sections provide greater details of the modelling components.

5.3.1 The Optimisation Model

The optimisation process involves two-stages. One is a dynamic programming model to maximise expected net present value (NPV) across *T* periods in the light of the possibility for inter-temporal tradeoffs in groundwater allocations, currently in place in the case study catchment. The second stage is a linear programming (LP) model designed to optimise resource use across hydrological response units (HRUs) (*N* land parcels, denoted by subscript *i*) within a single period, *t*. These resource use decisions are based on the expected future surface water allocations. The objective function for the dynamic optimisation model is given by the recursive equation:

$$V_t\{G_{it}\} = \max_{G_{it}} \left[E\{\pi_t(G_{it})\} + \beta E\left\{V_{t+1}\left(\overline{Ga}_{it} - G_{it}\right)\right\} \right] \quad (i = T, \ldots, 1) \tag{1}$$

Where:

$V_t(.)$ is the optimal value function from period *t* to the end of the planning period, *T*;

β is the discount factor and equals $1/(1+r)$;

G_{it} is the volume of groundwater pumped by the i^{th} HRU in period *t*,

$\overline{Ga_{it}}$ is the groundwater allocation available for extraction by the i^{th} HRU in period *t*;

The term $\pi_t(G_{it})$ is the basin profit in period *t*, as a function of the control variable, G_{it} (groundwater use decisions of HRUs in period *t*). The value of $\pi_t(G_{it})$ is found through the LP model, which is used to perform static optimisation of resource use within one season. This is done via decision variables for individual HRUs: surface water use for crop *j* (S_{ijt}), groundwater use for crop *j* (G_{ijt}), crop choice (J_{it}), irrigation system choice (Z_{it}), and water allocations purchased (Wd_{it}) or sold (Ws_{it}).

This is represented by:

$$\Pi_t(G_{it}) = \sum_{n=1}^{N}\left(\sum_{j=1}^{J}\pi_{it}(S_{ijt},G_{ijt},Z_{it},J_{it})\cdot AJ_{it} - P_w Wd_{it} + P_w Ws_{it}\right) \tag{2}$$

where $\pi_i(S_{ijt},G_{ijt},Z_{it},J_{it}) = \left(P_j Y_{ijt} - C_{ijt}\right)\cdot J_{it} =$

$$= \left(P_j \cdot f_{ijt}\left(W_{ijt}\right) - \left(\sum_{z}^{Z} FCI_{iz} + WA_{ijt}(z)\cdot a_{ijz} + P_s S_{ijt} + P_p G_{ijt} + OtherCosts_j\right)\right)\cdot J_{it} \tag{3}$$

Refer to Inset 5.1 for a full description of the variables in Eqs. (2) and (3).

Irrigation water can either be diverted from surface water bodies or pumped from groundwater. Application costs are higher when using the groundwater source for each irrigation technology, because of pumping equipment and fuel (Smith and Richards 2003). However groundwater is a more reliable source than the surface water, and is therefore the marginal source used whenever there is shortage of surface water.

Inset 5.1

$\pi_{it}(.)$ is profit per hectare in the i^{th} HRU in period *t*, expressed as the sum of profit per hectare of *J* crops produced in the i^{th} HRU in period *t*;

AJ_{it} is the area planted under crop *j* in hectares of the i^{th} HRU in period *t*;

P_w is the market price of water;

Wd_{it} is the amount of water bought by the i^{th} HRU in period *t* through the water market;

Ws_{it} is the amount of water sold by the i^{th} HRU in period *t* through the water market;

S_{ijt} is the per hectare surface water applied to crop *j* in the i^{th} HRU in period *t*;

G_{ijt} is the per hectare ground water applied to crop *j* in the i^{th} HRU in period *t*;

J_{it} is the crop choice in the i^{th} HRU in period *t*, given by:

$$J_{it} = \begin{cases} 1 & if \quad crop = j \\ 0 & otherwise \end{cases}$$

Z denotes to the irrigation system used for irrigation;

r is the discount rate;

P_j is the price received for crop *j*;

FCI_{iz} is the annualised fixed cost per hectare including initial investments and continued maintenance costs of using irrigation technology z in the i^{th} HRU;

W_{ijt} is the effective water consumption per hectare by crop *j* in i^{th} HRU in period *t*;

$WA_{ijt}(z)$ is the water applied per hectare to crop *j* in the i^{th} HRU in period *t*, and is a function of the irrigation system used;

a_{ijz} is the application cost of irrigation, depending on the choice of irrigation system z;

P_s is the per unit cost of using surface water, including the pumping cost from the river and usage charge;

P_p is the per unit cost of using groundwater, including the pumping costs from the aquifer and usage charge;

$OtherCosts_j$ is the fixed cost per hectare of producing crop *j*, excluding irrigation costs.

A distinction is made between effective water consumed by crop j (W_{ijt}) and applied water (WA_{ijt}) because the volume WA_{ijt} is likely to be greater due to losses in conveyance and application, depending on the irrigation system used (Foley and Raine 2001). The variable WA_{ijt} is contingent on the irrigation requirement per hectare for the crop in the i^{th} HRU, limited by the volume of surface and ground water available to it. This is augmented by water bought, Wd_{it}, and diminished by water sold, Ws_{it}. The total volume of water used for irrigating crops J in the i^{th} HRU in period t is subject to the following constraint:

$$0 \leq \sum_{j=1}^{J} WA_{ijt}(z) \cdot AJ_{it} + Ws_{it} - Wd_{it} \leq S_{it} + G_{it} \tag{4}$$

The volume of surface water allocation available for extraction, S_{it}, is a proportion of the total river flow at time t, $\delta \overline{S}max_t$. The volume of surface water that can be extracted is therefore subject to the following constraint:

$$S_{it} = \delta \overline{S}max_{it} \tag{5}$$

The term $\overline{S}max_t$ refers to the maximum total extraction limit, which is in turn determined by the total available river flow in period t. This is discussed in section 5.3.2. The conditions on G_{it} will be discussed further with reference to the dynamic programming.

5.3.2 Environmental Constraints

The environmental constraints are imposed on an annual basis, in the form of DD (which has implications for the occurrence of salinity) and environmental flow targets. These are defined in the LP model as:

$$\sum_{n=1}^{N} \sum_{j=1}^{J} (WA_{ijt} - W_{ijt}) \leq DD_t \tag{6}$$

$$\overline{S}max_t \leq \overline{FL_t} - CTP_t \tag{7}$$

Where $\sum_{j=1}^{J} (WA_{ijt} - W_{ijt})$ is the water lost to DD in application, given by the difference between applied and effectively consumed water for all crops in the N HRUs. The total DD in water application must be less than or equal to a set target for the basin in period t, DD_{it}. This constraint is to analyse the effect of meeting end-of-valley targets, aimed at reducing salinity contribution from the basin to the Murray-Darling River system.

Constraint (7) specifies that $\overline{S}max_t$ must not exceed the basin river flow in period t, $\overline{FL_{it}}$, less the Commence-To-Pump limit, CTP_t (the level that in-river flow must reach before extractions can begin). This constraint on surface water use simulates the surface Water Sharing Plan for the case study, which limits the extractive water taken from the river and ensures that environmental flows are provided. The term CTP_t is synonymous to environmental flow rules, and these rules were implemented in SWAT. The effect of further environmental flow rules is exemplified through the constraint on water extractions in Eq. (7), which is parameterised to evaluate the effect of different environmental flow levels on basin profit. This essentially represents a reallocation of extractive water use towards environmental purposes. Annual surface water allocations are therefore successively reduced from the full allocation to simulate the economic impact of increased environmental flow requirements. Similarly, the economic impacts of DD targets are parameterised by varying the value of DD_t in constraint (6). The value of DD is also reduced successively from the unconstrained occurrence associated with full water allocations. The relationship between these environmental constraints and its associated economic impact can thus be derived, and used to form cost functions for achieving environmental targets. The following sections present the optimality conditions given these environmental, and resource constraints.

5.3.3 Static Optimisation Framework

The static optimisation problem, for efficient resource use in the i^{th} HRU, can be represented by the following Lagrangian adapted from the model presented in profit Eq. (3), and constraint Eqs. (4) and (6):

$$\begin{aligned}
L\left(S_{ijt}, G_{ijt}, DD_{ijt}\right) = {} & \left(P_j \cdot f_{ijt}\left(W_{ijt}\right) - \left(\sum_z^Z FCI_{iz} + WA_{ijt} \cdot a_{ijz} + P_s S_{ijt} + P_p G_{ijt} + OtherCosts_j\right)\right) \cdot J_{it} - P_w Wd_{it} + P_w Ws_{it} \\
& -\lambda_{DD}\left(\sum_{i=1}^N \sum_{j=1}^J \left(WA_{ijt} - W_{ijt}\right) - DD_t\right) \\
& -\lambda_w\left(\sum_{j=1}^J WA_{ijt} \cdot AJ_{it} + Ws_{it} - Wd_{it} - S_{it} - G_{it}\right)
\end{aligned} \tag{8}$$

where λ_{DD} is the Lagrangian multiplier on deep drainage constraint, which can be interpreted as a shadow value of deep drainage (DD), and λ_{iw} is the Lagrangian multiplier on the water availability constraint, which can be interpreted as the shadow value of water for the i^{th} HRU. Drainage is regarded as 'pollution' that is conjoint with the use of water to produce irrigated crops. To simplify the discussion of theoretical solutions to the LP problem, some notational substitutions are made. The term $\sum_{i=1}^{N}\sum_{j=1}^{J}\left(WA_{ijt}-W_{ijt}\right)$ is substituted by $g\left(DD_{ijt}\right)$, to denote the total drainage in the basin as a function of the drainage in each HRU. Similarly, the term $\sum_{j=1}^{J} WA_{ijt}\cdot AJ_{it}+Ws_{it}-Wd_{it}$ is substituted by $h\left(S_{ijt},G_{ijt}\right)$, to denote basin water use as a function of surface and groundwater use in each HRU. Assuming an internal solution exists, the first-order condition (FOC) for optimality with respect to surface and groundwater used by the i^{th} HRU, and the associated 'optimal' pollution in the form of drainage, DD_{ijt}, are represented by:

$$\frac{\partial L}{\partial DD_{ijt}}=\left(P_j\cdot\frac{\partial f_{ijt}\left(W_{ijt}\right)}{\partial DD_{ijt}}-\frac{\partial WA_{ijt}}{\partial DD_{ijt}}\cdot a_{ijz}\right)\cdot J_{it}-\lambda_{DD}\frac{\partial g\left(DD_{ijt}\right)}{\partial DD_{ijt}}=0 \tag{9}$$

$$\frac{\partial L}{\partial S_{ijt}}=\left(P_j\cdot\frac{\partial f_{ijt}\left(W_{ijt}\right)}{\partial S_{ijt}}-\frac{\partial WA_{ijt}}{\partial S_{ijt}}\cdot a_{ijz}-P_s\right)\cdot J_{it}-\lambda_{iw}\left(\frac{\partial h\left(S_{ijt},G_{ijt}\right)}{\partial S_{ijt}}\right)=0 \tag{10}$$

$$\frac{\partial L}{\partial G_{ijt}}=\left(P_j\cdot\frac{\partial f_{ijt}\left(W_{ijt}\right)}{\partial G_{ijt}}-\frac{\partial WA_{ijt}}{\partial G_{ijt}}\cdot a_{ijz}-P_p\right)\cdot J_{it}-\lambda_{iw}\left(\frac{\partial h\left(S_{ijt},G_{ijt}\right)}{\partial G_{ijt}}\right)=0 \tag{11}$$

$$\frac{\partial L}{\partial \lambda_{dd}}=g\left(DD_{ijt}\right)-DD_t=0 \tag{12}$$

$$\frac{\partial L}{\partial \lambda_{iw}}=h\left(S_{ijt},G_{ijt}\right)-S_{it}-G_{it}=0 \tag{13}$$

Eq. (9) suggests that the optimal level of DD_{ijt} should be such that its marginal value in production (alternatively the marginal benefit of pollution as defined in Hartwick and Olewiler, 1986), $P_j\cdot\frac{\partial f_{ijt}\left(W_{ijt}\right)}{\partial DD_{ijt}}$, equates with the application cost associated with reducing deep drainage (marginal cost of abatement), $\frac{\partial WA_{ijt}}{\partial DD_{ijt}} \cdot a_{ijz}$, plus the marginal value of emitting DD elsewhere in the catchment, $\lambda_{DD} \frac{\partial g\left(DD_{ijt}\right)}{\partial DD_{ijt}}$. The occurrence of DD is associated with the water use in the basin such that, without water trade, reductions in DD are confined to reductions in associated water use within the HRU. In the presence of water trade, DD becomes mobile and can be transferred to various parts of the basin. This then resembles a DD market, where DD is transferred through water trade, and the 'efficient' DD_{ijt} occurrence on a basin scale is then achieved as drainage is shifted to its highest value use.

Eq. (10) suggests the optimal level of surface water use S_{ijt} should occur at a point where its marginal value in production, $P_j \cdot \frac{\partial f_{ijt}\left(W_{ijt}\right)}{\partial S_{ijt}}$, equates with the application cost (marginal factor cost) of S_{ijt}, $\frac{\partial WA_{ijt}}{\partial S_{ijt}} \cdot a_{ijz}$, plus the opportunity cost of S_{ijt} in the water market, and the opportunity cost of S_{ijt} in production elsewhere in the catchment, $\lambda_{iw}\left(\frac{\partial h\left(S_{ijt}, G_{ijt}\right)}{\partial S_{ijt}}\right)$. In the absence of water trade, the opportunity cost of S_{ijt} is zero and the opportunity cost of water use is confined to the marginal value product (MVP) of irrigated production within the HRU. Where a water market exists, water is traded to its highest value use within the catchment such that the MVP of S_{ijt} is equated across the HRUs. This leads to an efficient outcome that maximises basin profit from surface water use.

Eq. (11) suggests the optimal level of G_{ijt} is where its marginal value in production, $P_j \cdot \frac{\partial f_{ijt}\left(W_{ijt}\right)}{\partial G_{ijt}}$, equates to the application cost of groundwater (marginal factor cost) G_{ijt}, $\frac{\partial WA_{ijt}}{\partial G_{ijt}} \cdot a_{ijz}$, and the opportunity cost of G_{ijt} in production elsewhere in the HRU,

$\lambda_{iw}\left(\frac{\partial h\left(S_{ijt}, G_{ijt}\right)}{\partial G_{ijt}}\right)$. Since groundwater is not traded in the water market, its opportunity cost is limited to the MVP of irrigated crops within the HRU. Equations (12) and (13) denote the requirement that resource constraints are exactly met.

5.3.4 Dynamic Programming Framework

The dynamic constraint on using groundwater can be represented as follows. The volume of groundwater pumped by HRU *i* in period *t*, G_{it}, must be less than its groundwater allocations available in that period, $\overline{Ga}_{it}$. Excess allocations that are unused can be rolled over to the next season, to a maximum of three consecutive years. In addition, in any one season a maximum amount of two-season's worth of allocations can be extracted (DLWC 2003). This only applies for groundwater resources, because surface water can not be banked in an unregulated system.

Given the inter-temporal nature of groundwater extraction, the optimisation of groundwater use across ten years through the control variable, G_{it} (HRU groundwater use in period *t*), is represented by the recursive Eq. (1) which is reproduced here for the convenience of the reader:

$$V_t\{G_{it}\} = \max_{G_{it}}\left[E\{\pi_t(G_{it})\} + \beta E\left\{V_{t+1}\left(\overline{Ga}_{it} - G_{it}\right)\right\}\right] \quad (i = 9, \ldots, 1) \tag{1}$$

Assuming a planning period of 10 years, the final value of stock remaining at the 10th (final) period:

$$V_{10}\left\{G_{i,10}\right\} = F\left\{G_{i,10}\right\} \tag{14}$$

And boundary conditions:

$$G_{i1} = \overline{Ga}_{i1} \tag{15}$$

$$\lambda_{10} = dF / d\overline{Ga}_{i,10} \tag{16}$$

Where the first boundary condition, Eq. (15), is the initial stock level which equals the available groundwater allocation in period one ($\overline{Ga}_{i1}$). The second condition, Eq. (16), is the inter-temporal value of the stock at the final period.

The available water allocation in each period t ($\overline{Ga}_{it}$) is the sum of seasonal allocations ($\overline{G}_{it}$) over three-consecutive periods:

$$\overline{Ga}_{it} = \sum_{t}^{t+2} \overline{G}_{it} \tag{17}$$

With a condition on the control variable, G_{it}, that the maximum volume used in one period is confined by two-consecutive period's water allocation:

$$G_{it} \leq \sum_{t}^{t+1} \overline{G}_{it} \tag{18}$$

And the necessary conditions for inter-temporal optimality:

$$\frac{\partial \pi_t(.)}{\partial G_{it}} = -\beta \lambda_{t+1} \left(\frac{\partial \overline{Ga}_{i+1}}{\partial G_i} \right) \text{ and } \lambda_t = \frac{\partial \pi_t(.)}{\partial \overline{Ga}_t} + \beta \lambda_{t+1} \left(\frac{\partial \overline{Ga}_{i,t+1}}{\partial G_{i,t}} \right) \tag{19}$$

This first term suggests that the immediate gains (losses) must equal to the present value (*PV*) of future losses (gains) in determining G_{it}^*. That is, the optimal choice of G_{it} should take into account the user cost: groundwater extractions should be increased until the marginal gains offset the *PV* of future losses, and vice versa, until the inter-temporal optimality condition holds. The second condition suggests that the optimal increase in the value of groundwater stock in period *t*, equals the additional period gain plus the discounted value of groundwater stock at period *t+1*.

5.3.5 Conceptualising Water Trade

In any given period, an irrigator can choose to buy or sell surface water through the water market. This decision can be conceptualised in the following way. An irrigator offers to sell some water on the market if the market price of water is greater than its MVP in irrigated agricultural production, and conversely the irrigator becomes a water demander at the margin if the market value of water is lower than the MVP in production (Figure 5.1). This buying and selling behaviour depends on each irrigators' derived demand for water, Q_d, at an exogenous price, P_w, and surface water allocation, Q_1, *ceteris paribus*. The mechanism for water trade for an individual irrigator is illustrated in Figure 5.1, with the gains from trade for the buyer denoted by area *ADE*, and for the seller the gains from trade are shown by area *ABC*.

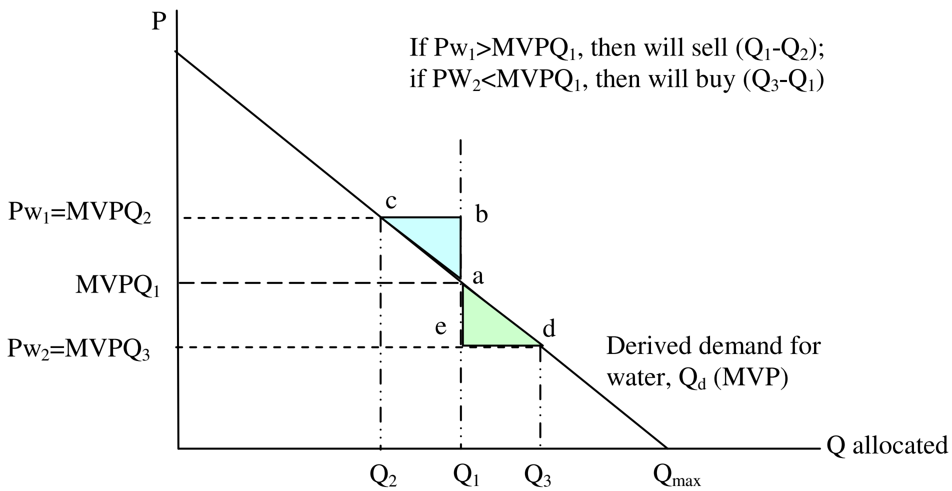

Figure 5.1: Irrigator's water demand and supply behaviour.

Despite the perceived benefits of water trade, many producers in the Mooki do not participate in the recently established water market because the purchase of water allocations is considered too expensive. Instead, some irrigators have commented that it is more cost-effective to invest in water saving technologies (Morgan 2005, pers. comm.). There are also perceived risks associated with offloading entitlements now and buying them back at higher prices in future. For the buyer, the entitlement only ensures a share of available flows of varying reliability, which reduces the expected value of the investment. For these reasons, water trading is regarded as unfavourable by many irrigators. However, if the benefits of temporarily trading water as a commodity could be credibly demonstrated, it may encourage trade, especially if it can be shown to augment the returns from investments in water efficient technology.

In this thesis, water prices are parameterised exogenously, such that market prices are unaffected by the purchasing and selling activity of irrigators in the catchment. The model takes the form of an interregional competition model as per Heady et al. (1973), whereby each irrigator has its own resources but can also compete for a shared resource, subject to transportation costs. In terms of water, the transportation cost could be represented as the water loss associated with 'transporting' water upstream or downstream. However, it is assumed that intra-regional water trade in a small catchment would be relatively efficient and the transport coefficient would be close to 100%. This interregional competition model is similar to spatial equilibrium models (Takayama and Judge 1971), except that the price here is exogenously set, rather than endogenously determined in the model. While an interregional trade model does not result in an equilibrium (endogenous) price of water in trade, the equilibrium water price can be represented by the shadow value of water when the price of water is set to zero, $P_w = 0$, since, at zero cost, the irrigator will continue to demand water until its value of the marginal product falls to zero. Where water supply is binding, its shadow value for the considered HRUs would represent the market-clearing price.

5.3.6 Internal and External Water Trading

An external trader is introduced to the model by including an additional term in the objective function, Eq. (2), to represent the value of water to an external agent:

$$\Pi_t\left(G_{it}\right) = \sum_{n=1}^{N}\left(\sum_{j=1}^{J} \pi_i(t) \cdot AJ_{it} - P_w Wd_{it} + P_w Ws_{it}\right) + \int f\left(P_w\right) dP_w \tag{20}$$

A linear factor demand for water is assumed for the external buyer, in the form of $f\left(P_w\right) = a - bP_w$, and the objective is to maximise the gain from water trade (internally) to other irrigators, or trade to the external agent. In the presence of an external buyer, equilibrium trade occurs where marginal value product (MVP) is equated across the irrigators and the external buyer of water:

$$MVP_i(W_{ijt}) = MVP_k(W_{kjt}) = \ldots = MVP_E(W_E) \tag{21}$$

Where W_E is the water demanded by the external agent.

5.3.7 *Water Allocation and Crop Choice*

The optimal allocation of water from a social perspective is such that the MVP of the last unit of water used for each crop is equated with all private and social costs associated with the crop produced. It is assumed that the primary irrigation water demand would be sourced from the river until the cost of using surface water, P_s, outweighs the cost of groundwater, P_p, or if surface water is limiting (Zilberman and Lipper 2002). The producer has the option of growing dryland crops in response to water shortages if it becomes the most profitable option. When the farmer decides to switch to dryland production, the operating cost associated with irrigation is eliminated from the objective function for that period. The fixed costs of the irrigation technology, however, would be sustained regardless of whether dryland or irrigated crop is produced because the investment has already been made. An additional condition has been defined to represent the dryland production option, whereby if *j=m* and *m* is a dryland crop, production costs become the fixed costs of the irrigation system plus other dryland production costs, i.e. the cost function in Eq. (3) becomes $C_{ijt} = \sum_{z}^{Z} FCI_{iz} + \{\text{dryland costs}\}$.

5.4 SUMMARY OF CHAPTER 5

The combined linear programming and dynamic programming economic model used in this thesis builds on theories of optimality, and is applied to the case study Mooki catchment in the Namoi Valley. The underlying assumptions of the model are justified based on the characteristics specific to the Mooki. The next chapter will provide a description of specific features of the Mooki basin that form considerations for the modelling framework.

Chapter 6.

CASE STUDY – THE MOOKI BASIN

A description of the case study considered in this thesis, the Mooki basin in the Namoi Valley, is presented in this chapter. Firstly, a geographical description of the Mooki basin is provided. This is followed by the various characteristics that are specific to this catchment and are addressed in this thesis.

6.1 GEOGRAPHY OF THE CASE STUDY AREA – THE MOOKI BASIN

The Mooki River basin is a tributary of the Namoi River Valley located in northern NSW, and forms part of the Murray-Darling Basin. The Mooki basin lies between the townships of Gunnedah and Quirindi, and includes the upstream reaches of Phillips, Warrah and Quirindi Creek water sources (Figure 6.1). These irrigation areas form part of the Upper Namoi and produce 21% of the irrigated production in Namoi, valued at AUD$526 million (11% of the value of NSW irrigated industry). This is despite total irrigated agriculture covering just 1.5% of the catchment (NSW Irrigators' Council 2001; Aluwihare et al. 2001; Trewin 2006).

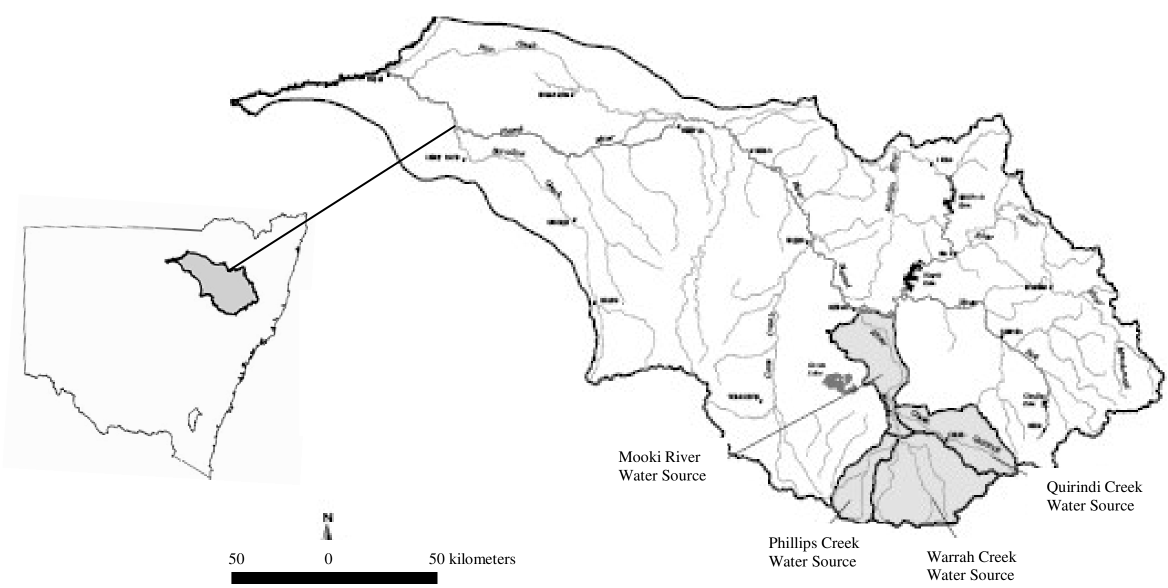

Figure 6.1: Location of the Mooki basin (shaded) in the Namoi catchment (Source: DIPNR 2004b).

Irrigators in the Mooki Basin hold unregulated licences, meaning there is no upstream head dam to 'regulate' flows downstream. As a result, producers in unregulated systems held area-based licences, and were confined by the area they can irrigate rather than the volume extracted. Due to the extreme variability of flows in the Mooki, irrigators make the most of passing flows by pumping as much water as possible whenever the opportunity arises. The median flow is 10ML/day, but for 25% of the year no flows occur in the Mooki, and for almost 20% of the year flows are above 100ML/day (Figure 6.2). Flows above 1,000ML/day are rare, occurring just 4% of the time although this is highly variable from year to year. Flows above 3,000ML/day occur less than 2% of the time (DIPNR 2004b). Given this distribution of flow, large on-farm storages are therefore common in the Mooki River basin and extraction occurs wherever there is enough water in the river. Without strict rules constraining individual irrigators' extraction level, there is a tendency for inefficient (too high) levels of water being extracted.

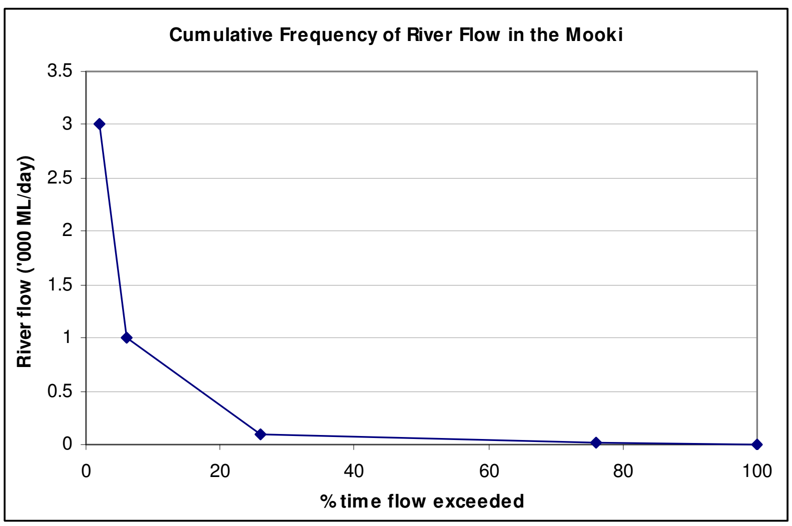

Figure 6.2: River flow in the Mooki (source: adapted from DIPNR 2004b).

As with many other catchments in the MDB, Namoi has recently had a number of environmental policies put in place. All catchments in NSW are now required to have Water Sharing Plans (WSP) for both surface water and groundwater, to limit extractions and to ensure fair distribution. With the introduction of the WSP, these area-based licences are converted into volume-based licences, essentially based on an estimate of irrigation requirements from its history of use. Surface WSP stipulates a minimum environmental flow requirement as well as a set of extraction rules for irrigators sharing the same hydrological system. Groundwater has been a significant water source for the region since the 1980s, during which groundwater entitlements have been offered as supplementary water to river extractions. However, due to the over-allocation, there are now plans to severely cut back the number of groundwater entitlements, up to 90% in certain zones. The groundwater WSP is to be phased in over the next ten years (CARE 2003).

The set of WSP rules governing the pumping of river flow varies depending on the irrigation area and on the level of flow. Essentially, pumping cannot commence until flows exceed a Commence-To-Pump (CTP) level, which amounts to imposition of an environmental flow rule. For the Mooki Water Source, the CTP is 100ML/day before extractions may begin; for Phillips Creek Water Source and Quirindi Creek Water Source, the CTP is 2ML/day; for the Warrah Creek Water Source the CTP is 4ML/day (DIPNR 2004b). However, flows above this level could be fully extracted. While the limits on individual extractions should mean equal share of flows, the uppermost irrigator inevitably has priority to available flow (up to his/her daily extraction limit) and less becomes available to downstream irrigators (Powell 2006, pers. comm.). These are considerations that enter into biophysical simulations of surface water access by different areas within the Mooki.

The total area of the Mooki Basin as reported in DIPNR (2004b) is 3,741km^2 (374,100ha), which is somewhat smaller than the GIS referenced area in the biophysical model of 4,525km^2. This discrepancy is due to the uncertainty with regards to the actual size of the catchment, since in flood years surround water course flow into the Mooki and affect the delineation of the catchment boundary (Vervoort 2006, pers. comm.). However, the difference in catchment boundary does not affect the landuse distribution, which was based on 2002 data from DIPNR as shown in Table 6-1. It was assumed that all irrigated areas are producing cotton, which makes up 397 km^2.

Table 6-1: Landuses in Mooki Basin (source: Vervoort 2006, pers. comm.).

Landuse	Area of Basin (ha)
Pasture	273,939
Row-crops	105,759
Irrigated crops	39,700
Forest-mixed	22,631
Urban	6,690
Water	3,426

6.2 COTTON PRODUCTION IN THE NAMOI VALLEY

The first irrigated cotton operation in Australia began in the Namoi Valley during the 1960s (Thomson 1979), which has become the second largest cotton growing region in Australia (CCCCRC 2007). Along with the production of cottonseed oil, a significant oil seed crop in Australia, the cotton industry is regarded as the highest value crop in NSW (Anthony 1998). However, it is also one of the highest water consuming agricultural industries. More than 84%of Australian cotton is grown under irrigation, which accounts for about 1,819GL of agricultural water use in 2004-05 (Cotton Australia 2007; Trewin 2006). This represents 18% of irrigation water use in Australia, making it the second largest consumer of water following pasture for grazing (28.7%). On a per hectare basis, cotton also rates as the second highest consumer of water (Trewin 2006). The average gross value of cotton per megalitre is lower than sugar but higher than rice, which are also water intensive industries (Table 6-2). While the cotton industry in Australia has a good reputation for being the highest yielding and most water efficient in the world (Tennakoon and Milroy 2003), there remains the case for improving the industry's water use efficiency.

Table 6-2: Water use and value of irrigation industries in Australia (source: Trewin 2006).

Crop	Average volume applied (‘000ML)	Average application rate (ML/ha)	Average gross value ($/ML)
Rice	618.9	12.1	163
Cotton	1,819.3	6.7	519
Sugar cane	1,171.9	5.5	836
Fruit trees	177.3	5	-
Pasture for grazing	2,896.5	3.4	-

Of the six largest cotton producing regions in Australia, the highest crop water use efficiency was found to be 3.2kg/ha/mm in the Darling Downs, and the lowest was 2.0kg/ha/mm in the Namoi (Tennakoon and Milroy 2003). This translates to 9.8 bales/ha for Darling Downs and 6.2 bales/ha for Namoi, assuming crop water requirements of 7ML/ha. This derived yield for Namoi is very low compared to figures reported in Boyce (2005), which reported yields of 9.1-10.3 bales/ha for upper and lower Namoi. Anecdotal evidence also suggests that irrigators in Namoi get around 8-10 bales/ha, applying 7-8ML/ha of irrigation water (Norrie 2006, pers. comm.). Based on these figures, it appears that cotton yield in the Namoi can range between 6-10bales/ha, with a water consumption of around 7ML/ha (Tennakoon and Milroy 2003).

6.3 DEEP DRAINAGE IN THE NAMOI VALLEY

As discussed in the earlier chapters, salinity is closely linked to groundwater hydrology. The cultivation of shallow-rooted crops and clearance of native vegetation has increased the level of deep drainage, leading to saline shallow watertables. Rising watertable levels can contribute to soil salinisation and saline deep aquifers (Mawhinney 2005). Deep drainage refers to water that moves below the maximum effective plant zone, having not been absorbed by vegetation, gradually filling (recharging) shallow aquifers and bringing salts to the surface. Saline water can also seep from the ground and permanently intersect with the base of the river as return flow (UNL 2007; DEWR 2007).

Some drainage is required to carry salts out of the soil, and the rate of deep drainage should be kept to the natural contribution from rainfall. However, where this natural rate is exceeded, excessive levels of drainage occur and contribute to rising watertable and increased salinity levels. The dynamics of salinity, however, makes it difficult to establish exactly how deep drainage contributes to salinity, and how long it takes to reach a new equilibrium; soils with high drainage rates can take as short as one year to achieve equilibrium, but low draining soils can take many decades (Jolly et al. 2001). This time lag creates difficulties in establishing the damage drainage causes, given that by the time an increase in salinity is detected the system has already shifted to a new equilibrium, and the benefits of reducing drainage now will not be realised for several decades after (Silburn and Montgomery 2005).

The amount of deep drainage which occurs in the Namoi, and other catchments, is uncertain. It is generally assumed that deep drainage can be close to zero for heavy clay soils in the northern inland areas. Namoi soils (sodic grey vertisols) have low, but not insignificant, drainage due to lower permeability (Silburn and Montgomery 2005). The average drainage per irrigation is 0.015ML/ha (1.7%) for Namoi, which is very low relative to the adjacent Gwydir valley (14%). The review in Silburn and Montgomery (2005) shows annual deep drainage for the Namoi ranging from 0.03-9 ML/ha per year, although averaging around 1-2ML/ha per year.

The interception of groundwater aquifers with the river is not thought to occur in the Mooki, since the shallow and deep aquifers sit far below the stream bank and rarely rise enough to 'return flow' to the river (Lavitt 1999; Vervoort 2007, pers. comm.). Deep drainage does not become groundwater supply in the Mooki, due to the segregated shallow and deep groundwater layers. The shallow aquifer sits on an unconfined layer above the deep aquifer which is where groundwater pumping occurs (Table 6-3). Deep drainage, however, recharges the shallow aquifer which has a very high salinity and is unsuitable for irrigation. Where there is significant drawdown on the deep aquifer due to pumping, some shallow aquifer water could leak into the deep aquifer, potentially increasing the salinity level (Vervoort 2007, pers. comm.). For the Namoi region, the time lag between deep drainage and recharge to the shallow aquifer was found to be relatively short (Karen Ivkovic 2005, pers. comm.). This suggests that any changes in deep drainage will be realised quickly. While the exact nature of the interaction between deep drainage and the groundwater aquifers are up to conjecture, the focus should be to reduce deep drainage thereby salinity risk. For greater detail on groundwater hydraulics, see Appendix A.

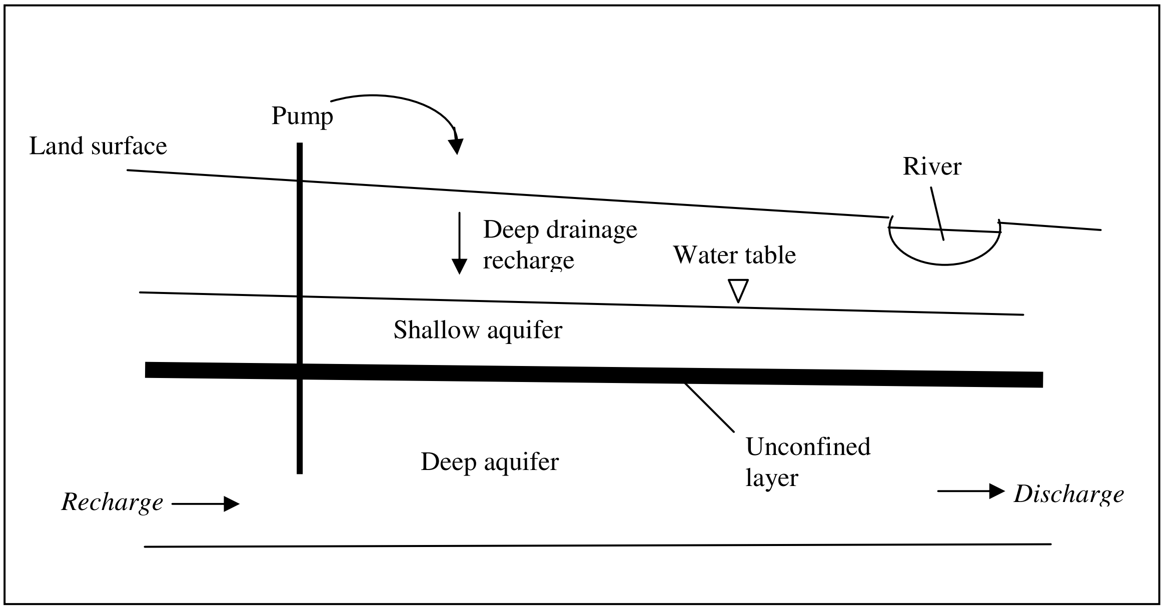

Figure 6.3: Shallow and deep aquifer interaction in the Mooki (source: Vervoort 2007, pers. comm.)

6.4 SALINITY IN THE NAMOI VALLEY

Depending on the standard that is considered, salinity in Namoi can either be below target or significantly over. According to the Australian and New Zealand Environment and Conservation Council (ANZECC) recommendations, 330µS/cm is the limit for healthy ecosystem protection[2]. However, the Namoi Catchment Blueprint allows a higher target of 550µS/cm as a more realistic target due to the history of human activity in the catchment (DLWC 2002), which is in line with keeping the end-of-valley salinity target at Morgan[3] (in South Australia) below 800µS/cm. Salinity readings can also be expressed as decisiemens per metre (dS/m) or as a salt load according to the following equation:

$$\textit{1 dS/m = 1,000 EC (or µS/cm) = approx. 640 mg/kg}$$

[2] µS/cm is the electrical conductivity (EC) reading for salinity concentration in water.
[3] Morgan is where the salinity measurement is taken, upstream of the mouth of the river Murray in South Australia.

That is, one megalitre with an electrical conductivity (EC) of 1,000µS/cm contains about 640kg of salts[4]. The soil salinity that results depend on the class of soil; the slower draining, the higher soil salinity is because the salt content of water seems to be retained in the soil. Poorly drained clay soil can become three times as saline as the water applied, so the water salinity limit (the maximum EC of irrigation water to avoid losses in crop yield) should be about one-third of the soil salinity. For example, for very slow draining soils, the water salinity limit is 330µS/cm given a crop EC tolerance of 1,000µS/cm. (NSW DPI 2006b).

A comparison of median EC at three locations downstream of major cotton-growing areas, at Namoi River (Bugilbone), Mehi River (at Bronte), and Barwon River (Mungindi) is presented in Figure 6.4. This figure shows highest median EC readings at Namoi Valley in most years (Mawhinney 2005).

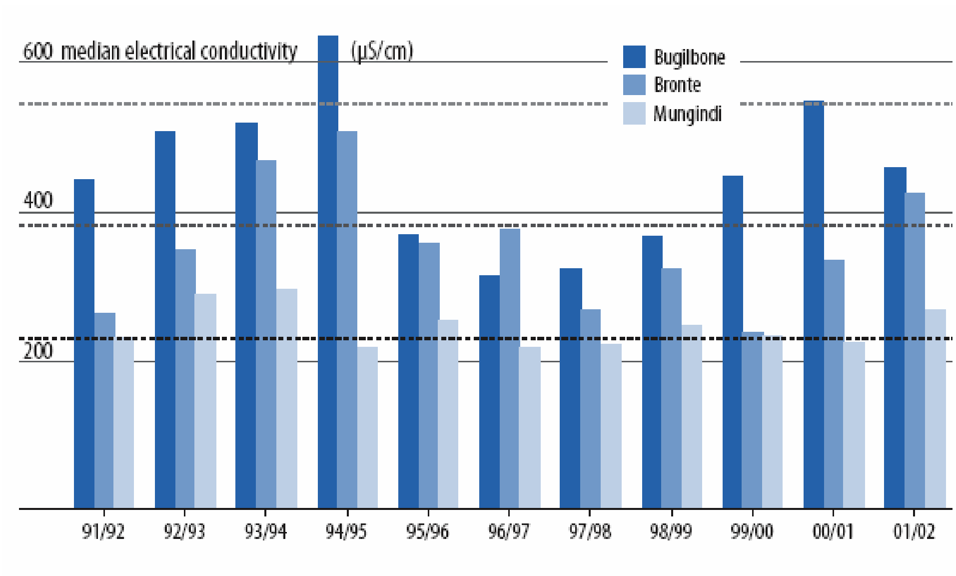

Figure 6.4: Mean EC for major cotton growing valleys (Source: Mawhinney 2005 p.269).

While Namoi is relatively saline, currently there is little impact on agriculture within the catchment. The cost of salinity to agricultural producers in the Namoi, Gwydir and Border Rivers region totals $6Mill per year, majority of which is associated with

[4] The salt load can vary between 600-800 mg/kg depending on the chemical composition of water.

infrastructure and maintenance costs. Less than $2Mill of this is attributed to losses in agricultural income (Wilson and Ivey 2001). One reason for a salinity target in the Namoi is to ameliorate the downstream impact of salinity discharged from Namoi on the Barwon-Darling system, which flows into the Murray-Darling. However, given the relatively high salt tolerance of cotton, which has a salt tolerance level of 1,700µS/cm, there is little incentive for cotton irrigators in Namoi to internalise their salinity contributions into their production decisions. However, river salinity as high as 1,000µS/cm has been recorded in Gunnedah on occasion, which could cause crop losses of up to 10% if irrigation is conducted during the seeding stages (NSW DPI 2006b).

The median reading for the Mooki over 2003-06 was 534µS/cm, which suggests that for every megalitre (ML) of deep drainage, 342kg of salt is carried into the river (assuming a one-to-one ration between drainage and salt load). This reading places Mooki within the target salinity level specified in the Namoi Catchment Blueprint of 550µS/cm, with the highest median EC reading recorded in Mooki River (Breeza) and Coxs Creek (Boggabri) over 2000-2001.

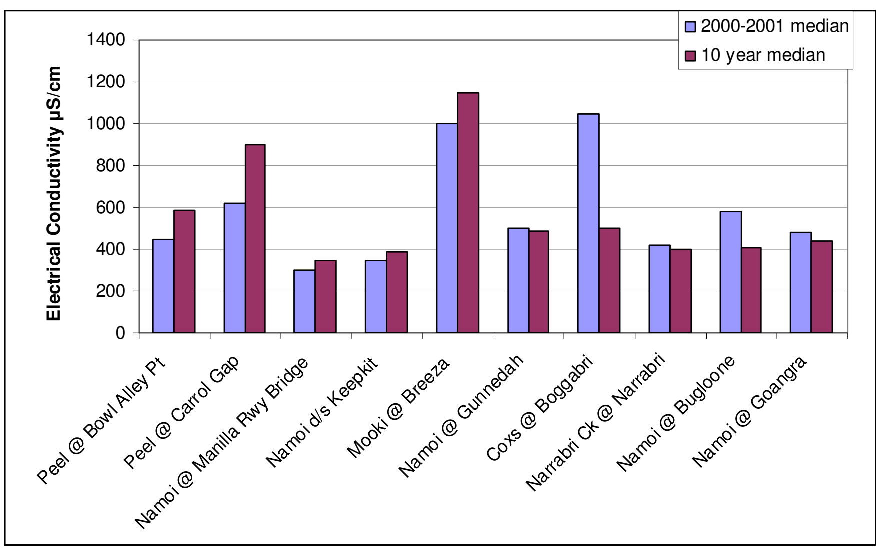

Figure 6.5: Median electrical conductivity in the Namoi Catchment (source: DLWC 2002)

In terms of salt load, the end-of-valley target for the whole of Namoi at Goangra is 127,600t/yr. This Blueprint salt load target suggests that the maximum level of deep drainage for the whole of Namoi should be:

$$\frac{127,600,000kg}{534\mu S / cm} = 238,951 ML / yr$$

For the Mooki itself, the salt load measurement is 3,000t/yr which translates to 5,618ML of deep drainage assuming an EC of 534µS/cm. This was an exceptionally low reading, attributed to the low flows experienced during the 2003/04 season (DLWC 2002). During normal flow years, the salt load contribution from the Mooki is expected to be higher.

6.5 GROUNDWATER ENTITLEMENT REDUCTION

Irrigators in the Namoi have developed a reliance on groundwater as a result of early water policies. Conjunctive licences to withdraw groundwater were initially handed out to irrigators to alleviate water shortages from severe drought in 1983-84, which subsequently remained as a staple water source (Haisman 2005). Due to policies set at the time, which effectively implied 'mining' of the groundwater resources, the number of extractive groundwater licences became over-allocated. This led to severe declines in groundwater levels that were noticeable within three years of distribution (Hamparsum 2004). Irrigators had become aware of the depletion and agreed to voluntary cuts to entitlements of up to 35% in 1995. It was not until the end of the 1990s that the NSW government had moved towards sustainable management of groundwater resources, eventually resulting in the groundwater Water Sharing Plan (WSP) in 2002 following from the writing of the Water Management Act 2000 (CARE 2003). The groundwater WSP was gazetted in December 2002, although it did not commence until the 2004-05 irrigation season (CARE 2003). The intention of the groundwater WSP was to limit the aquifer access licences (AAL) to its 'sustainable' recharge level determined for each Zone, such that its use can be maintained indefinitely without depleting the resource. The estimated sustainable extraction rate will be reviewed and adjusted if necessary.

However, it stands to reason that all recharge estimates are subject to significant uncertainty. It has also been suggested that there is no such thing as a 'sustainable' yield that can be indefinitely maintained. This is due to long-term effects on deep aquifer discharge and recharges (see Figure 6.3), which changes from its equilibrium state when groundwater is pumped (Alley and Leake 2004). Groundwater pumping may increase the upstream recharge rate while reducing the downstream discharge rate, which affects the groundwater supply to irrigators at both ends of the aquifer system. A more appropriate term would be groundwater 'capture', which implies there are third-party impacts upstream or downstream of the pumping site (Bredehoeft 2006). In this sense, there is a trade-off between upstream and downstream use of groundwater that should be factored into economic decisions. Nevertheless, the concept of sustainable recharge has been used by the Department of Natural Resources for its WSP determinations based on simulations of rainwater infiltration, without consideration for changes in groundwater equilibrium from pumping the 'sustainable' recharge (Vervoort 2007, pers. comm.).

Each groundwater source area is divided into zones; the Upper Namoi (UN) covers 3,800km^2 upstream of Narrabri, containing 12 zones. The Lower Namoi (LN) is managed as one zone, covering 7,630km^2 downstream of Narrabri to Walgett (Figure 6.6). Groundwater serves as the main source of irrigation water in UN although surface water is preferred due to lower pumping costs (CARE 2003; Morgan 2005, pers. comm.).

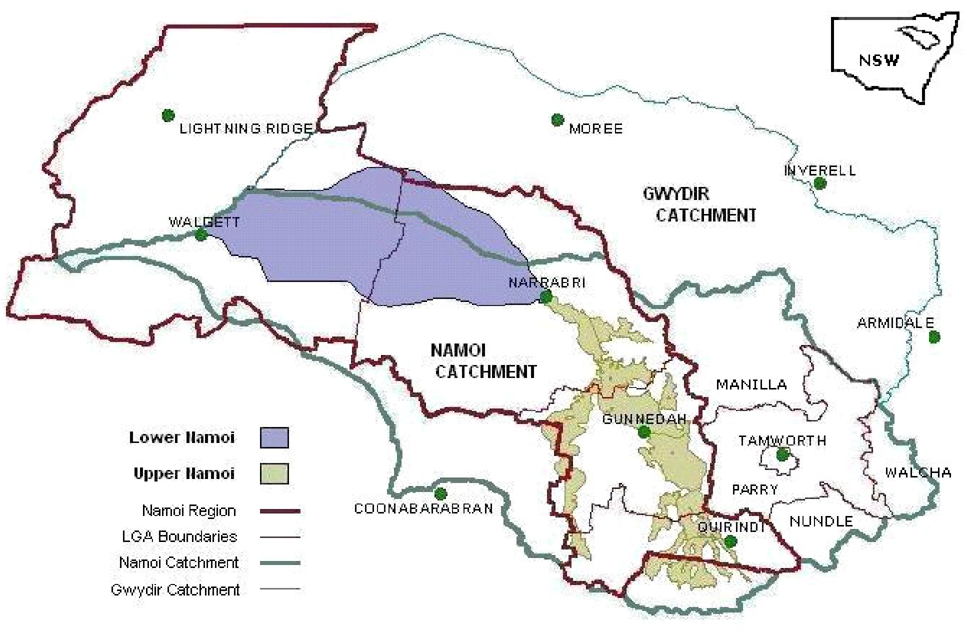

Figure 6.6: Upper and lower Namoi (Source: CARE 2003).

A majority of irrigators in the Namoi are expected to have their groundwater entitlements (AAL) cut by between 41-87% in the UN, including Mooki and its upstream reaches, and 51% in the LN region. While these cuts are significant, the intention of the groundwater WSP was to limit the aquifer access licences (AAL) to its sustainable groundwater recharge level determined for each Zone. The reduction to entitlements is zone-specific, with greater reductions in some zones than others (Figure 6.7 and Table 6-3). The entitlement reductions to be made are inclusive of the voluntary cuts of 10-35% in the mid 1990s (CARE 2003).

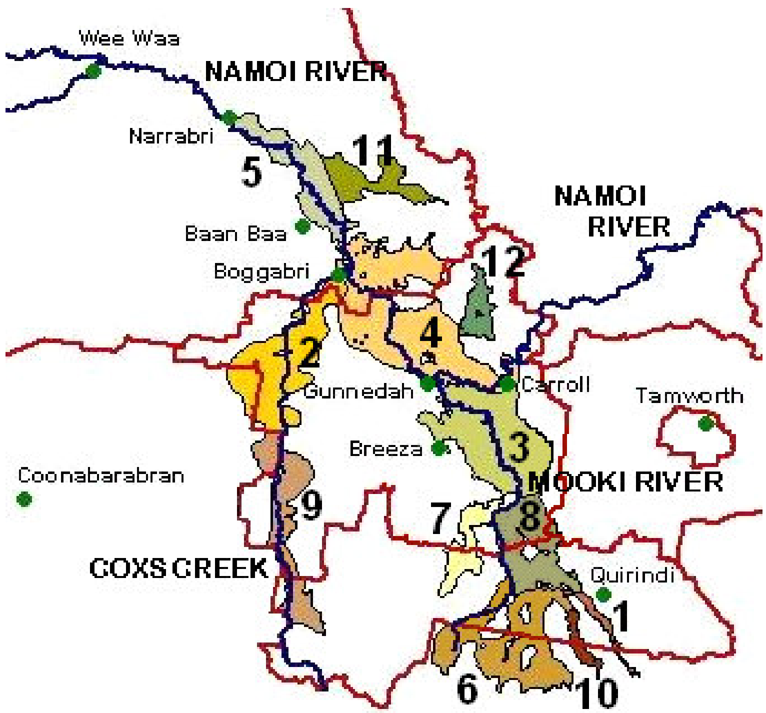

Figure 6.7: The Upper Namoi zones (source: CARE 2003).

The amended access licences under the groundwater WSP are based on a reduction of the yearly entitlements on existing licences. Essentially, the amended AAL is a portion of the total available recharge after high priority allocations (TWS) have first been met. At the start of each water year (in July), available water determinations are announced for each water source for AAL and supplementary licences and accredited to the licence holder's water account.

Table 6-3: Reduction in entitlements in the Mooki Basin (source: Aquillina 2003 p.13).

Zone	Reduction (%)
1	87
3	69
6	-
7	41
8	67
10	-

However, there remains no record of groundwater allocations on the public register of water licences (DNR 2007), even though the groundwater WSP has been gazetted since 2002. In this thesis, it is assumed that the available sustainable recharge remains unchanged and is fully allocated during the planning period. Therefore, while the term ‘AAL’ refers to groundwater entitlements, in this thesis its meaning is synonymous with the share of groundwater recharge or groundwater allocations. These terms will be used interchangeably when referring to groundwater allocations. The estimated annual recharge determined in the groundwater WSP for all zones is shown in Appendix B. For the relevant zones in this thesis, which lie within the Mooki Basin, the share of recharge (net of TWS) is shown in Table 6-4.

Given the significant reduction in entitlements, there are obvious implications for irrigators in the Mooki. Supplementary water allocations (SPW) were issued to irrigators with a history of use greater than their new share component to reduce the economic impact, given by history of use minus the amended access licence share component. These supplementary licences will be reduced after year 5 and phased out completely after year 10 of the WSP. During this time, financial assistance such as incentives for on-farm water use efficiency investments, business diversification, farm investment plans or purchasing additional licences will be available.

Table 6-4: Groundwater allocations (AAL), supplementary water (SPW) and extraction limits.

Zones	Gw AAL	Gw SPW		Gw Total		% Share Total Allocation
		Yr 1-5	Yr 6-10	Yr 1-5	Yr 6-10	
1	992	1,071	669	2,063	1,661	4
3	16,338	7,645	4,247	23,983	20,585	43
6	6,915	-	-	6,915	6,915	12
7	2,810	205	205	3,015	3,015	5
8	13,794	5,693	3,163	19,487	16,957	35
10	949	-	-	949	949	2
			Total	56,412	50,082	100

While these policies have the objective of assisting irrigators to adjust to the recent changes, and to reduce the social impact of reductions in groundwater entitlements, it appears to contradict the objective of achieving greater allocative efficiency in the water economy. Reducing groundwater entitlement according to history of use is uneconomic and unfair because it rewards those who were inefficient. Furthermore, financial assistance for investments in water efficiency (such as irrigation systems) or purchases of water licences provide distortionary signals and allow for inefficient irrigators to remain in operation. This precludes the reallocation of water to its highest value use.

Banking allocations, which was previously not permitted, has also been allowed in part to compensate for the cut-back in groundwater entitlements. Unused volumes can be carried over to the following year to a maximum of 300% of the share component, and for one water year a 200% extraction of the share component is permitted. Basically, in any year the maximum groundwater use is the two-season allocation, and in any three-rolling years total extractions must not exceed the three-year allocation. This banking rule also applies to surface water allocations; however due to the uncertainty in river flow irrigators in the Mooki Basin generally use the entire allocation within one water year (Hamparsum 2006, pers. comm.).

6.6 WATER CHARGES

A surface Water Sharing Plan (WSP) has been gazetted for the Mooki Basin in 2004 and there are now 26 Water Access Licences (WAL) under the current Water Management Act. The first Available Water Determination (AWD) was made on 1st July 2004 (Powell et al. 2003). AWD is the volume of water available for extraction at the start of a water year. This is expressed as either a percentage or volume per unit of share component. For example, the AWD for the 2004 water year for 'Unregulated River' licences was 2 megalitres (ML) per unit share held by the irrigator (Table 6-5).

Although no gauges are in place between Breeza and the downstream gauge at Ruvigne, the extraction rules are enforced through a penalty system. If the downstream gauge records flow levels below the stipulated CTP, it indicates that irrigators in the basin are 'cheating' and over-extracting above their licensed volume. As a penalty, if the averaged extraction across three years exceeds the stipulated long-term extraction limit by 5% or more, the extraction rules will be revised downwards. Irrigators therefore have incentive to comply, because otherwise the river access licences will be reduced by an amount that brings total water extraction back in line with the long-term extraction limit (Powell 2005, pers. comm.; DIPNR 2004b).

Table 6-5: Water Access Licences in Mooki water source under Water Management Act 2000 (DIPNR website 2005b)

Access Licence Category	No. of WAL	Total Share Component	Share Component Unit	Cumulative AWD	Cumulative AWD Unit	Water made Available (ML)
Domestic and Stock	3	23.5	ML	200	% of Share Component	47.0
Local Water Utility	0	0	ML	0	% of Share Component	0.0
Domestic and Stock	9	82.0	ML	200	% of Share Component	164.0
Unregulated River	26	29,526.5	unit shares	2	ML per share	59,053.0

WAL holders are charged the greater of a base charge of $54.31/yr, a two-part tariff consisting of an entitlement charge and usage charge per ML actually extracted, or a volume of entitlement charge (dollars per unit share or ML). The maximum two-part tariffs which applied for 2005 onwards are shown in Table 6-6, indexed to the CPI (IPART 2005; IPART 2006). For the Namoi, a two-part tariff applies to all irrigators.

Table 6-6: Water charge for 2005-06 onwards (source: IPART 2005 p.9).

	Maximum annual charges				
	Area based charge ($/ha)	Volume of entitlement charge ($/ML)	Two-part tariff: entitlement ($/ML of entitlement or $/unit share)	Two-part tariff: Usage ($/ML)	Usage charge only (local water utilities and major utilities ($/ML)
Border	12.26	3.82	2.3	1.53	1.72
Gwydir	12.26	3.82	2.3	1.53	1.72
Namoi	12.26	3.82	2.3	1.53	1.72
Peel	12.26	3.82	2.3	1.53	1.72
Lachlan	13.57	3.07	1.85	1.24	1.88
Macquarie	13.57	4.52	2.71	1.8	1.88
Far West	13.57	2.07	1.26	0.84	1.88
Murray	7.72	3.09	1.85	1.24	0.97
Murrumbidgee	13.57	5.43	3.26	2.16	1.88
North Coast	13.57	4.1	2.47	1.65	1.88
Hunter	11.75	2.65	1.6	1.07	1.63
South Coast	13.57	3	1.8	1.2	1.88

Groundwater licences are also subject to entitlement and usage charge indexed to the CPI. The applicable charge for the extractor depends on whether they are located in Groundwater Management Areas (GMA) that are metered, or in non-managed areas that are not metered. Irrigators within a GMA are subject to a base charge per property plus entitlement and usage charges, while non-GMA extractors are subject to a base charge per property plus an entitlement fee (IPART 2005). The groundwater management plan for the Mooki has not been finalised (NWC 2005), which suggests that the groundwater source is non-GMA and irrigators only pay a fixed cost for water (Table 6-7). This is discussed further in the following chapter.

Table 6-7: Charges for ground water access licence holders in the Namoi (source: IPART 2005 p. 21).

	Base charge ($/ property)	Entitlement charge ($/ML)	Usage Charge ($/ML)
GMA	187.72	0.85	0.43
Non-GMA	81.48	0.85	-

6.7 IRRIGATION SYSTEMS: PIVOT v DRIP

The main irrigation systems used in the cotton industry is furrow, pivot and drip irrigation. In NSW, the total area under drip systems is 54,000ha and for pivot irrigation systems is 63,000ha, while furrow irrigation occupies 678,000ha (Trewin 2006). Relative to 910,000ha of irrigated area, drip and pivot represent 5.9% and 6.9% of irrigation systems used. In Australia, the proportion of cotton irrigators that have adopted water efficient irrigation technologies is even smaller. Less than 4% of Australian cotton crop is grown under large irrigation machines, while less than 2% are grown under drip systems (Raine et al. 2000; Foley and Raine 2001). There are different reasons for the low adoption rate of water efficient technologies.

One of the reasons why pivots are not more commonly used is due of land constraints, which restrict irrigation system choices to furrow or require expensive earth works to prepare the land for pivot systems. There are also maintenance problems with pivot systems; in particular there is a steep learning curve to operate the machine (Figure 6.8a and b). The water savings, however, makes investing in pivot a more cost-efficient alternative to purchasing water entitlements. Field experiments have shown yield improvements while using 35-37% less water compared to furrow systems, due to greater control over water application (Foley and Raine 2001).

Due to the high costs of installation and maintenance, drip irrigation systems have generally not been favoured. The system itself costs around double that of pivot systems, but with a relatively short lifespan. It also needs to be removed every year when the cropping area is being prepared or cultivated. While anecdotal evidence suggests that drip irrigation allows for significant water savings, there is only a small yield increase to offset the capital investment. The capital is thought to be better spent on purchasing water entitlements (Morgan 2005, pers. comm.; O'Halloran 2005). Therefore, while the level of water savings is beneficial, unless there is chronic water shortage it does not appear to be cost-effective.

Figure 6.8: (a) Land prepared for centre pivot irrigation systems (source: MDBC 2006c); (b) centre pivot or lateral move machine (source: Foley and Raine 2001)

6.8 SUMMARY

The characteristics inherent to the case study, the Mooki basin, have been presented in this chapter. This was to clarify the assumptions that have been made in the modelling approach, and to highlight the environmental and resource concerns relating to this particular catchment. In the following chapter, the data and methodology used to carry out the analysis in this thesis is presented.

Chapter 7.

DATA AND METHODOLOGY

In this chapter, the aim is to describe the steps taken to carry out the analysis based on the case study basin, the Mooki, using the conceptual model developed for this thesis. The data assumptions for this study, the methodology used and the different scenarios simulated for the analysis are discussed in detail.

7.1 THE BIOPHYSICAL MODEL: SOIL AND WATER ASSESSMENT TOOL

Available Geographical Information System (GIS) and geophysical data specific to the case study Mooki were used for this research. The GIS used include Digital Elevation Model (DEM) data (Geosciences Australia) and soil data layer (University of Sydney Database and Department of Natural Resources – DNR), agricultural management data (NSW DPI), precipitation data and other climatic data (Commonwealth Bureau of Meteorology – BOM), and stream flow data (DNR). Land use data were also derived from a land use survey by DNR. In SWAT, the catchment was partitioned into 32 sub-basins, defined as a unique collection of streams that drain to a single outlet. Together, the 32 sub-basins contain 608 hydrologic response units (HRUs). An HRU is defined as a homogeneous land unit with a specific soil type and land use. A GIS image of the modelled catchment is given in Figure 7.1.

The total number of irrigated cotton HRUs referenced is 53, making up total area of 397km^2. These HRUs are scattered across three irrigation areas referenced as Ruvigne, Caroona, and Breeza. Each of these irrigation areas roughly corresponds with the Mooki River Water Source, Phillips and Warrah Water Source, and Quirindi Water Source in the surface Water Sharing Plan, respectively.

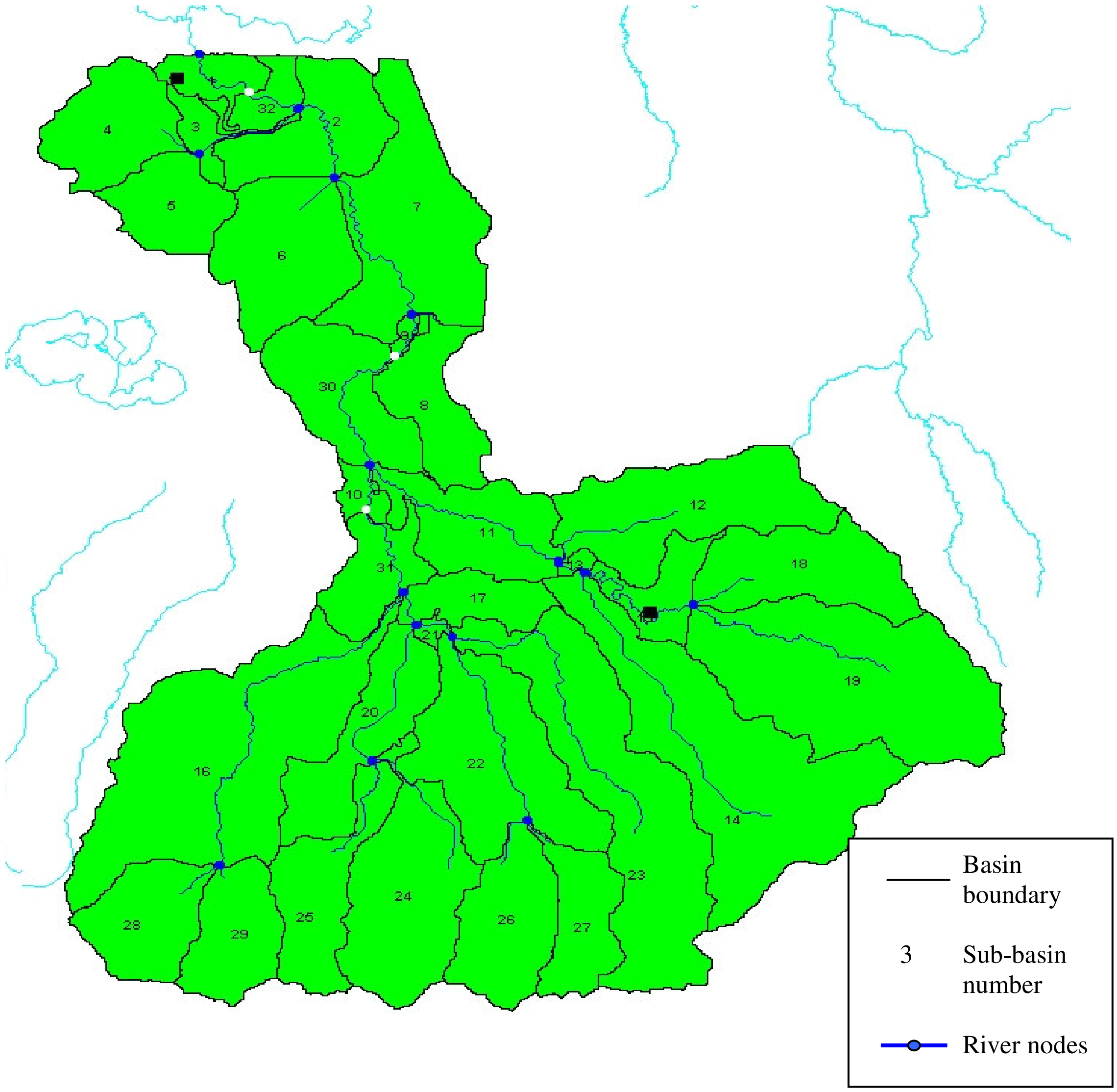

Figure 7.1: GIS delineation of the Mooki.

Within Ruvigne, irrigated cotton HRUs are situated in the downstream sub-basins 2-5, 7, 11, 31, 32; in Caroona, these HRUs are situated in the upstream sub-basins 16, 22-27; and in Breeza the HRUs are located in upstream sub-basins 15, 18, 19. Each of the three irrigation areas has a 'node' in SWAT that measures how much water is available for HRUs within the irrigation area (Figure 7.2). The scope of the analysis is narrowed to these 53 irrigated cotton HRUs, each of which is regarded as an individual farm. This assumption was made in line with the conjecture that soil types dictate the crops that are grown, so it is reasonable to assume that these land parcels could be modelled and are indeed as individual farms, even though they may be a part of a larger farm in a cadastral sense, or more farms might be situated in an HRU.

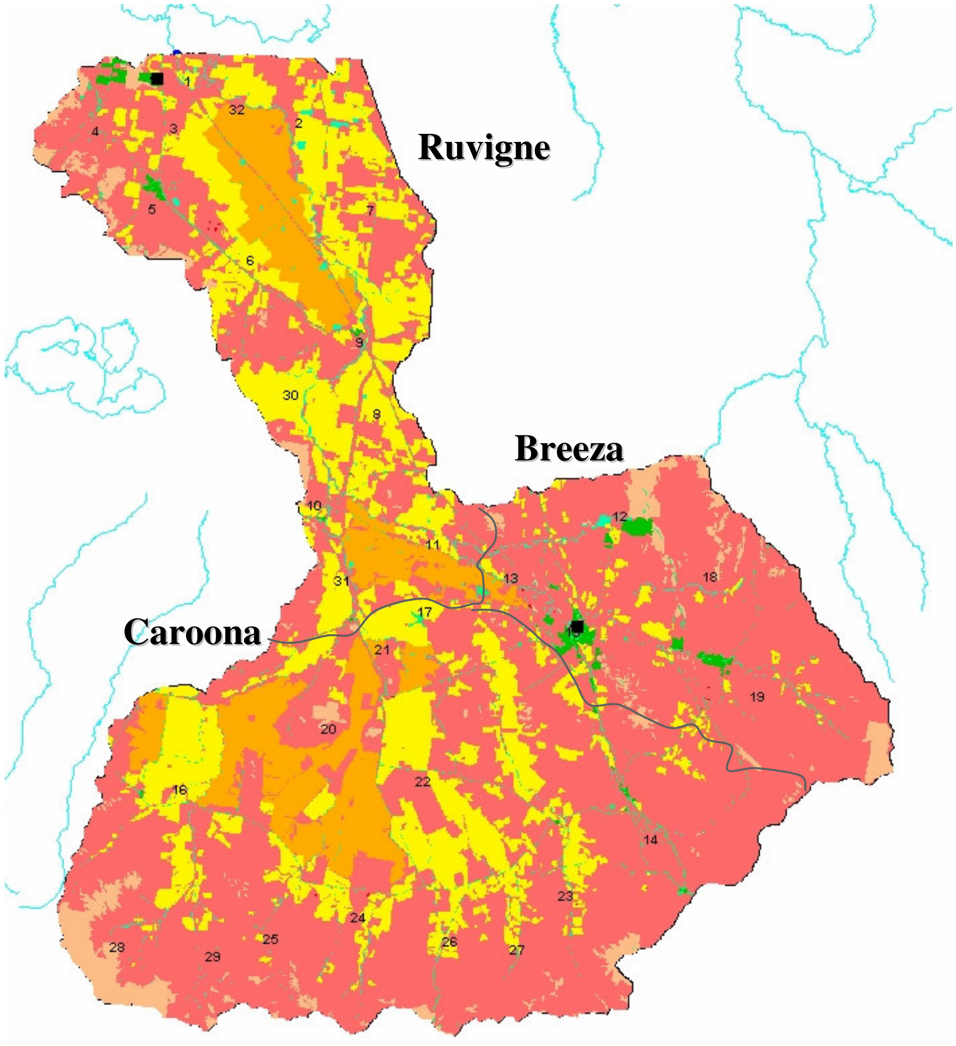

Figure 7.2: SWAT delineation of irrigated areas in the Mooki Basin (in orange).

Simulations were run in SWAT over ten years to obtain production outcomes for each HRU under the simulated activities outlined in Table 7-1. For each production activity, the SWAT model generated biophysical information, significantly relating to water use, associated deep drainage, and yield. The yield obtained for the HRUs generated in SWAT was weighed against the average yield over the HRUs, to reflect its relative productivity. This relative yield was then fitted to a discrete probability distribution of yields for corresponding northern NSW crops, based on the period 1965-2005 (ABARE 2005). The distributions for the crops under consideration – irrigated cotton, dryland cotton, dryland wheat, and grain sorghum – were used.

Table 7-1: Set of possible production activities for each HRU.

Activity number	Crop production	Source of water	Irrigation[5]
1	Cotton	Surface	Furrow (120mm/10,000m^2)
2	Cotton	Ground	Furrow (120mm/10,000m^2)
3	Cotton	Surface	Pivot (50mm/10,000m^2)
4	Cotton	Ground	Pivot (50mm/10,000m^2)
5	Cotton	Surface	Drip (according to soil deficit)
6	Cotton	Ground	Drip (according to soil deficit)
7	Grain/Sorghum	None (Dryland)	None
8	Wheat	None (Dryland)	None
9	Dryland Cotton	None (Dryland)	None

The biophysical information generated from SWAT were used as an input into a catchment level mathematical programming model with an objective to maximise net social benefit from the HRUs, subject to environmental constraints and constraints on water availability. The revenue, variable and fixed costs, and profit were calculated for these outcomes using price data obtained from NSW Department of Primary Industry Budget Sheets (NSW DPI 2006a). The revenue for cotton was given by income obtained from sale of cotton lint and cotton seed, and the cost was given by a per hectare cost of production ($OtherCosts_j$ in Eq. (1)) plus variable costs associated with irrigation. These variable costs include usage charge per megalitre (ML), taken from the 2005-2006 IPART water price determination[6], and pumping costs under furrow irrigation per ML according to figures from NSW Department of Primary Industries. The pumping cost varies depending on the source of water used, with groundwater being more expensive due to higher pumping costs.

The water availability constraint was set according to Eq. (4), and the deep drainage (DD) constraint was set according to Eq. (10). The environmental flow constraint, *CTP*, was imposed by setting a command in SWAT which requires that a specified minimum river flow level is present before water extraction for irrigation can begin. This was set

[5] 100mm per 10,000m^2 = 1ML over 1 hectare
[6] The current pricing arrangements do not factor in the water scarcity rent in pricing water, and is priced only for cost-recovery of water services.

according to the surface Water Sharing Plan (WSP), depending on the water source in which the HRU is located. The value of these constraints was then varied (parameterised) to determine the impact of various environmental policies on the basin economy.

7.2 SIMULATION SCENARIOS

Simulations of the economic model were run using the optimisation program, 'What's Best!' (Lindo Systems 2007), with the objective to determine the profit-maximising combination of choices for the whole basin (including crop choice, source and quantity of water in irrigation, irrigation systems used, and amounts of water traded), given resource and environmental constraints. Each scenario was designed to address the research questions framed in Chapter 1. There were four separate treatments under which every simulation scenario was run. The first treatment is the Status Quo (Treatment One), which was simulated to establish a baseline to be used for comparison. Under this treatment irrigators only undertake furrow irrigation, and use only the water allocated initially without the opportunity to trade water. The second treatment (Treatment Two) simulates a different technological setting. Irrigators are able to use alternative irrigation technologies (furrow, pivot irrigation, or drip). However in this treatment, there are no opportunities to trade water. In the third treatment (Treatment Three), alternative irrigation systems (AIS) can also be used, and in addition there is the opportunity to buy and sell water in a water market. In the fourth treatment, an external agent is introduced to the water market and competes for water with internal irrigators.

Under all four treatments, the common choice variables were crop choice, water source, and cropping area. Irrigation water per hectare was assumed to be fixed. The reason for this was that crop water demand is fairly inelastic, and anecdotal evidence suggests that irrigators generally reduce irrigation area rather than irrigation rate to maximise yield per hectare. Therefore, factor input per hectare and yield is assumed to be in fixed proportions, as are economic revenue and costs, in line with proportionality assumptions underlying linear programming (LP) models (Paris 1991). As a result, the shadow value to water use is reflected through the additional area that can be irrigated. Profit is therefore linearly increasing with water applied, and imposing a constraint on water use reduces the number of hectares under irrigated cotton. So the *pattern* of optimal location of irrigated industries across the basin landscape would be established, rather than an optimal *irrigation rate* for an irrigation enterprise.

Surface water allocation for each HRU is proportional to its size, so larger HRUs have a greater share of allocations available to its irrigation area, and smaller HRUs have a smaller share. Meanwhile, the deep drainage (DD) constraint is set on a basin-scale, such that the total drainage occurring is constrained at the basin level, and its spatial distribution is dependent on its associated water use across the basin. Under each setting, the impacts of DD and water caps are examined by parameterising the water and DD constraints. Water caps are successively reduced from the 2004 season's surface water allocation of 59,000ML, while drainage caps are reduced from the DD occurrence given the season's water allocations. The change in profit with gradual falls in these constraints is then used to form cost curves for reducing drainage or increasing environmental flows. The difference in the costs of meeting the environmental targets *between* each setting is a proxy for the benefit of adopting water efficient technologies (AIS) and water trading, while the difference in the costs under scenarios *within* each treatment represents the benefit of each policy instrument. A comparison is also made of the DD (which contributes to salt load), water use and profit obtained under each scenario. This is done in part to determine the efficacy of water and DD instruments to achieve environmental targets, pertaining to environmental flows and salinity reduction, at the least cost.

The results from these simulated scenarios shed some light on the value of markets in ameliorating the economic impact of environmental targets on the catchment. Where the water market is present in the third treatment, the cap on DD essentially resembles a (indirect) drainage cap-and-trade scheme. This is under the assumption of perfect information regarding the relationship between water applied and DD. The introduction of a separate DD instrument to contend with salinity is an attempt to remedy the market failure arising from the conjoined nature of water use and salinity. However, the usefulness of dual-instruments to manage these conjunctive resources is debatable since the administrative costs of dual-instruments will be quite high, meanwhile there is much uncertainty regarding its effectiveness in reducing environmental damage.

The presence of an external water buyer (a coal mine in this instance) is also simulated, and the effects on water use patterns in the catchment is analysed. For a given water market price, P_w, irrigators could choose to trade internally to other irrigators for crop production, or externally to the coal mine. It is assumed that only surface water could be traded. For simplicity, the gain from external water trade is calculated in terms of net benefit to the external agent from being able to use water, based on its derived demand for water estimated from industry data. The (exogenous) water price is parameterised in the interval from zero to $160/ML. Water sellers can make a profit from the volume which meets market demand at the given price, both internally and externally. As P_w increases or decreases, an irrigator would become a water supplier or demander depending on its derived demand for water, implicit in the LP. This allows for an analysis of the effect of water price changes on water demand and supply in the water market. Inset 7.1.presents a summary of the four treatments and the three policy scenarios:

Inset 7.1

Treatment One – Status Quo, no alternative irrigation systems (AIS) or water trade

Scenario 1.1	Base Case: no constraints except current environmental flow stipulations set out in the WSP.
Scenario 1.2	Water Cap (successively reducing water supply)
Scenario 1.3	Drainage Cap (successively reducing allowable drainage)

Treatment Two – Simulate use of alternative irrigation technologies (AIS), no water trade

Scenario 2.1	Base Case: no constraints except current environmental flow stipulations set out in the WSP, introduce AIS
Scenario 2.2	Water Cap
Scenario 2.3	Drainage Cap

Treatment Three – Simulate the introduction of water trading and AIS

Scenario 3.1	Base Case: no constraints except current environmental flow stipulations set out in the WSP, introduce water trade
Scenario 3.2	Water Cap
Scenario 3.3	Drainage Cap

Treatment Four – Simulate water trading, AIS and an External Water User

Scenario 4.1	Base Case: no constraints except current environmental flow stipulations set out in the WSP (identical to Scenario 3.1)
Scenario 4.2	Increase water prices successively, with trade among agricultural users only
Scenario 3.3	Increase water prices successively, with trade among agricultural and non-agricultural users.

7.3 DATA AND ASSUMPTIONS

7.3.1 Surface Water allocations

It is difficult to determine the actual history of use with respect to surface water for unregulated river systems. There is no legislated metering required for unregulated systems, and any private meters are regarded as confidential information that is not publicly disclosed. As a result, there is insufficient information to accurately determine river flow and implement extraction rules according to the flow class, or check on compliance. Withdrawal volumes could therefore only be estimated according to best guess (Hudson 2005, pers. comm..).

Therefore, although the number of licences issued and consequently the entitlements are observable for each Zone, it was not possible to identify the exact location of irrigators who hold water licences. Based on limited information, only an estimate of extractions could be made. The HRUs were divided according to the Zone in which they are located, and the Total Share Component (TSC), which refers to the surface water rights held, were summed together. This was used to estimate the proportion share of the basin water supply received in each Zone (Table 7-2).

The water availability for the zones was then compared to the estimates of river supply from SWAT. At each 'node' for Breeza, Ruvigne and Caroona there would be 'storage' for all HRUs to extract from, with the 'storage volume' determined through SWAT estimates of river flow at each node. On average, the estimated storage volume in one year was 34,518ML, with 2,762ML extracted at Caroona (8%), 10,679ML extracted at Breeza (31%), and 21,077ML extracted at Ruvigne (61%). The total storage water supply was rounded to 35,000ML for ease of calculation. While the total surface water available is quite similar between the SWAT estimates and the TSC recorded, the distribution between zones is somewhat different. According to CARE (2003), zones 1, 7 and 10 have no surface water entitlements, which suggest that irrigators in these zones are solely reliant on groundwater. However, SWAT simulations show that some surface water is actually available to these zones. It can be assumed that these irrigators will make use of the passing in-river flows to supplement their groundwater supplies.

Table 7-2: Surface water access in each SWAT node compared to SWAT estimates (adapted from Care 2003, p. 35),

SWAT Node	Zones	Sub-basins	TSC (ML)	SWAT estimate Area (ha)	SWAT estimate Surface Water (ML)
Ruvigne	3	2-5, 7, 32	25,158	12,113	21,000
	8	31, 11	4,989	5,502	
Caroona	6	22, 24-27	332	13,064	3,000
	10	23	-	200	
Breeza	1	15, 18, 19	-	8,217	11,000
	7	16	-	601	
		Total	33,241	39,699	35,000

Redistribution was therefore made to account for the access to surface water, such that access to river water is possible in all zones. This was done according to the irrigation water requirement for individual zones. The total surface water requirement for each HRU was estimated assuming that all HRUs produce irrigated cotton using surface water. The proportion share of Water Made Available is given by the ratio of surface water requirement in a particular Zone to the sum of water required by all zones. Each HRU has a share of the water available to its Zone, proportional to its size. This leads to a distribution which corresponds to the size of the irrigation area: Ruvigne receives 51% of the total Water Made Available; Caroona receives 35%; and Breeza receives 14% (Table 7-3). The methodology of distributing water based on land size and history of use is akin to the way government initially issued area-based water licences in unregulated systems. It was assumed that the proportional share of total availability in each Zone is unchanged, such that the water available in each Zone is in fixed proportions of the Water Made Available to each irrigation area.

Table 7-3: Surface water requirements in each Zone and share of water available, based on outcomes of SWAT simulations.

SWAT Node	Zone	Surface Water Required	% share of total availability
Ruvigne	3	41,643	20.67
	8	61,857	30.71
Caroona	6	65,556	32.55
	10	3,128	1.55
Breeza	1	28,190	14.00
	7	1,044	0.52
	TOT	201,416	100

7.3.2 Groundwater allocations

The purpose of the groundwater Water Sharing Plan (WSP) was to reduce water extractions to sustainable levels, equal to the recharge rate determined for each groundwater Zone in the Upper Namoi. This is envisaged to allow groundwater use to be maintained indefinitely into the future, at a consistent rate of use that has been stipulated for the next ten years. As mentioned earlier, the zones examined in this thesis are 1, 3, 6, 7, 8, and 10, which fall within the Mooki Basin. While it was not possible to verify the exact overlap of these zones and the sub-basins in SWAT, the GIS delineation of sub-basins in Figure 7.1 and the Zone referenced in Figure 6.7 were overlain to estimate the boundaries of each Zone. The groundwater entitlements and the supplementary water (SPW) for each of the relevant zones, as well as the irrigation areas in which the zones are located, are presented in Table 7-4. It is uncertain which of these HRUs belong to Groundwater Management Areas (GMA), for which the usage charge associated with groundwater use is different to non-GMA zones (see Section 6.6). To keep factor costs constant, it was assumed that the water usage charge (price per ML of water used) is the same as for the surface water. This then confines the cost differential between water sources to pumping costs, which form a more significant portion of costs associated with water use. It was also assumed that the proportion of groundwater allocation in each Zone is unchanged, such that the groundwater available to a Zone is in constant proportion of the total availability ("% Share Total Allocation").

In this study groundwater allocations were not reduced beyond the specified reductions in the groundwater WSP, so that total extraction remained at its estimated sustainable groundwater use across the planning period. Based on the groundwater entitlements, Ruvigne has the greatest share of groundwater resources, followed by Caroona then Breeza.

Table 7-4: Groundwater (Gw) allocations according to irrigation areas within Mooki.

	Zone	Gw Entitlements	Gw SPW		Gw Total		% Share Total Allocation
			Yr 1-5	Yr 6-10	Yr 1-5	Yr 6-10	
Ruvigne	3	16,338	7,645	4,247	23,983	20,585	43
	8	13,794	5,693	3,163	19,487	16,957	35
Caroona	6	6,915	-	-	6,915	6,915	12
	10	949	-	-	949	949	2
Breeza	1	992	1,071	669	2,063	1,661	4
	7	2,810	205	205	3,015	3,015	5
				Total	56,412	50,082	100

There are also carry-over rules that apply to groundwater allocations. In any year, a maximum of two-season's allocations can be used, while total extractions across three-rolling years must not exceed allocations received during this time. The limits on groundwater extraction over the planning period are shown in Table 7-5. The drop in allocations after year 5 is due to the phasing out of supplementary groundwater extraction licences.

Table 7-5: Annual groundwater (Gw) allocations and extraction limits.

Yr	Gw Total Annual Allocation (ML)	Two-year Gw Total Extraction Limit (ML)	Three-year Total Gw Extraction Limit (ML)
1-4	56,412	112,824	162,906
5	56,412	106,494	156,576
6-10	50,082	100,164	150,246

With reference to the dynamic programming model, the groundwater available for extraction each period, $\overline{Ga_{it}}$, is subject to the two-year extraction and three-year extraction limits shown above.

7.3.3 Parameter Assumptions

7.3.3.1 Crop yields

Assumptions for irrigated cotton yields were based on the 10-year trend reported by ABARE (2005), NSW DPI (2006a), Boyce (2005) and Tennakoon and Milroy (2003). The average irrigated cotton yield (bales[7]/ha) has increased from 5bales/ha in the 1960s, when irrigated cotton first began in Australia, to 8bales/ha in 2001 (ABARE 2006). From NSW DPI (2006a) and Boyce (2005), it has become common to achieve yields of 8-10bales/ha. Anecdotal evidence also suggests that yields between 8-10bales/ha is fairly common. However, Tennakoon and Milroy (2003) have reported yields of 6 bales/ha in the Namoi Valley. The yield estimate from SWAT was therefore standardised to a yield range of [6, 10], assuming a normal distribution with mean of 7.4 bales/ha and standard deviation (S.D.) of 1.3. Dryland cotton yield assumptions were based on NSW DPI (2006a) and Marshall et al. (2002). The yield range for dryland cotton was assumed to be [0.9, 4], with a mean of 2.6 bales/ha and S.D. of 0.9. Cotton seed was also sold as a by-product to cotton lint production. Based on NSW DPI (2006a), the ratio of seed to lint was assumed to be 1.59; i.e. for every kilogram (kg) of cotton lint, 1.59kg of seed is produced.

Other dryland crops grown in the Namoi region include wheat and sorghum. These dryland crop yields were similarly based on NSW DPI (2006a) and ABARE (2005). A distribution with mean 3.6 t/ha and S.D. of 1.4 was assumed for wheat, with yield range of [1.5, 6]. A mean yield of 4 t/ha was assumed for dryland sorghum, with a yield range of [3, 5] and S.D. of 0.6.

[7] One bale is 227kg

7.3.3.2 Crop Prices and Input Costs

Price and cost data were based on ABS (2005) and NSW DPI (2006a) for the Northern Zone. The 10 year-average price received for cotton was $2/kg, with production costs, excluding water, of around $2,333/ha. The price of cotton seed was assumed to be $175/t. For dryland cotton, prices received for lint and seed were the same, except that the production costs were much lower at $965/ha. Of course, the yield of dryland cotton was also much lower compared to irrigated cotton. The price of dryland wheat was around $150/t while the cost of production was around $309/ha. The average price for grain sorghum was $140/t while production costs were $375/ha (NSW DPI 2006a; ABS 2005). For the purposes of this study, all prices were assumed in real terms to exclude the effect of inflation. The crop yield, price and cost assumed in this thesis are shown in Table 7-6.

Table 7-6: Crop yield, price and cost assumptions.

Crop	Mean yield	S.D.	Crop Price ($)	Cost per hectare ($)
Irrigated Cotton	7.4 bales/ha	1.49	2/kg	2,333
Dryland Cotton	2.6 bales/ha	0.9	(454/bale)	965
Dryland Wheat	3.6 t/ha	1.4	150/t	309
Dryland Sorghum	4 t/ha	0.6	140/t	375

7.3.3.3 Water Charge

The water prices assumed were according to the Independent Pricing and Regulatory Tribunal determinations. Only the usage charge of $1.53/ML has been included in variable costs, since the entitlement charge is regarded as a sunk cost for holding the water right. Nevertheless, the charge for water itself makes up only a fraction of the costs associated with using water. Most of the costs of using surface or groundwater were due to expenditure on fuel used for water pumping (see Table 7-9), which forms the most significant factor in water use decisions. Again, all prices were assumed to be in real terms.

7.3.4 Irrigation Scheduling

In SWAT, there are a set of 'management files' through which HRUs are given commands, such as crop planting, harvest, irrigation, fertilisation etc. In this study, the only irrigation crop considered is cotton, for which irrigation scheduling is regarded as an important determinant of yield. Too much or too little water at crucial times in a season could cause significant yield losses (Milroy et al. 2002). The sensitivity of cotton crops to irrigation is in-built into SWAT, and much attention was given to the formulation of a set of appropriate irrigation scheduling. The decision criterion in SWAT of whether to irrigate or not was to satisfy crop water demand, given the in-stream flow at the scheduled irrigation time and the irrigation rate specified by the user.

There were two ways users can define irrigation events: by heat-unit irrigation and date-irrigation. Irrigation according to heat-units is more realistic, since irrigation can be timed according to the stages in crop growth. However, this method has high computational requirements and through trial simulations did not produce good yield response. Date-irrigation sets specific dates for irrigation to occur, although scheduled irrigation events are not necessarily in sync with crop requirements. Both heat-unit and date-irrigation scheduling have disadvantages in which irrigation timing was not perfectly aligned with water requirements; but in this instance date scheduling was more accurate than heat-unit irrigation and was chosen for this study. Full details regarding the setup of irrigation scheduling is given in Appendix C.

7.3.5 External Buyer – Derived Demand for Water

Production data taken from a large Australian-based mining firm production and sustainability report was used to determine the water demand function in coal mining Due to the relatively more constrained access to water resources for the mine, coal production was taken as a function of water available. A quadratic relationship was assumed, and a regression was run using the available data set, consisting of six years of observations.

Table 7-7: Coal production and water use (source: BHP 2006; BHP 2007).

Yr	Tot production ('000 t)	Fresh Water Intensity (litres/t)	Inferred Water demand (ML)
2001	4,877	290	1,414
2002	4,997	290	1,449
2003	7,783	240	1,868
2004	9,692	290	2,811
2005	9,695	220	2,133
2006	10,089	230	2,320

Assuming no coal can be produced without water, such that the intercept is zero, the estimated production function was $Coal = -0.1721W^2 + 4318.2W$, with R^2=0.81 and s.e. of 1,169,471. The factor demand function was then obtained by invoking Hotelling's Lemma:

$$\pi = P_y\left(-0.172W^2 + 4318.2W\right) - P_w W$$

$$\frac{\partial \pi}{\partial W} = P_y\left(-0.344W + 4318.2\right) - P_w = 0$$

$$W = \left(\frac{P_w}{P_y} - 4318.2\right) \Big/ (-0.344)$$

$$= \frac{4318.2}{0.344} - \frac{1}{0.344} \cdot \frac{P_{wy}}{P}$$

Where P_y is the output price for coal, P_w is the market price of water, and W is the quantity of water used in coal production. Assuming $P_y = \$45$ per ton (ABARE 2006), the coal mine's derived demand for water is $W^* = 12{,}522 - P_w/15.48$.

7.3.6 Water Efficient Irrigation Systems

Once an investment is made in an alternative system, the irrigator is locked into the system due to the need to recoup initial investment. The repayment period is assumed to be ten years, with annualised fixed costs incurred by the irrigator for the remainder of that period once the investment is made. The average capital cost and annualised repayments (at 5% interest over ten years) are shown in Table 7-8.

Table 7-8: Capital investment and annualised fixed costs (adapted from Foley and Raine 2001).

Annualised Fixed Costs	Capital Cost/Ha $ (ave)	Annualised Fixed Cost/Ha ($)
Pivot or Linear Move low pressure (river)	2,000	$246.68
Pivot or Linear Move low pressure (bore)	2,400	$296.01
Drip/Jet spray	4,500	$555.02

The operational cost of water efficient irrigation systems is shown in Table 7-9. The cost of operation is based on the fuel consumption required to deliver water to the field, which .increase with pressure requirements ("pumping head") for the system. It is assumed that all irrigators use diesel-fuelled systems at a cost of 75c/litre, which is considered to be the higher end of fuel costs (Smith 2005, pers. comm.).

Table 7-9: Operational costs of different irrigation systems (source: Smith and Richards 2003)

Irrigation System	Labour requirement	Assumed Pumping Head (metres)					Pumping Costs $ per Megalitre			
		Static Lift	Pipe Friction	Hose Loss	Operating Pressure	Total Head	Diesel Cost per litre @ cents = $			
							40	45	55	**75**
Surface Furrow (River)	high	10				10	5.06	5.69	6.95	9.48
Surface Furrow (Bore)	high	30	5			35	17.70	19.91	24.33	33.18
Pivot or Linear Move low pressure (river)	low	10	10		10	30	15.17	17.06	20.86	28.44
Pivot or Linear Move low pressure (bore)	low	30	10		10	50	25.28	28.44	34.76	47.40
Drip/Jet spray	low	25	15		10	50	25.28	28.44	34.76	47.40

7.4 OTHER CONSIDERATIONS

7.4.1 Deep Drainage and Return Flows in the Mooki

As mentioned in earlier chapters, increasing water use efficiency would have two opposing effects; the reduction in deep drainage (to the natural level of recharge) would lead to reduced salinity risk, but on the other hand it would also have an adverse effect in reducing return flows (Heaney and Beare 2001). If water quality has negligible effects on crop growth then the upstream user would only reduce deep drainage to the extent which does not impede with downstream supply.

The contribution return flows make to in-stream or groundwater supply can be estimated through SWAT, which simulates groundwater recharge and return flows through water balance equations. However, the 'optimal' level of return flows for maximising basin profits would require that the deep drainage and return flow relationship for every HRU, under every production activity is considered in the optimisation problem. This would command enormous computational expense, because each time one upstream HRU varies its landuse activity, there are downstream repercussions in terms of water supply changes. The number of possible outcomes is therefore a permutation of the number of choice variables and HRUs (see Appendix D). Moreover, a new simulation needs to be conducted each time a constraint changes. From a programming perspective, one optimisation would require an amount of time and technical resources beyond the scope of this thesis. It is therefore not feasible to include the entire range of possible outcomes in the presence of return flows. An alternative would be to aggregate the HRUs to reduce the number of possible outcomes, which at best could be lumped according to the irrigation areas (Ruvigne, Caroona, and Breeza). However, this takes away the advantage of using spatially explicit information.

The effect return flows have on river salinity depends, amongst other factors, on the salinity of groundwater where return flows originate (Heaney and Beare 2001). If the recharge is high and ground water salinity is high, the salt concentration may decrease if the amount of saline recharge transported to the river system is reduced. Reducing deep drainage therefore lowers the negative externalities imposed on downstream users. In contrast, if return flows originate from irrigation areas with relatively low underlying groundwater salt concentration, it can provide dilution flows downstream. Return flows would then provide positive contributions to downstream water supply and diluting effects, so trade or improved irrigation efficiency reduces these beneficial return flows. For the Mooki, there are no significant salinity problems within the catchment, but the salt load and EC reading for the Mooki and Namoi Rivers are relatively high compared to other cotton producing catchments (see Section 6.4). The main concern is the salt load contribution to the downstream Barwon-Darling River, which carries the salt into the Murray-Darling system. Given the relatively high salt concentration in the Namoi and Mooki Rivers, and its high river-aquifer connectivity, it is considered optimal to minimise the level of saline return flows since it generate greater negative than positive externalities.

7.5 SUMMARY OF CHAPTER 7

In this chapter, the data and the methodology used in the modelling process were presented. The interdisciplinary approach, involving the use of the biophysical model, SWAT, was first described. Attention was given to the way SWAT delineates the basin, which allows for a spatially explicit examination of the water resource use. This was followed by a detailed discussion of the data and assumptions made in the model, which were according to the characteristics of the case study basin, the Mooki, as introduced in Chapter 6. In this next chapter, the results of the empirical study are analysed, to shed light on the effectiveness of various catchment policies aimed at correcting these distributional problems.

Chapter 8.

RESULTS OF THE EMPIRICAL STUDY

In this chapter, the results of the empirical research based on the case study of the Mooki catchment are presented. The results are discussed in the following way. In Section 8.1, a summary statistics of the different simulation treatments is provided. This enables a general overview of the outcomes under the assumptions made for each treatment. This is followed by a review of the inter-temporal resource use, focusing on the role of the carry-over rules for groundwater allocations, in Section 8.2. An analysis of the change in production activities, with respect to changes in water allocation and deep drainage, is then presented in Section 8.3. Changes in basin profit and water use for various environmental flow policies are then examined in Section 8.4. In Section 8.5, the effect of a dual-instrument on basin profit is assessed, through simultaneously imposing deep drainage caps and surface water caps. This is followed by a comparison of the effectiveness of using each instrument separately in achieving environmental objectives and minimising the impact on basin profit, in Section 8.6. For each of the above sections, the focus in on the three treatments: Treatment One (Status Quo: furrow irrigation only); Treatment Two (with alternative irrigation systems – AIS – pivot and drip irrigation); and Treatment Three (with water trading and AIS). In Section 8.7, the results under Treatment Four (external water trade) are presented. This is to examine the economic implications of increased competition for water from an agent outside of the region. A comparison is then made to the outcome where only internal water trading is allowed. Where water trading is considered, under Treatment Three, the market price for water is assumed to be zero, such that water trading is costless. Any ‘trade’ of water is simply a reallocation to its highest value use, and represents the maximum value of establishing a water market in the case study basin. The impact of water market price changes is only considered in Treatment Four.

8.1 SUMMARY STATISTICS

8.1.1 Summary of Activities

As mentioned in Chapter 7, there are nine production activities that are considered. These are reproduced here for ease of reference (Table 8-1).

Table 8-1: Description of activities.

Activity number	Source of water	Crop production	Irrigation
1	Surface	Cotton	Furrow
2	Ground	Cotton	Furrow
3	Surface	Cotton	Pivot
4	Ground	Cotton	Pivot
5	Surface	Cotton	Drip
6	Ground	Cotton	Drip
7	Dryland	Grain/Sorghum	-
8	Dryland	Wheat	-
9	Dryland	Cotton	-

The average SWAT output parameters and profitability of each activity is shown in Table 8-2. Between the nine possible activities, pivot irrigation using groundwater (activity 4) is the most profitable activity. However, this is very similar to the profit under activity 2, which also source groundwater but using the less water efficient furrow systems. Likewise, pivot irrigation using surface water (activity 3) is also comparable to furrow irrigation using surface water (activity 1). This suggests that, the yield increase under pivot systems just offset the increased capital outlay, making it marginally more profitable than traditional furrow systems. However, pivot systems have additional benefits in terms of reduced water use and deep drainage (DD), allowing for greater water use efficiency. Drip irrigation (activities 5 and 6) has the lowest average profit/ha of the irrigation systems, although it has the greatest water savings and DD reduction compared to other irrigation systems. This is because of the significantly greater capital investment required, which is more than double the cost of pivot systems. Furthermore, crop yield does not increase appreciably. Drip irrigation is therefore not very profitable because the yield improvement is inadequate to cover the significant capital outlay. These outcomes are in line with anecdotal evidence; many irrigators report water savings and yield increase under pivot systems, but not significant yield increase under drip irrigation. This makes drip an unfavourable option because there is no yield increase to recoup the capital investment, despite significant water savings. In this regard, pivot systems have the advantage of allowing irrigators to maintain profitability in situations where water supply is scarce.

Table 8-2: Summary of activities.

Activities	Ave. Profit ($/ha)	Ave. Irrigation (ML/ha)	Ave. DD (ML)/ha	Ave. Yield[8] /ha
1	1,074.18	6.30	1.25	6.89
2	1,199.80	8.44	2.03	7.41
3	1,097.29	5.63	1.18	7.47
4	1,209.03	7.65	1.52	7.95
5	585.59	5.37	1.14	6.99
6	648.90	6.09	1.33	7.26
7	308.61	-	-	3.93
8	136.40	-	-	3.77
9	475.02	-	-	2.49

A general trend that is observed from the above table is that, for any irrigation system, the profit where groundwater is used (activities 2,4 and 6) is greater than where surface water is used (activities 1,3 and 5). This is despite the cost of groundwater being higher than surface water. The difference in profit between ground and surface water is on account of the reliability of the water source, which affects crop yields. In the SWAT model, on-farm storages cannot be modelled for each HRU so surface water cannot be pumped and stored on-farm. Therefore water sourced from the river is applied directly to the crops when it is available. The irregularity of river flow in an unregulated system means that irrigation occurs less frequently, and can be insufficient to meet crop water requirement, leading to lower yields. On the other hand, the relative reliability of groundwater means it is more readily available and full irrigation occurs at regular intervals. As a result, irrigation water, yield and DD are lower when sourcing surface water and higher when groundwater is used. Therefore, even though groundwater is more costly to use, the

[8] Cotton in bales per ha (activities 1-6, and 9), wheat and sorghum in tons per hectare (activities 7 and 8).

security of supply allows greater crop yields which offsets the pumping cost. In a sense, the extra cost of pumping groundwater can be regarded as a 'premium' paid for increased security of supply, which is compensated by increased yields. It is therefore the water source that is relied upon by irrigators in the Mooki basin, which is in line with reality. This result also reflects the significance of secure water supplies, which enables the capital cost of water efficient technologies to be recouped and provides an incentive for irrigators to make the investment.

Considering dryland crop choices, dryland cotton (activity 9) is the most profitable dryland production. While this is less than half the profitability of irrigated cotton, compared to wheat and sorghum, it is the preferred dryland crop as water becomes scarce. This will be observed in the following sections.

8.1.2 Comparison of Base Case Scenarios

In this section, a comparison is made between the Base Case scenarios under each treatment. The following table shows the resource use and annual profit given the 2005 surface water allocation of 59,000ML, and 56,412ML groundwater allocation (Table 8-3). Seasonal water allocations are assumed to be fixed throughout the planning period of 10 years.

Table 8-3: Base Case scenarios water use and annual profit

Treatment	Base Case Scenarios				
	Surface Water Use (ML)	Groundwater Use (ML)	Total Water Use (ML)	DD (ML)	Profit/yr ($Mill)
One	53,628	56,059	109,687	25,419	35.32
Two	58,698	56,200	114,898	24,200	36.85
Three	59,000	56,241	115,241	24,710	40.15

The increase in annual profit moving from Treatment One (status quo) to Treatment Two (no water trade, with alternative irrigation systems – AIS) is $1.5 Mill, which the increase in surface water use is 5,070ML. Total water use increases by 5,211ML. That is, when AIS are available profit increases by 4% while surface water use increases by 9%, or

4.7% in total water use. This is while DD falls by 1,219ML, or 5%. This result suggests that, water use efficiency increases where AIS are used, although irrigators take advantage of the increased water efficiency to expand production, leading to an increase in water use and profit.

A comparison between Treatment Two and Treatment Three (with trade and AIS) shows that profit increases by $3.3Mill, and surface water use by 302ML. Relative to Treatment Two, this equates to a 9% increase in profit for a 1% increase in surface water use. In terms of total water use, an increase of 0.3% is observed. This result suggests that a relatively large increase in profit is achieved for a small increase in water use where there is water trade, although DD increases by 2% relative to Treatment Two. However, DD is still lower relative to Treatment One; drainage falls by 3% despite an increase in total water use of 5%, while profit increase by 10% compared to the status quo.

Considering the relative change in profit to water use under each treatment, it appears that AIS and water trading can improve catchment profit and water use efficiency. This is despite an increase in total water used. Some producers that initially could not grow irrigated cotton profitably under furrow systems (Treatment One) find it possible to profitably produce irrigated cotton where water efficient technologies are available (Treatment Two), which leads to an increase in water use and fewer losses to drainage. Furthermore, where water trading is possible, water demand in the catchment is further increased as previously inactive allocations are mobilised (Treatment Three). While water trading leads to greater water use, it also allows for the highest profit due to greater allocative efficiency, and reduced unproductive water lost to DD. There is a natural incentive for water to be used efficiently by creating a market value for water, since there is an opportunity cost associated with water lost to drainage. The objective of reducing DD can therefore be achieved through the creation of a water market or promoting the use of AIS, without the need for additional administrative control over water use.

8.2 INTER-TEMPORAL RESOURCE USE – STOCHASTIC SURFACE WATER

Due to the carry-over rules for groundwater allocations, dynamic resource use decisions need to be made with respect to groundwater use. Each year, extractions must not exceed the two-year water allocation, while total extractions within three-rolling years must not exceed the three-year allocation. While the same rules apply for surface water, the ephemeral nature of the river flow in the Mooki (an unregulated system) means that it is optimal to extract river water whenever the opportunity arises, such that surface allocations are exhausted within the season (Hamparsum 2006, pers. comm.). Due to the phasing-out of supplementary groundwater entitlements scheduled after year 5 of the groundwater Water Sharing Plan, there are two sets of extraction constraints over the 10-year planning period. The extraction limits were shown in Chapter 7 but are replicated here for ease of reference (Table 8-4).

Table 8-4: Groundwater (Gw) allocations and extraction limits.

Yr	Gw Annual Allocation (ML)	Two-year Gw Extraction Limit (ML)	Three-year Gw Extraction Limit (ML)
1-4	56,412	112,824	162,906
5	56,412	106,494	156,576
6-10	50,082	100,164	150,246

Under the Base Case scenario of each of the treatments, the rate of groundwater extractions is much lower than the two-year and three-year extraction limits, as shown in the previous section (see Table 8-3). Rather than exhausting the entire two-year groundwater allocation within one season, optimal extraction appears to be at a point close to its annual allocations. The reason for this is because the temporal value of groundwater resources is integrated into the resource use decisions, such that the rate of extraction is reduced to where the discounted value of the groundwater resource is equated between periods. Where there is no stochasticity in surface water supply or other parameters, surface and groundwater use (and hence deep drainage occurrence) is fairly consistent because optimal production in one season is not different from year to year. This is a consistent trend throughout all three treatments, and after year 5 resource use drops slightly due to cuts in entitlements, but also maintains a regular production pattern. Under conditions of stochastic surface water supply, *ceteris paribus*, the optimal groundwater extraction rate is expected to differ from year to year depending on the river flow available.

Stochastic surface water scenarios were analysed for all treatments, to examine how irrigators may choose to 'bank' or 'borrow' groundwater allocations. The surface water supply was set annually according to each of the 8 years simulated in SWAT, from 1996 (year 1) to 2003 (year 8). Water allocation for 2004 (year 9) was assumed to be the same as allocations for 2005 (year 10) of 59,000ML (Table 8-5).

Table 8-5: Stochastic and deterministic water supply.

Year	Surface Water Allocation (ML/yr)		Groundwater Allocation (ML/yr)
	Stochastic	Deterministic	Deterministic
1	27,579	59,000	56,412
2	31,265		
3	97,346		
4	33,727		
5	64,840		
6	64,623		50,082
7	14,931		
8	26,394		
9	59,000		
10			

8.2.1 Treatment One – Status Quo

Under Treatment One, where surface water is stochastic, the rate of groundwater extraction moves in the opposite direction of surface water supply (Figure 8.1). This is expected because, where surface water is erratic, some banking or borrowing of groundwater allocations is needed to compensate for the irregularity of the alternate water source. Where surface water is deterministic, however, it is not necessary to hedge groundwater allocation so the rate of groundwater extractions is constant and close to the annual allocation. It can be inferred that, where surface water is stochastic, some banking and borrowing of groundwater allocations occur, but where surface water is deterministic and constant, annual groundwater allocations are used up every year. That is, there is no banking or borrowing. This suggests that the carry-over rule for groundwater is especially important for irrigators in an unregulated system, to reduce the economic impact of the scarcity and irregularity of surface water.

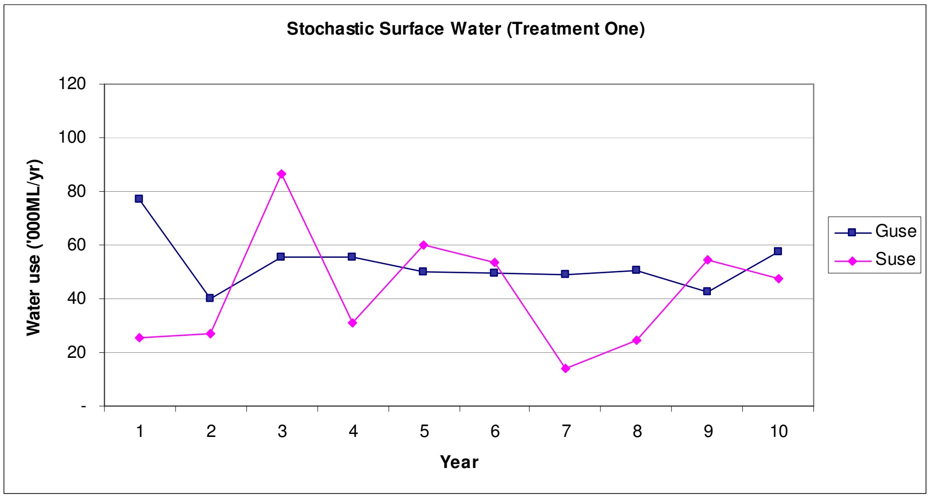

Figure 8.1: Groundwater use (Guse) with stochastic surface water (Suse), Treatment One.

8.2.2 Treatment Two – No Trade, with Alternative Irrigation Systems (AIS)

Under Treatment Two, the observed trend in groundwater use is similar to Treatment One, although there is less carry-over of groundwater allocations. Groundwater extraction generally moves in opposing direction to surface water supply, at a rate that compensates for the lack of (or surplus in) surface water (Figure 8.2). Where surface water allocations are constant from year to year, groundwater use is constant and no borrowing or banking occurs. On the other hand, where surface water is stochastic, some groundwater banking or borrowing would occur although under Treatment Two only a small amount is carried over.

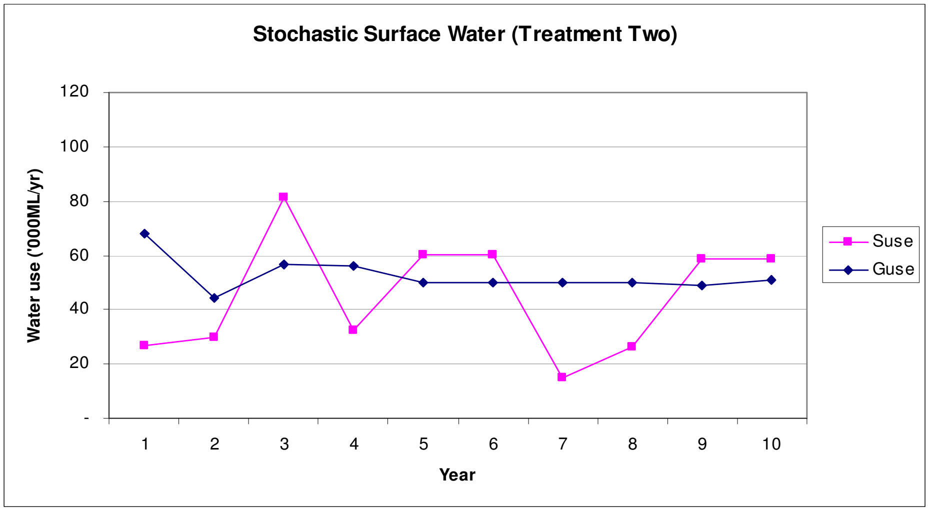

Figure 8.2: Groundwater use (Guse) with stochastic surface water (Suse), Treatment Two.

8.2.3 Treatment Three – With Trade and Alternative Irrigation Systems (AIS)

Under Treatment Three, the water use pattern is also similar to Treatments One and Two. There is a negative correlation between the rates of groundwater extraction and surface water supply, where surface water is stochastic (Figure 8.3). Where surface water supply is certain and constant, groundwater use is likewise constant and the full annual allocation is used without any banking. This again highlights the importance of the carry-over rule for the Mooki, since, under all three treatments, the banking and borrowing of groundwater appears to be a significant adjustment mechanism to the erratic nature of surface water supply.

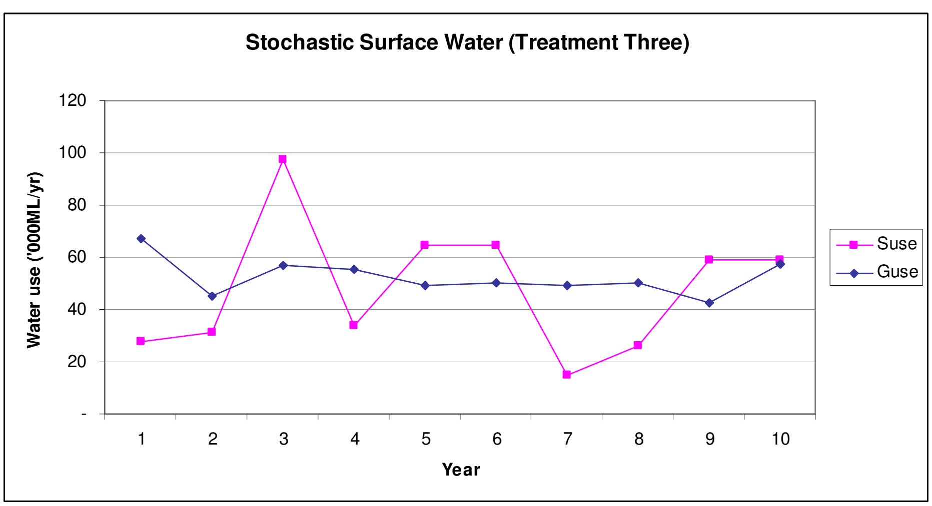

Figure 8.3: Groundwater use (Guse) with stochastic surface water (Suse), Treatment Three.

8.2.4 Summary of Inter-temporal Resource Use

Where surface water is certain and unchanged from year to year, the groundwater carry-over rule does not appear to have much currency since the optimal extraction rate is essentially the annual allocation. This is because the profit-maximising production pattern is unchanged from year to year. However, where surface water is stochastic, the carry-over rule is essential to ameliorate the economic impact of the erratic nature of river supplies. However, the rate of banking or borrowing does not appear to vary significantly between treatments. As can be seen in Figure 8.4, groundwater borrowing occurs when surface water (Sw) is low, and banking occurs when surface water supply is high. The groundwater-borrowing behaviour corresponding to Treatment One (T.One), Treatment Two (T.Two) and Treatment Three (T.Three) are positive when borrowing (above the x-axis), and negative when banking (below the x-axis). Where there is no carry-over or borrowing of groundwater allocations, the lines sit on the x-axis, which occur for much of the planning period. This is because the rate of extraction is subject to the two-year and three-year extraction limits, such that groundwater use remains relatively close to its annual allocations each year. Given that there is effectively an 'expiry date' for groundwater allocations, it is optimal to extract groundwater at a rate corresponding to the increase in groundwater stock. It seems that the carry-over rules are the dominating factor in banking decisions.

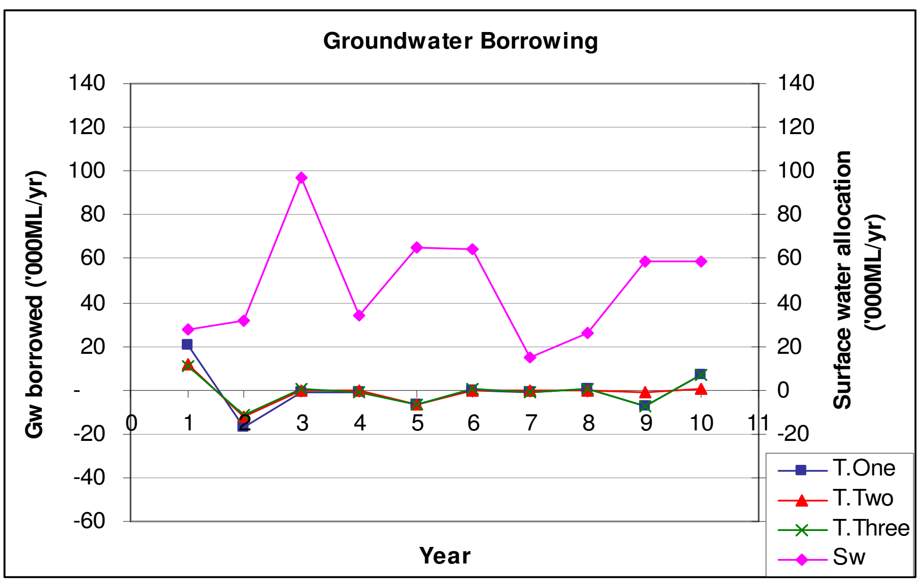

Figure 8.4: Comparing groundwater borrowing under the three treatments.

For the sections to follow, the analyses are based on the expected value of annual water supply, such that surface water is deterministic and constant over the planning period. This is to separate the impact of uncertain water supply from the impact of environmental policy on irrigators in the Mooki, such that the focus remains on the expected outcome of policy changes and how irrigators can adjust at least cost. It is assumed that for each season, the expected volume of surface water available is as estimated by the Department of Natural Resources (2007) of 59,000ML for the Mooki. Annual groundwater allocations are also assumed to remain at the expected volume of recharge according to the groundwater Water Sharing Plan (CARE 2003).

8.3 PRODUCTION ACTIVITY CHANGES

In this section, production activity changes under water and deep drainage (DD) constraints are discussed. Results are presented in three sub-sections, according to the treatment being considered. Each sub-section begins with a discussion of production activities under the Base Case scenario of the treatment (with full season's water allocations and no constraints on DD). This is followed by a discussion on the changes in production activity when water caps are imposed, and when DD caps are imposed. With surface water caps, the outcome under water caps of 55,000ML, 40,000ML and 20,000ML are compared. This is to simulate the impact of future reductions in surface water supply, potentially from the government buy-back of entitlements for environmental flows. With DD caps, the outcome under DD caps of 20,000ML, 14,000ML and 10,000ML are also compared. The purpose is to simulate the impact of basin-wide DD caps that may be required to meet end-of-valley salinity targets. Under each scenario, the results are further disaggregated into irrigation areas (Ruvigne, Breeza and Caroona), which allows for a clearer understanding of the trends observed. Production activity changes are examined based on one season, which is representative of all other seasons due to the relatively consistent production across the planning period.

8.3.1 *Treatment One – Status Quo*

8.3.1.1 Scenario 1.1 – Base Case

For now, only activities 1 and 2 (furrow irrigation sourcing surface and groundwater, respectively) are considered, since the assumption under Treatment One (Status Quo) is that only traditional furrow irrigation systems are used. Water trading is also not considered in this treatment. Dryland crops that can alternatively be produced include wheat (activity 7), sorghum (activity 8) and dryland cotton (activity 9). These dryland cropping options are constant for all treatments.

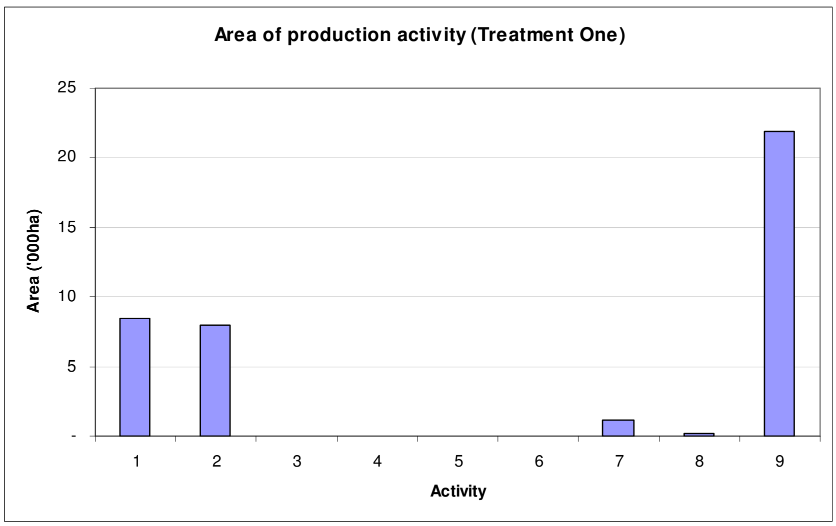

Figure 8.5: Scenario 1.1 production activities.

Under Treatment One, the area of irrigated cotton HRUs is almost at par with the number of HRUs suited to dryland crops (Figure 8.5). Dryland cotton and wheat makes up around half the area of the basin, and the other half of the catchment produce furrow irrigated cotton equally sourcing either surface (activity 1) or groundwater (activity 2). This could suggest that, where there are no alternative irrigation systems (AIS) or water trade, much of the HRUs which are currently under irrigated production may be more profitable under dryland production. This may reflect an overdevelopment of irrigation enterprises in the Mooki currently, and that some areas could be retired from irrigation at relatively low economic cost.

Disaggregating production activities by irrigation areas shows that at the optimum irrigated cotton has the highest prevalence in Ruvigne, followed by Caroona and Breeza (Figure 8.6). This suggests that, under the status quo optimum solution, irrigation should occur mostly in the downstream region of Ruvigne, which has a heavy reliance on groundwater (activity 2). With regards to surface water, however, Caroona appears to be the main user considering it has the greatest area under activity 1 (furrow irrigation sourcing surface water). Breeza seems to be the least suited to irrigation, since it has the smallest proportion of its area under irrigated cotton. The relatively large areas under dryland cotton in all irrigation areas implies that the distribution of water according to area – the way area-based licences were issued to irrigators in unregulated systems – was not efficient. There is scope to reallocate water and improve allocative efficiency and basin profit.

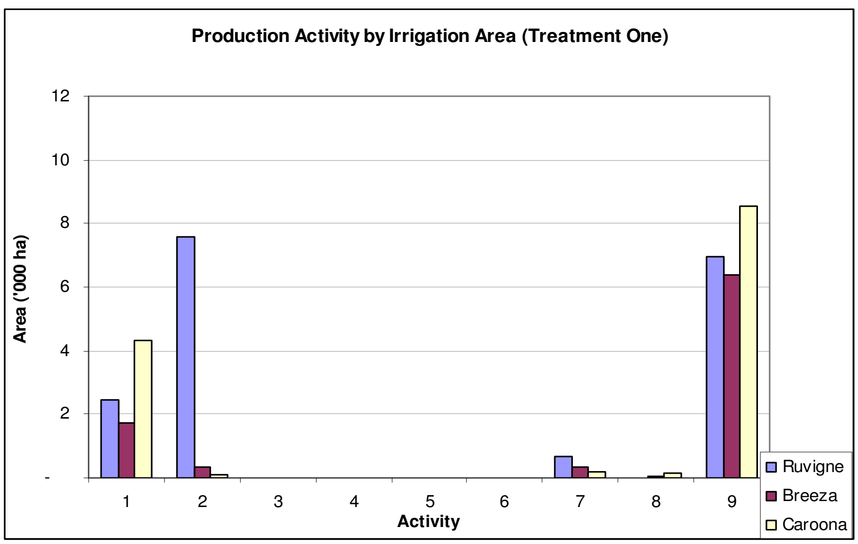

Figure 8.6: Production activity by irrigation area, Treatment One Base Case.

8.3.1.2 Scenario 1.2 – Water Caps

Under scenarios with water caps, the outcomes where surface water supply is capped at 55,000ML, 40,000ML and 20,000ML are presented. This is compared to the base case where the full season's surface water supply, of 59,000ML, is available.

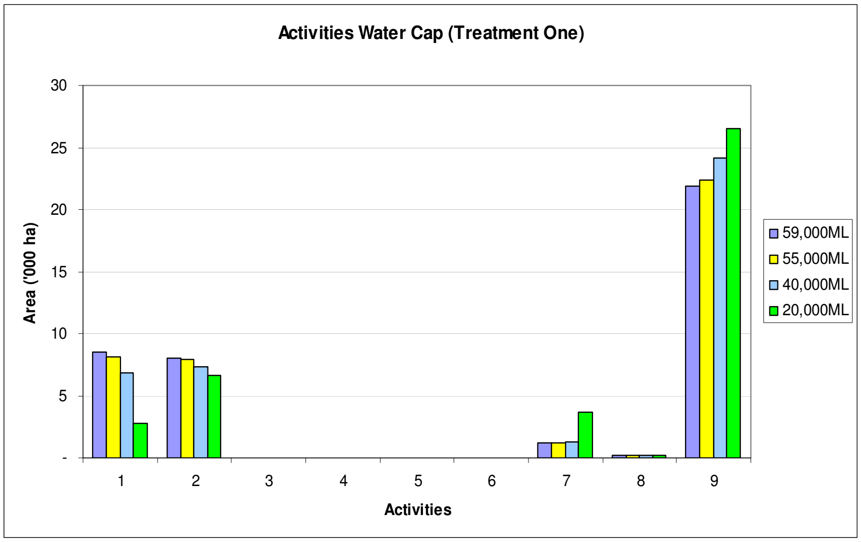

Figure 8.7: Scenario 1.2 production activities.

The impact of water caps on production activities, at the aggregate scale, is shown in Figure 8.7. As expected, where water caps are imposed, the area under surface irrigated cotton (activity 1) is reduced successively with a corresponding increase in the area under dryland production. Since groundwater is unaffected by the water caps, the total groundwater used remains unchanged for all constraints.

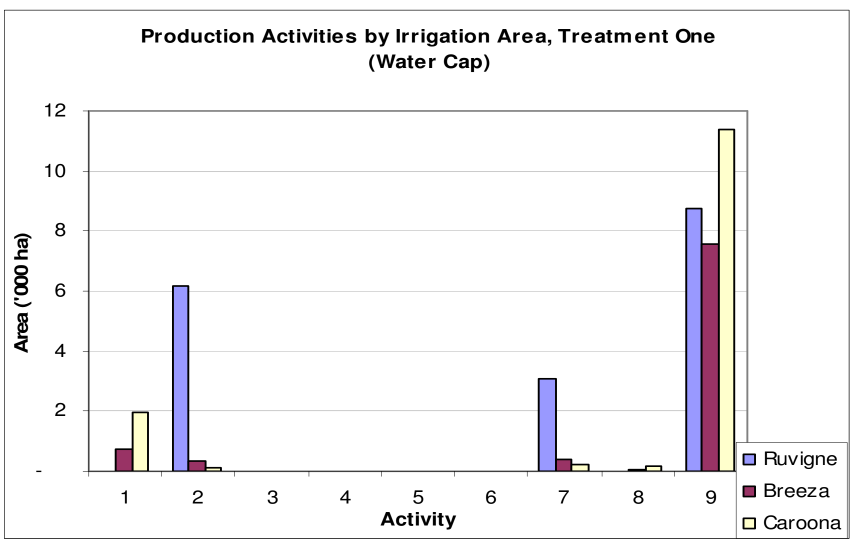

Figure 8.8: Production activities by irrigation area, Treatment One under Water Cap.

The change in production patterns, by irrigation area, is considered under a water cap of 20,000ML (Figure 8.8). Irrigated cotton under activity 1 (surface furrow irrigation) is reduced significantly in all irrigation areas, particularly in Ruvigne where all surface irrigated cotton is eliminated. However, Ruvigne still has the largest irrigated cotton production, under activity 2 (groundwater furrow irrigation). This suggests that cotton production in Ruvigne is relatively less affected by surface water caps since it is reliant on groundwater sources. On the other hand, Breeza and Caroona, which rely on surface water as its primary water source, experience comparatively greater impact from reductions in surface water allocations. It can be expected that, if environmental flow requirements are increased, the upstream irrigation areas would bear greater economic losses than Ruvigne. It will be seen later that the overall opportunity cost of meeting environmental flow targets is also greater than necessary, since surface water resources have not been distributed efficiently.

8.3.1.3 Scenario 1.3 – Deep Drainage Caps

Under scenarios with DD caps, the outcomes where drainage is capped at 20,000ML, 14,000ML and 10,000ML are presented. This is compared to the base case where DD is unconstrained, at 25,000ML.

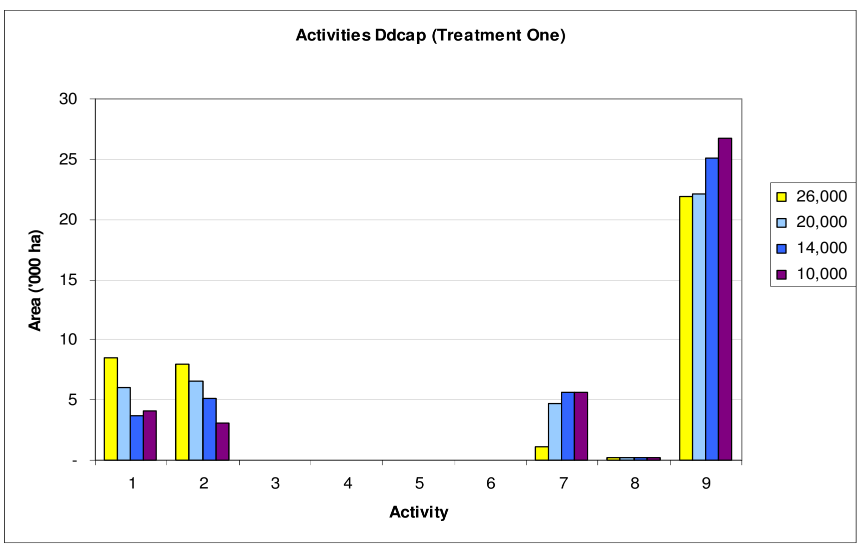

Figure 8.9: Scenario 1.3 production activities.

Where DD caps are imposed, the area under irrigated cotton sourcing both surface (activity 1) and groundwater (activity 2) are reduced as the DD target becomes stringent (Figure 8.9). There is a distinct drop in irrigation areas under activity 2, compared to where water caps were imposed. This is because a constraint on DD affects all irrigators – ground or surface water users – rather than just irrigators reliant on surface water. While this means that only the least efficient irrigators switch to dryland production, DD caps also affect the level of groundwater use, which has been set to the estimated 'sustainable' recharge rate according to the groundwater Water Sharing Plan (WSP). Assuming this recharge rate is correct and can be maintained indefinitely, reducing groundwater extractions below this level would result in a sub-optimal outcome since the full capacity of groundwater resources is not utilised. This is a common trend under each treatment where DD caps are imposed.

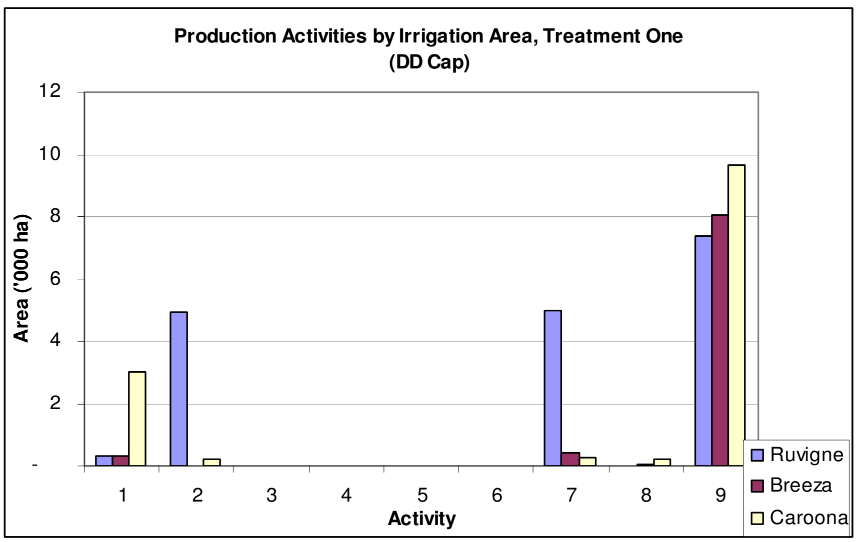

Figure 8.10: Production activities by irrigation area, Treatment One under DD Cap.

In Figure 8.10, the change in production activities under a DD cap of 20,000ML is disaggregated into irrigation areas. The impact of a DD target is more evenly spread between production under groundwater irrigation (activity 2) and surface water irrigation (activity 1), compared to a water cap. It can be inferred that DD targets aimed at reducing salinity would affect all irrigation areas, since both surface and groundwater use are affected by a cap on drainage. This suggests that the economic impact on irrigators located in Ruvigne irrigation zone would be relatively higher, while those located in Caroona and Breeza zones would experience lower economic impacts under a DD cap, relative to a water cap. However, the effect of DD caps on groundwater would be additional to the substantial reductions in groundwater entitlements that have already occurred in the Mooki. This may inflict unjustified costs on irrigators reliant on groundwater resources in downstream Ruvigne.

8.3.2 Treatment Two – with Alternative Irrigation Systems (AIS)

8.3.2.1 Scenario 2.1 – Base Case

Under Treatment Two, it is assumed that irrigators are given the option to invest in pivot irrigation systems (activities 3 and 4) or drip irrigation systems (activities 5 and 6) where it is profitable. Otherwise, irrigation could remain under the traditional furrow irrigation (activities 1 and 2). Water trading is still not possible, such that irrigators must produce only with the initial water allocations given.

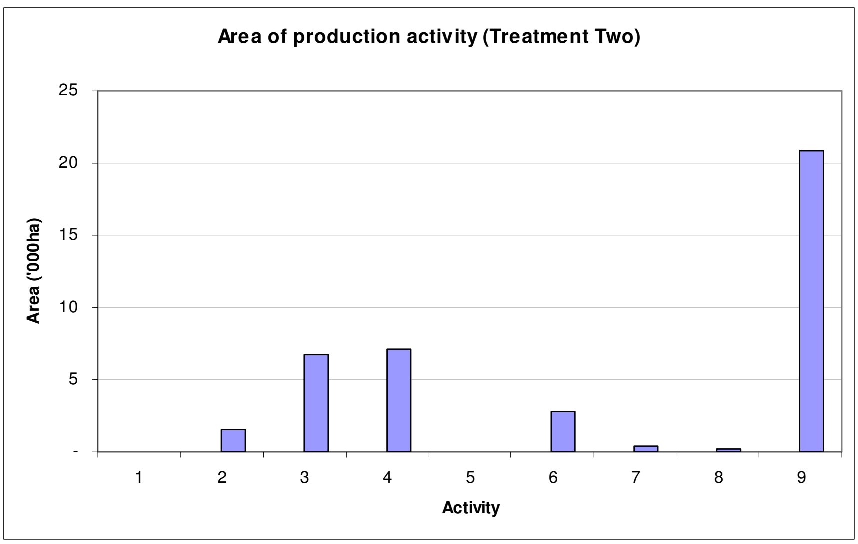

Figure 8.11: Scenario 2.1 production activities.

In Figure 8.11, it can be seen that most irrigators would opt for pivot irrigation systems, with almost equal areas sourcing surface (activity 3) and groundwater (activity 4). In fact, almost all surface water is used conjunctively with pivot systems. Only a handful of irrigation areas remain under furrow irrigation, sourcing groundwater (activity 2). This is rational because pivot systems allow for greater yields for the same level of water use as furrow irrigation. Some HRUs adopt drip irrigation conjunctively with groundwater (activity 6). This could be explained by the reliability of groundwater, which allows greater crop yields to be achieved since irrigation could occur regularly, and ensures the capital investments in drip systems (which have higher capital costs than pivot) are recouped. Dryland cotton (activity 9) still dominates the catchment, with a small portion producing dryland wheat (activity 7). Given the areas under dryland cotton and wheat are relatively unchanged compared to Treatment One, it can be inferred that many HRUs may in fact be more suitable to dryland crops. Environmental flows may be sourced from these areas at relatively low cost to the basin, since they are more profitable under dryland production.

Considering the activities by irrigation area, the most irrigation still occurs in Ruvigne which remains heavily reliant on groundwater (Figure 8.12). The difference is that pivot systems have become the most profitable irrigation technology, followed by drip irrigation systems and by furrow irrigation.

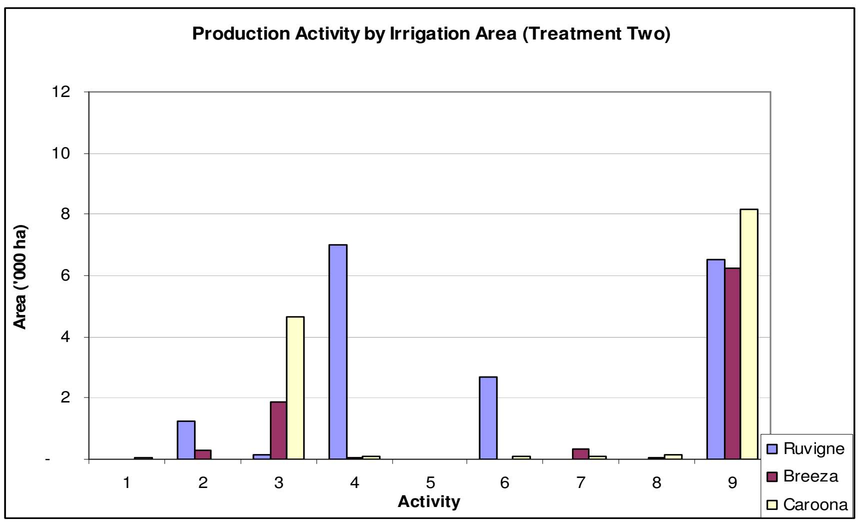

Figure 8.12: Production activities by irrigation area, Treatment Two Base Case.

It can be seen that, where AIS is used, the primary source of water in all irrigation areas does not change. However, it is almost always optimal to switch to water efficient technologies. The area under dryland cotton still dominates a significant portion of each irrigation area, which is similar to Treatment One. This reiterates the notion that water has not been efficiently distributed in such a way that reflects its highest value use. The use of AIS can improve the productivity of water; but this improvement is confined to the farm-level.

8.3.2.2 Scenario 2.2 – Water Caps

In Figure 8.13, the aggregate effect of water caps, under Treatment Two are presented. Water caps are again imposed at 55,000ML, 40,000ML and 20,000ML. The outcomes are compared to the base case surface water supply of 59,000ML.

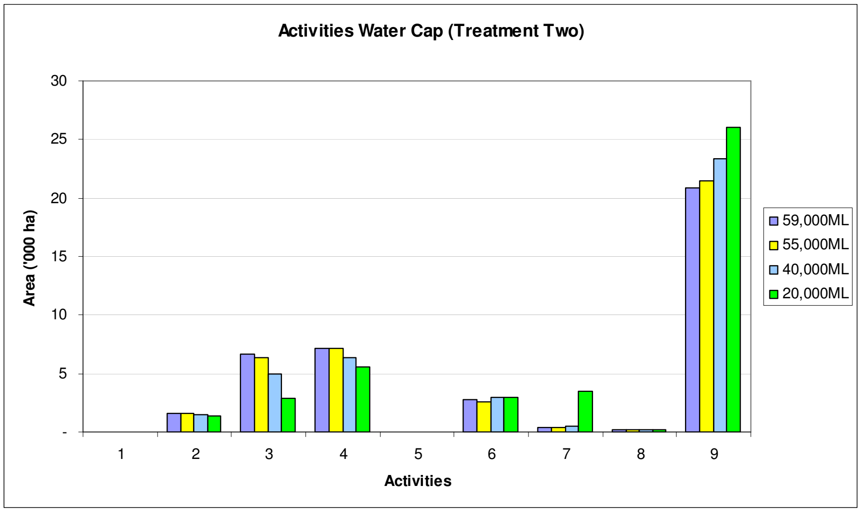

Figure 8.13: Scenario 2.2 production activities.

It can be seen that, where water caps are imposed, most of the HRUs that sourced surface water (activity 3) switch to dryland crops, while irrigated production sourcing groundwater (activities 2, 4 and 6) is again relatively unaffected. This result is logical since water caps only affect surface water users, and is a consistent trend under each treatment.

The impact of water caps by irrigation area is presented in Figure 8.14. Since almost no surface water is used in Ruvigne, much of the water reductions occur in Caroona and Breeza. The main impact of water caps is therefore inflicted on Caroona and Breeza irrigators, while Ruvigne is relatively unaffected. Similarly to Treatment One, where greater environmental flows requirements are imposed on the Mooki irrigators, the most economic impact would occur in the upstream irrigation areas.

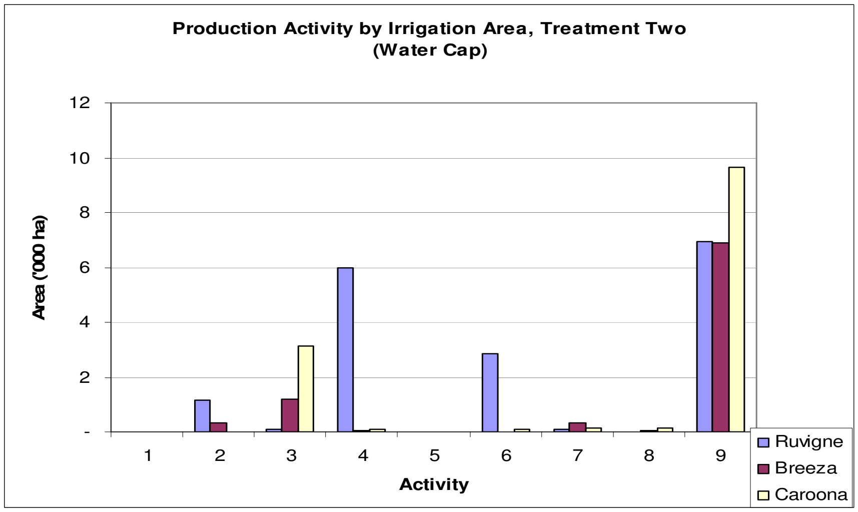

Figure 8.14: Production activities by irrigation area, Treatment Two under Water Cap.

8.3.2.3 Scenario 2.3 – Deep Drainage Caps

In Figure 8.15, the changes in production activities as DD caps are imposed at 20,000ML, 14,000ML and 10,000ML are presented. This is compared to the outcome under the unconstrained scenario with a DD of 25,000ML.

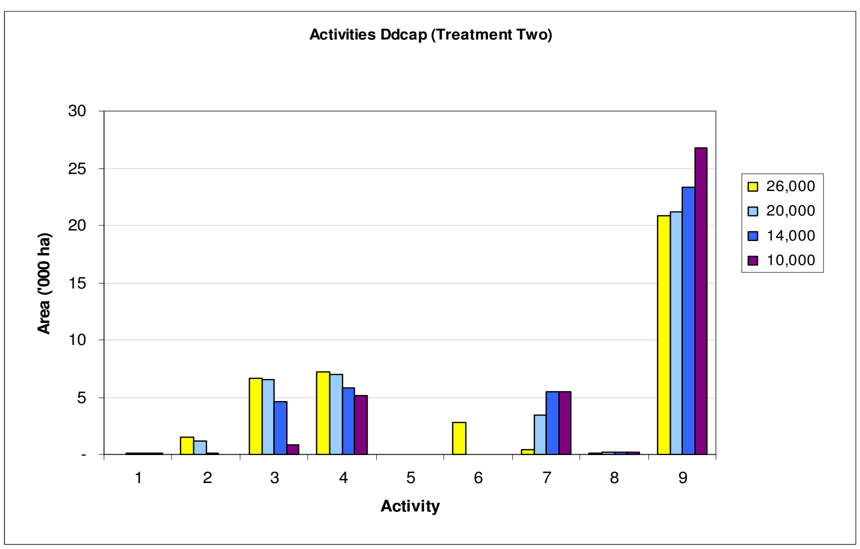

Figure 8.15: Scenario 2.3 production activities.

A considerable reduction in both surface and groundwater use is observed where DD caps are imposed. A distinctive change is the area under pivot irrigation sourcing surface water (activity 3), which falls significantly at stringent DD targets. While groundwater use (activities 4 and 6) is not reduced as dramatically as observed under Treatment One (Scenario 1.3), its extraction level nevertheless drops below the sustainable extraction rate for all DD constraints.

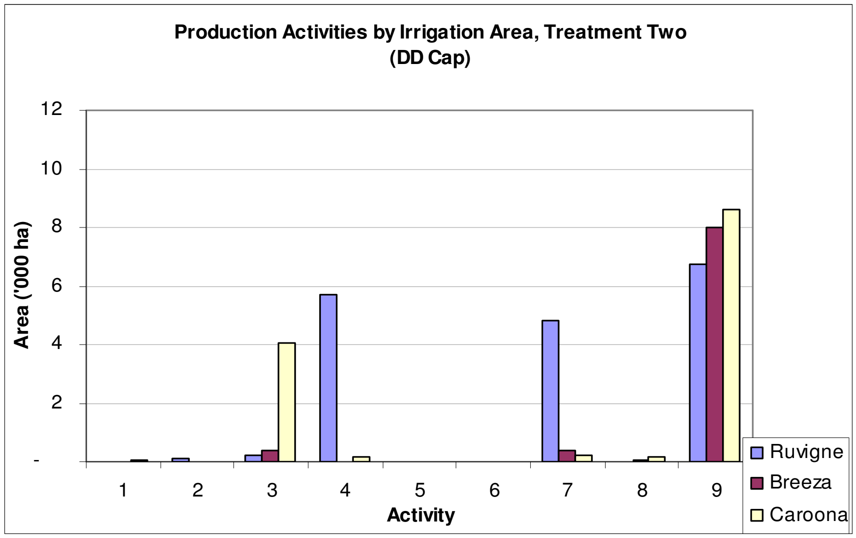

Figure 8.16: Production activities by irrigation area, Treatment Two under DD Cap.

The impact of DD caps according to irrigation area is shown in Figure 8.16. Much of the reduction in water use occurs in Ruvigne, where groundwater use is reduced significantly under activity 4 (pivot system) and activity 6 (drip system). Again, the economic impact of DD caps on groundwater may lead to excessive costs on irrigators reliant on groundwater resources, particularly in Ruvigne, due to the substantial cuts in groundwater entitlements that have already occurred.

8.3.3 Treatment Three – with Water Trade and Alternative Irrigation Systems (AIS)

8.3.3.1 Scenario 3.1 – Base Case

Under Treatment Three, water trading is introduced and irrigators have the choice to use AIS. Given the option to trade water and use water efficient technology, the area under dryland cotton (activity 9) falls considerably, with a corresponding increase in the area that under irrigated cotton. This is distinctly different to previous treatments without water trade, under which a significant portion of the landscape is under dryland crops (Figure 8.17).

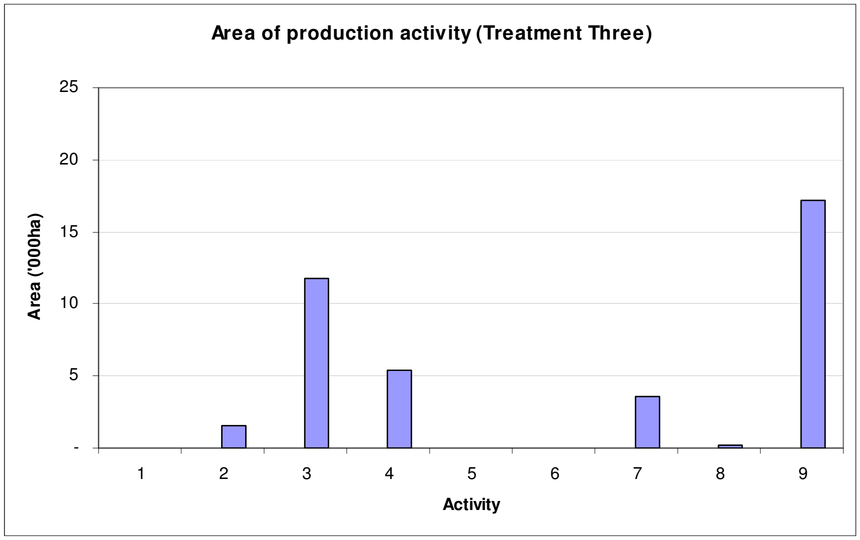

Figure 8.17: Scenario 3.1 production activities.

Based on the changes in production observed, it appears that under previous treatments, a significant area of the catchment was forced to produce dryland crops due to limited access to water. Therefore, where surface water could be obtained through the water market, these HRUs switch to irrigated cotton under pivot irrigation systems (activity 3). Pivot irrigation sourcing groundwater (activity 4) remains a prominent production activity, although drip systems (activity 6) become unfavourable, and are not used. Given the possibility of purchasing surface water, the HRUs that were using drip irrigation (activity 6) in previous treatments, appear to find more profitable to invest in pivot systems and source surface water instead. That is, where it was not possible to purchase water through the water market, these HRUs were limited by the availability of surface water and were confined to irrigating with groundwater under drip systems. This also implies that these HRUs had higher values for surface water than other HRUs that initially had a greater allocation of surface water.

Closer inspection of the production activities by irrigation area shows that much of the surface water is traded to downstream Ruvigne, such that almost all irrigation occurs in this area (Figure 8.18). While Caroona remains the second largest area under irrigated cotton, the proportion under irrigation has shrunk significantly compared to previous treatments.

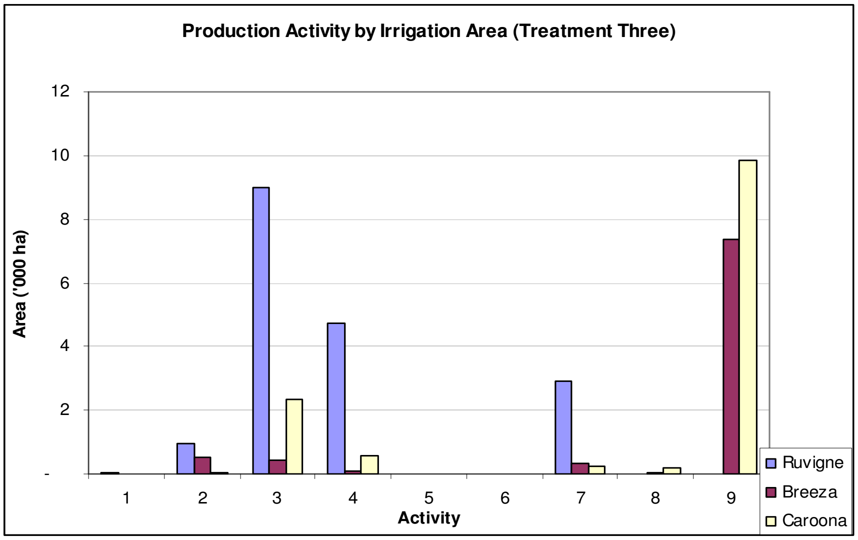

Figure 8.18: Production activities by irrigation area, Treatment Three Base Case.

This change in production pattern between Treatment Three and previous treatments can be explained by the location of Caroona and Breeza, which are the upstream-most irrigation areas with first access to surface water. Where no water trade is possible, surface water is extracted by these upstream irrigation areas, despite Ruvigne having the highest value for surface water. With water trade, surface water is almost completely traded downstream. This suggests that Ruvigne is the most productive irrigation area in the Mooki basin, and that basin profit can be maximised by establishing a system of water trade in this region. Since Upper Namoi is thought to have the highest conservation value, the trade of water downstream would result in a socially optimal outcome. Not only would the profitability of the basin be enhanced, the upstream environment could also be protected.

8.3.3.2 Scenario 3.2 – Water Caps

As per previous treatments, as water supply is reduced, the irrigated areas sourcing surface water (activity 3) fall and areas under dryland cotton (activity 9) rise. This is while areas reliant on groundwater (activities 2 and 4) remain relatively unchanged. Compared to treatments without water trade, however, surface water is distributed efficiently under Treatment Three such that its full value is realised, and only the least efficient surface water irrigators forego water use. This leads to lower overall opportunity costs compared to where water has not been distributed efficiently under previous treatments.

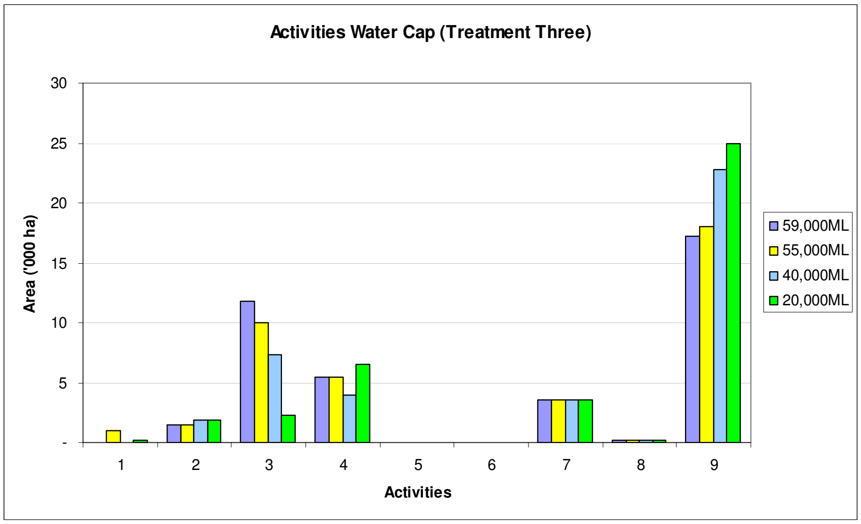

Figure 8.19: Scenario 3.2 production activities.

Considering the impact of water caps by irrigation area, it can be seen that the economic burden has also shifted from Caroona and Breeza to downstream Ruvigne (Figure 8.20). Under previous treatments, much of the economic impact occurs in Caroona and Breeza, since these areas relied on surface water. With water trade, most surface water shifts to Ruvigne from upstream irrigation areas, such that surface water caps affect only irrigators in this region.

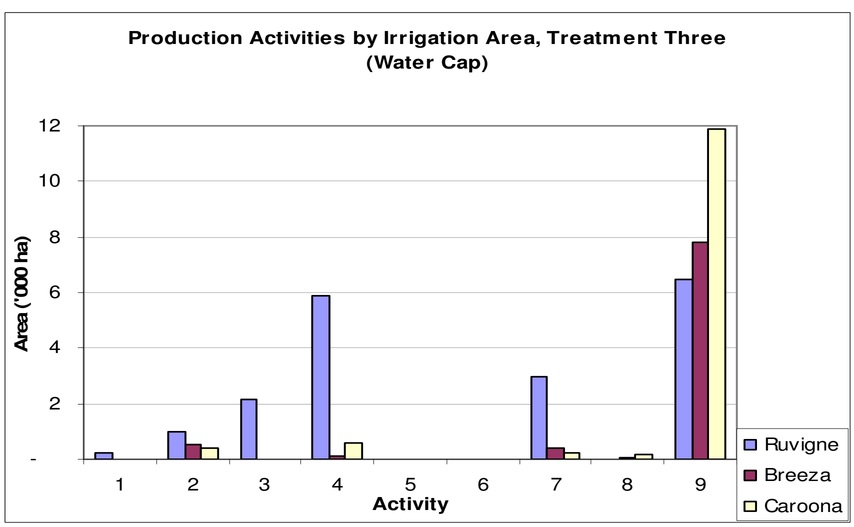

Figure 8.20: Production activities by irrigation area, Treatment Three under Water Cap.

While the economic impact is more pronounced in Ruvigne, irrigation still occurs in this region, while in Breeza and Caroona irrigation ceases completely. This suggests that, even where surface water allocations are cut significantly, Ruvigne remains the most productive irrigation area where much irrigated production would occur. This is while groundwater use (activities 2 and 4) is not affected.

8.3.3.3 Scenario 3.3 – Deep Drainage Caps

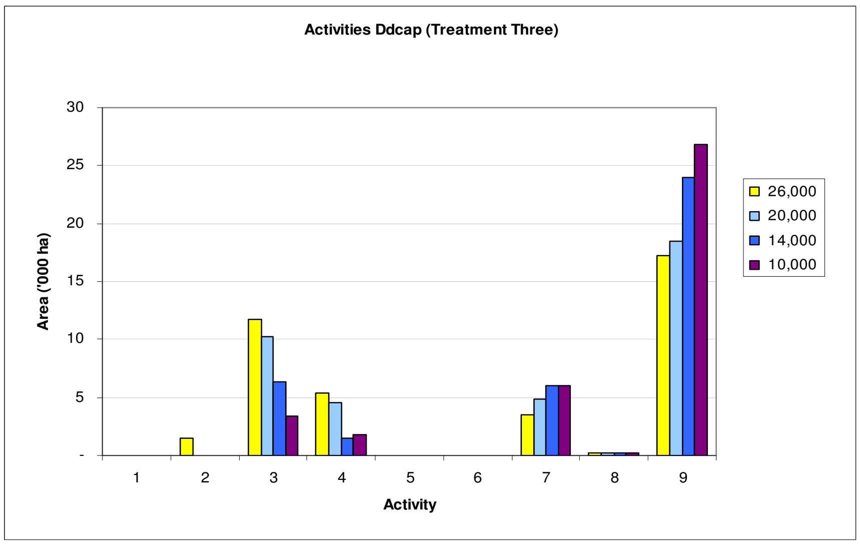

Figure 8.21: Scenario 3.3 production activities.

Where a DD cap is imposed under Treatment Three (Figure 8.21), a similar rate in water use reductions occur in pivot irrigation sourcing surface water (activity 3) and groundwater (activity 4). Compared to where water caps are imposed, however, it appears that the impact on groundwater use is more dramatic under a DD cap. Considering the changes in production by irrigation area (Figure 8.22), it can be seen that much of the impact of DD caps are also inflicted on Ruvigne since water is mostly used in this irrigation area.

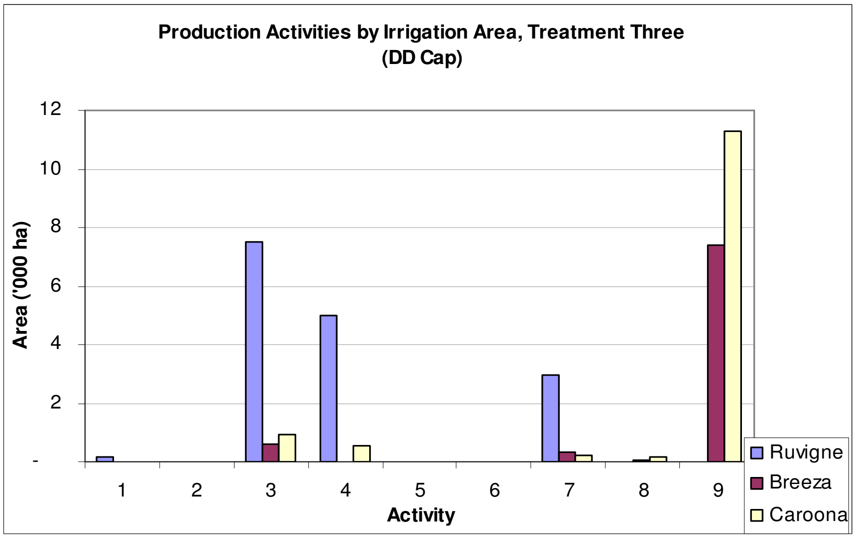

Figure 8.22: Production activities by irrigation area, Treatment Three under DD Cap.

As DD caps are tightened, groundwater use under furrow systems (activity 2) ceases in all irrigation areas. This is while pivot irrigation sourcing groundwater (activity 4) and surface water (activity 3) are also reduced considerably. Therefore, DD caps will affect both water sources, with the greatest impact being in Ruvigne. This is the case regardless if DD caps or water caps are imposed, since Ruvigne has the largest area under irrigation. This is in contrast to previous treatments where the upstream irrigation areas bear greater economic losses under water caps, because water has not been distributed efficiently. Where water trade is possible, the overall opportunity cost of meeting environmental targets would be much lower, because water has been shifted to its highest value use (in Ruvigne).

8.3.4 Shadow Prices for Water

The shadow values for surface water (Sw) and groundwater (Gw) were obtained for the Base Case scenario under each treatment for the HRUs in irrigation areas Ruvigne, Caroona and Breeza. The shadow price for each HRU within the irrigation area were summed then averaged for Treatments One and Two, since the value under non-trading treatments vary between HRUs. A single shadow price is reported for all irrigation areas under Treatment Three since it is the same for all HRUs in the presence of a water market. The shadow prices obtained in this section help shed light on the trends in changes in production activities observed, and can assist in explaining results in the subsequent sections in this chapter.

It can be seen that under the status quo (Treatment One), Ruvigne has the highest shadow value for resources, followed by Caroona and then Breeza (Table 8-6). This corresponds to reality, since it is known that most cotton irrigators are located within Ruvigne, with fewer irrigators in the other areas (Smith 2006, pers. comm.). Considering the production activities in Figure 8.6, much of the surface water was initially used in the upstream Caroona and Breeza regions. This confirms the finding that at the outset surface water resource was not allocated efficiently to where it has the highest value. Groundwater appears to have the highest value in Caroona, although only slightly greater than Ruvigne.

Table 8-6: Shadow values under Treatment One Base Case.

Treatment One	Ave. Sw shadow value	Ave. Gw shadow value
Ruvigne	118.69	68.05
Breeza	73.19	49.70
Caroona	87.24	71.61

Under Treatment Two (no water trade, with AIS), the value of surface water is increased for all irrigation areas; with a significant rise observed in Ruvigne (Table 8-7). This is while the shadow price of groundwater is reduced (or unchanged) for all areas. This could suggest that, where AIS are available, the value of surface water is enhanced such that the relative value of groundwater falls.

Table 8-7: Shadow values under Treatment Two Base Case.

Treatment Two	Ave. Sw Shadow value	Ave. Gw shadow value
Ruvigne	129.79	47.51
Breeza	80.29	49.70
Caroona	89.28	64.26

As expected, where water trade is introduced under Treatment Three, the shadow value of surface water is equated across all irrigation areas (Table 8-8). Much of the water was traded to the downstream area of Ruvigne, where, in the absence of a water market, the shadow value for surface water was highest. The equilibrium shadow price is $111.45/ML, which represents the market-clearing price in the water market. The equilibrium quantity traded is 37,500ML and much of the water is moved into downstream Ruvigne. This reflects the changes in production activity observed in Figure 8.18, which shows that almost all irrigated production occurs in this irrigation area. Under scenarios without water trade (Treatment One and Treatment Two), much of the surface water is used upstream, although it can be seen from the shadow price that it has the highest value in downstream Ruvigne.

Table 8-8: Shadow values under Treatment Three Base Case.

Treatment Three	Sw shadow value (with Gw)	Sw shadow value (w/o Gw)[9]	Ave. Gw shadow value
Ruvigne	111.45	148.91	62.27
Breeza			44.19
Caroona			70.48

The equilibrium shadow price obtained from the model reflects the empirical water market prices observed for the Namoi. From the WaterExchange website (WaterExchange 2007), the market price for temporary water trade in the regulated system in Namoi averages $100/ML, with a high of $120/ML over the last season. This market price seems to correspond well to the shadow value of $111.45/ML for the Mooki unregulated system. This validates the economic model, which generates a shadow value for water similar to that observed in the water markets in the Namoi region. Furthermore, it appears that there is scope for water trade to occur between the regulated (downstream) and unregulated (upstream) systems within the Namoi Valley, perhaps with more surface water flowing towards the regulated areas. This may result in an efficient outcome, due to the conservation value of Upper Namoi, given the presence of high value species and wetlands (Hudson 2005, pers. comm.; DLWC, 1998). However, the extent of trading may be limited since the marginal values are very similar. In addition, in the absence of groundwater, the shadow value of surface water grows to $148.91/ML. This has implications for future reductions in groundwater entitlements, which may create a

[9] Two shadow prices for surface water were calculated: one with full groundwater allocations available and one without (w/o) any groundwater supply available. This was to determine the value of surface water in the absence of an alternative water source.

greater demand for surface water in the Mooki and cause water to be traded upstream from Lower Namoi.

8.3.5 Summary

The results presented above have useful applications for policies designed to target water use or salinity reduction, and the impact it has on basin crop production. It can be seen that the impact of environmental targets, in the form of environmental flows or salinity reduction, will have varying impacts on production activities depending on the irrigation technology used and the possibility for water trade.

Initially, without water trade, the main areas reliant on surface water are upstream Caroona and Breeza, while downstream Ruvigne relies mainly on groundwater. Where DD caps are imposed to mitigate salinity contribution, both surface and groundwater users are affected since the DD cap indiscriminately reduces drainage from both water sources. Much of the impact occurs in Ruvigne, because groundwater is used primarily in this area. However, DD caps cause groundwater extraction to be reduced below its sustainable level, which leads to a sub-optimal outcome because the full capacity of groundwater resources is not used. In addition, surface water extractions are relatively less affected which means that a smaller amount of environmental flow is generated.

On the other hand, where the objective is to achieve greater environmental flows through limiting surface water extraction by imposing a cap, only irrigators sourcing surface water are affected. This means groundwater use remains at the estimated sustainable extraction rate according to groundwater Water Sharing Plans. Without water trade, much of the economic burden of water caps is incurred upstream in Caroona and Breeza. However, where water trade is possible, water is traded downstream to Ruvigne where it has the greatest productive value. Ruvigne becomes the main area of irrigated production. This suggests that, rather than clawing-back surface water from upstream Caroona and Breeza, where most surface water is initially used, it may be optimal to rely on market mechanisms and encourage surface water to be traded downstream. This allows inefficient users upstream to voluntarily exit the irrigation industry, and also enables environmental flows to be sourced at the least-cost. There is an additional benefit in that irrigation is concentrated in Ruvigne and reduced in upstream Caroona and Breeza, leaving a greater area of Upper Namoi for environmental conservation purposes.

Overall, it can be concluded that environmental targets would have different impacts on basin crop production, depending on the technological setting and the possibility for water trade. The associated opportunity cost of achieving environmental targets will therefore vary under these different circumstances, and will also depend on how stringent the targets are. The cost of achieving environmental targets, and how they can be achieved at least-cost, is considered in the following sections.

8.4 ESTIMATED COSTS OF ENVIRONMENTAL FLOWS

In this section, the cost to the irrigation industry associated with meeting environmental flow targets is analysed. A comparison of the total cost (TC) of achieving environmental flow targets, between each treatment, is made. A TC function is estimated for each scenario under Treatment One (status quo), Treatment Two (no water trade, with alternative irrigation systems – AIS) and Treatment Three (with trade and AIS). The effect of increased environmental flows was simulated by successively reducing surface water allocations from 59,000ML to zero, and the TC functions are calculated relative to the profit under Treatment Three. That is, the cost functions obtained represent the opportunity costs of providing environmental flows, relative to the basin profit in the presence of water trade, without requirements for additional environmental flow above that stipulated in the surface Water Sharing Plan.

As can be seen in Figure 8.23, these cost functions are fairly linear. This is an artefact of the assumption that, as water supply is reduced irrigators do not reduce water use per hectare. Instead, the area under irrigation is reduced such that each area receives the full crop water requirement. In this way, the value of water is reflected in the additional *area* that can be irrigated and translates to linear cost curves as water supply declines.

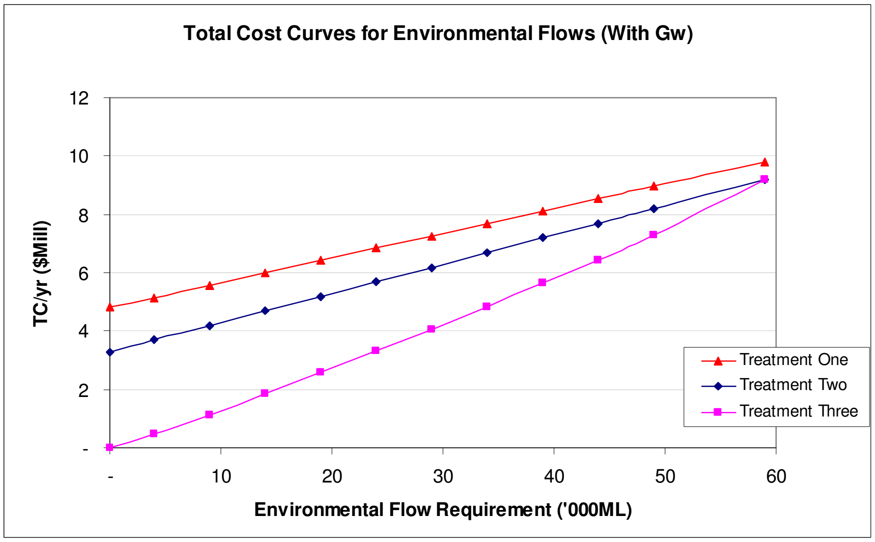

Figure 8.23: Total cost of meeting environmental flows.

The difference between the TC functions for Treatment One and Two represents the value of having the option to choose among various alternative irrigation systems (AIS), while the difference between TC functions for Treatment Two and Three represents the value of having a water market in the catchment, for any given environmental flow requirement. The cost difference between the functions suggests that, when the option of choosing among various AIS is present the total opportunity cost of meeting environmental targets can be reduced by $1.5Mill/yr. When the water market is in place in the catchment, the opportunity cost can be reduced by $3.3Mill/yr. The distance between the TC functions under each treatment is the greatest at low environmental flow requirements, and becomes smaller as environmental flows increase. That is, as the environmental flow requirements increase, the values of having the water market or having the option of AIS both diminish, and tend towards the same point. The non-convergence where all extractive water is reallocated to environmental flows is due to the available groundwater, which has a different value under different technological settings (water has a different value where AIS is used since it is used more productively, compared to furrow irrigation).

This trend implies that the benefit of AIS and water market is limited if extractive water use is reduced substantially, since the use of water trading or AIS can only do so much to reduce the opportunity cost incurred. If a significant amount of extractive water is reallocated towards the environment, there would be significant economic costs imposed regardless of what adjustment mechanisms are available. Unless there is a high valuation of the environmental benefits that the conserved water would provide, it may not be efficient to reallocate a substantial share of extractive water towards environmental purposes. The efficient allocation of water between extractive and non-extractive uses should be where the marginal value in production equates with the marginal value of water for environmental purposes.

8.4.1 Marginal Costs of Environmental Flows

The equilibrium shadow values of surface water, under water trade (Treatment Three), are presented in Figure 8.24. These shadow prices also represent the marginal cost of providing extra environmental flows diverted from extractive allocations. Where the catchment manager wishes to source environmental flows from the water market, the shadow values provide a useful guide for the marginal cost of additional environmental flows. It can be seen that the value of surface water is relatively constant, since the shadow price is unchanged for a range of surface water allocations. Again, this is an artefact of the assumption that irrigators use extra water to expand production. The shadow value of water reflects the additional area that could be irrigated, rather than the increased rate of irrigation. This also explains the relatively constant TC curves observed in the environmental flow cost functions in Figure 8.23. In addition it suggests that, for various environmental flow targets, the marginal costs are not different and a greater volume of water could be obtained without increasing the marginal cost incurred by irrigators.

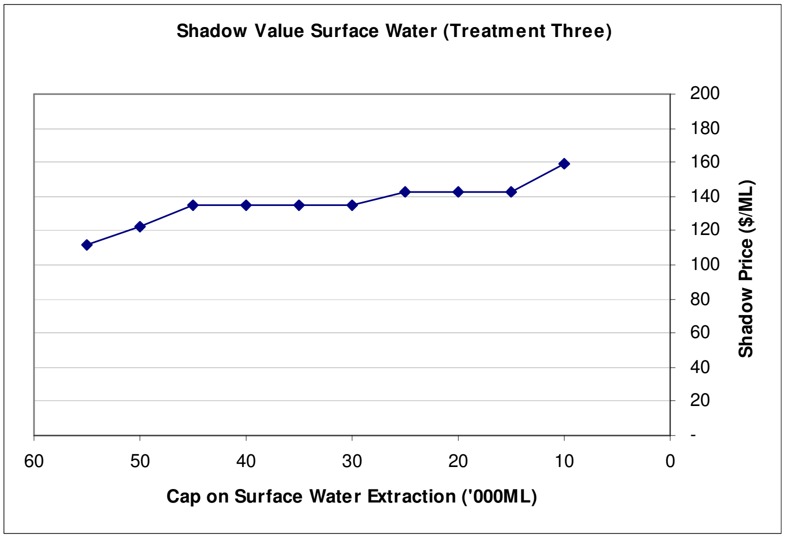

Figure 8.24: Shadow value of surface water at different surface water allocations, under Treatment Three.

8.4.2 Summary of Estimated Costs of Environmental Flows

It can be seen from the above results that investing in water efficient technologies and establishing a water market will be valuable for the Mooki basin, since it can improve the productivity of the irrigation sector significantly, as well as minimise the opportunity cost of securing environmental flows in the river. For the Mooki, it appears that the marginal cost of providing additional environmental flows is fairly constant, and environmental flows could be increased without increasing the marginal cost incurred by irrigators. While this suggests that setting a higher environmental flow standard will not increase the marginal economic impact, the overall cost will still be high. Efficient water allocation should be where its marginal value in production equates with the marginal value of water for environmental purposes.

8.5 DUAL INSTRUMENTS: DEEP DRAINAGE AND SURFACE WATER CAPS

In the previous section, the costs of achieving environmental flows were presented in order to assess the impact of environmental flow objectives, and how alternative irrigation systems (AIS) and water trade can reduce the economic burden. Suppose now that the catchment manager is also concerned about the level of deep drainage (DD), and must evaluate the usefulness of DD caps as an additional instrument to manage salinity, independently of environmental flow policies.

In this section, the economic cost of imposing DD constraints on the basin, in conjunction with surface water constraints, is analysed. The outcomes represent the impact of 'dual-instruments' that may be used by a catchment manager to control salinity and environmental flows separately. These results could shed light on the current situation in NSW catchments, whereby an end-of-valley salinity target has been imposed as part of the MDB Salinity Management Strategy. This has been introduced on top of surface Water Sharing Plans for individual valleys in NSW, with the prospect of increased environmental flows in the near future. The question is whether the combined use of a separate instrument is useful in controlling the 'joined' pollution, given water use and DD are interrelated.

Firstly, a comparison of the resource use, profit and the associated salt load, under the Base Case scenario of each treatment, is presented. This is done in order to appraise the estimated level of DD and salinity contribution from the Mooki, and put into perspective the effect of water and DD instruments. This is followed by an assessment of using dual-instruments to manage salinity risk, under Treatment One (status quo), Treatment Two (no water trade, with AIS) and Treatment Three (with water trade and AIS).

8.5.1 Salt Loads

A summary of outcomes under the Base Case scenario of each treatment is presented in Table 8-9.

Table 8-9: Comparing outcomes under Base Case scenarios.

Base Case scenarios						
Treatment	Profit/yr ($Mill)	Deep drainage (ML/yr)	Salt load (t/yr)	Surface water use (ML/yr)	Groundwater use (ML/yr)	Total water use (ML/yr)
One	35.32	25,419	8,693	53,628	56,059	109,687
Two	36.85	24,200	8,277	58,698	56,200	114,898
Three	40.15	24,710	8,451	59,000	56,241	115,241

The EC reading[10] for Mooki was reported as 534µS/cm, which means each megalitre of water carries 342kg of salt. This is based on the assumption that 1,000µS/cm equates to 640kg of salt per megalitre (NSW DPI 2006b).

It appears that DD and associated salt load are fairly similar between the three treatments, although there is a significant difference between the levels of water use. As observed in earlier results, water efficient technologies (AIS) seem to significantly improve water use efficiency. A reduction of 5% in DD is observed under Treatment Two (no trade, with AIS) relative to Treatment One (status quo). Where water trade is introduced (Treatment Three), DD and salt loads increase by 2% relative to Treatment Two, although a net reduction of 3% is still achieved relative to the status quo. This suggests that, even without additional policy instruments, DD and salt load could be partially reduced by simply encouraging the use of AIS and water trading in the basin.

The salinity concentration of water in the Mooki (534 µS/cm) is not a significant concern for the Namoi, since it is not sufficient to cause crop damage to cotton which has a salt tolerance level of 1,700 µS/cm. However, the main implication of saline return flows is its downstream impact on the Barwon-Darling system. It is thought that Mooki and the Peel River are the main contributors of salinity to the Namoi catchment. Considering that the Namoi end-of-valley salt load target is 127,600t/yr, the estimated contribution from the Mooki, of around 8,693t/yr, represents just 7% of the target. However, the salt contribution from Mooki relative to its area is moderately high. The Mooki study area, of 397km^2, comprises 1% of the total area of Namoi of 41,998km^2. This suggests that the proportional salt contribution from Mooki should be 1% of 127,600t, or 1,276t/yr. Compared to the estimated salt load of 8,693t/yr, the salt input from the Mooki is seven times greater than the proportion it may be expected to be contributing.

Where the objective is to reduce salt load below the Base Case scenario level of 8,693t/yr, an additional instrument to control DD contribution may be required and would

[10] Electrical conductivity (EC) is the measure for water salinity.

be imposed separately to surface water caps to control the conjoint pollution. The results of an analysis of such a dual-instrument are presented in the next section.

8.5.2 Dual-Instruments

Under each of the three treatments, DD constraints were reduced successively from 26,000ML to zero drainage. This is done under different water cap scenarios, holding available surface water at 59,000ML, 45,000ML and 25,000ML. The DD constraint is set on a basin-scale, such that the total DD across the 53 HRUs cannot exceed the target drainage level. The opportunity cost under each treatment are in terms of annual profit, and is relative to the profit under Treatment Three (with water trade), at 59,000ML surface water allocation without any constraints on DD. At zero allowed drainage, the HRUs are forced to produce dryland crops only. The total cost (TC) at zero DD is therefore a proxy for the value of irrigated crops, given the surface water allocation and treatment. To better convey the changes in TC between treatments, the results are discussed in the order: Treatment Three (water trade and alternative irrigation systems – AIS), Treatment Two (no water trade with AIS), Treatment One (status quo: no water trade, no AIS).

8.5.2.1 Treatment Three

Under Treatment Three, the impact of DD caps is considered in the presence of a water market and AIS. This is done while simultaneously capping surface water at 59,000ML (full season water allocation), 45,0000ML and 25,000ML.

For the set of TC functions in Figure 8.25, the impact of DD constraints are exemplified through the *slope* of the curves, reflecting the change in shadow price at different DD targets. The impact of reduced surface water availability is shown through the *shift* in the TC curves, which reflects the opportunity cost of diminished water availability relative to the full-scale water supply of 59,000ML.

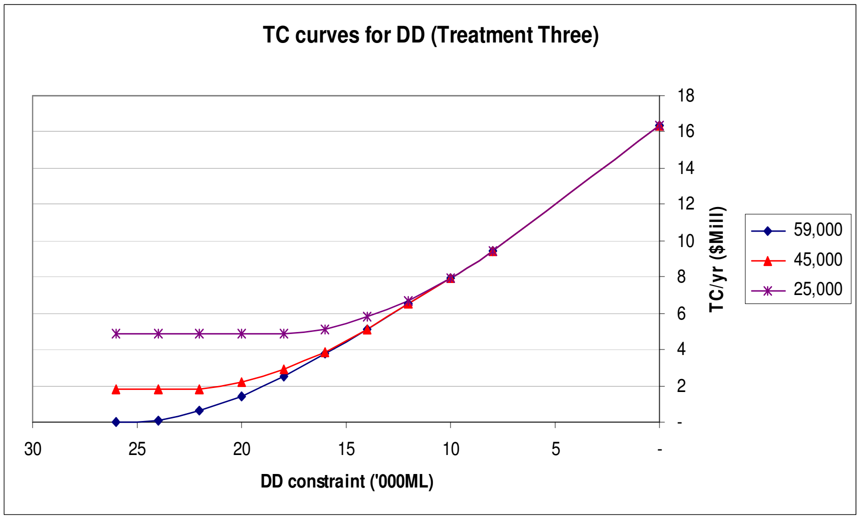

Figure 8.25: Total cost (TC) of DD constraints under Treatment Three, with Water Caps.

In the presence of a water market and AIS, the slope of the TC functions are initially different across the water constraint levels, but converge as the DD target falls below 13,000ML. This suggests that, at tight DD constraints, the shadow value of DD is unchanged under different surface water caps, such that the impact of DD targets does not vary much for different water constraint levels. That is, if a stringent drainage constraint is set, it would not affect the marginal cost incurred by the irrigators. On the other hand, at lax DD targets, the marginal cost of drainage reduction depends on the water availability. It appears that for some water caps, the marginal cost of DD is actually zero. This is implied by the point of inflection in the TC functions, whereby at lower water availabilities its slope does not begin to rise until tighter DD targets. For example, with a water constraint of 25,000ML, the marginal cost of DD targets is zero until the DD target falls below 16,000ML. Under a water constraint of 59,000ML, the marginal cost is $630/ML for the same drainage target. This is because when water use is reduced, DD is invariably reduced also. Therefore, it appears that stringent water and DD constraints can be simultaneously imposed without additional economic impact. However, it is perhaps meaningless having an extra constraint on DD if its occurrence is already limited by the reduced availability of the surface water supply. This is a consistent trend observed under each of the treatments, which are discussed in turn below.

8.5.2.2 Treatment Two

Under Treatment Two, the impact of DD caps are considered where there is no possibility to trade water in the water market; however there is the option to invest in AIS. Like in Treatment Three, water caps are simultaneously imposed at 59,000ML, 45,000ML and 25,000ML.

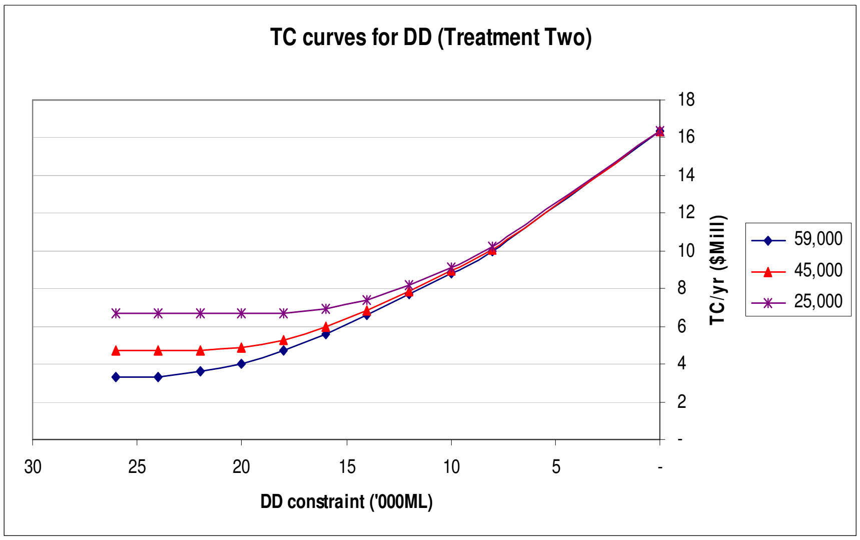

Figure 8.26: Total cost (TC) of DD constraints under Treatment Two, with Water Caps.

Under Treatment Two (no water trade, with AIS), the TC functions shift up relative to Treatment Three, reflecting the opportunity cost of water trade (Figure 8.26). The impact of DD is exemplified through the slope of the TC functions, which are slightly less steep than Treatment Three. This suggests that, without water trade, the shadow price of DD is lower because water has not been reallocated to its highest value use. Therefore without water trade, the marginal impact of DD caps are less than when water market is in place, although the total opportunity cost of no trade (expressed through the upward shift in TC functions) is significant.

Similarly to Treatment Three, the slopes of the TC functions initially vary at lax drainage caps. Its marginal cost under low surface water availability is low and sometimes zero, as implied by the flatness of the TC functions. The inflection in the TC does not occur until tighter drainage constraints, since DD occurrence is already reduced by the water cap. For example, under a surface water allocation of 25,000ML, the opportunity cost of reducing DD is zero until DD targets fall below 16,000ML. For the same DD target, at a higher water allocation of 59,000ML, the marginal cost is positive, at $430/ML. This reiterates the finding under Treatment Three, which suggests that an additional DD instrument may be superfluous if a water cap has already been implemented.

8.5.2.3 Treatment One

Under Treatment One, only furrow irrigation is used and there is no opportunity for water trading. DD caps are imposed while holding surface water caps at 59,000ML, 45,000ML and 25,000ML.

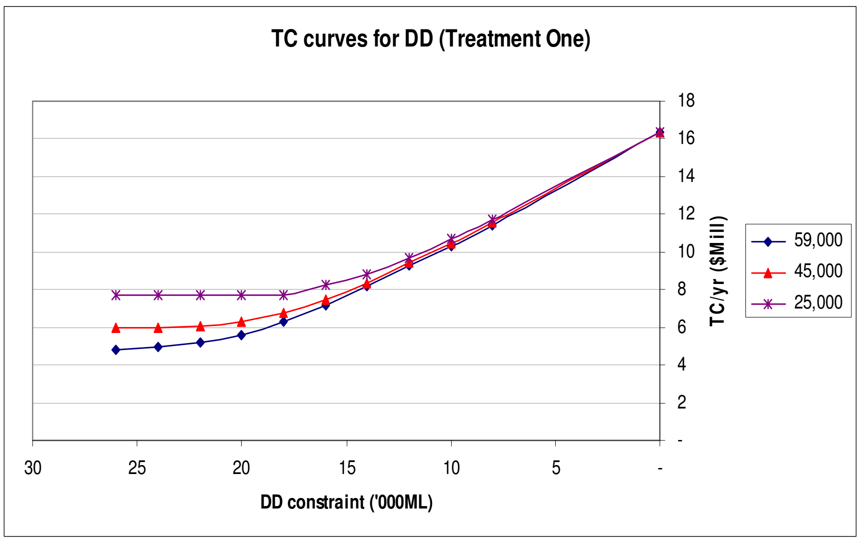

Figure 8.27: Total cost (TC) of DD constraints under Treatment One, with Water Caps.

Under the status quo (Treatment One), the TC functions shift up further compared to Treatment Two, which represent the value of having options to choose AIS to the irrigators in the catchment (Figure 8.27). The shadow price of DD is also reduced relative to previous treatments, as indicated by the gentler slopes. This is due to the lack of water trading to shift water to its highest value use, and also because water is used inefficiently under furrow irrigation so each unit of DD has a lower marginal value. Consequently, irrigators incur lower marginal costs for reducing drainage under Treatment One. However, the overall cost of not using AIS or the water market is significant, as indicated by the upward shift in the TC functions.

As was the case previously, the slope of the TC appears to be relatively unaffected by the water availability at tight drainage constraints. This result reiterates the finding that, even with stringent DD targets, it is not significantly more costly to reduce drainage given there is no substantial difference in the marginal cost of DD under different water supplies. The main difference between the TC functions is the point of inflection, which occurs at a more stringent DD constraint as water supply is reduced. With a 25,000ML water cap, the marginal cost of DD is zero for some drainage target levels, whereas the same DD at higher water supplies has positive marginal cost. This is a consistent trend, which is also observed under previous treatments.

8.5.3 Summary of Results for Dual-Instruments

The implication from the above results is that, although it may be difficult determining an efficient DD target at a basin scale, the difference in the *marginal* cost of reducing DD while simultaneously imposing water caps will not be excessive. This is evidenced by the similarity in the slope of the TC functions for drainage reduction, across different water caps. This implies that the marginal cost of reducing DD is independent of water supply for most DD targets, and may be set separately. The *total* cost incurred for stringent drainage targets, however, is considerable, and at tight DD constraints the overall economic impact on irrigators will be significant.

Under stringent water cap levels, for some DD constraints the marginal cost to reduce drainage is zero because drainage occurrence is limited by water availability. While this suggests that surface water and DD constraints can be jointly imposed without additional economic impact, an extra instrument to control the conjoined pollution, in the form of DD, may be unnecessary. This is because its occurrence is already reduced by low surface water supply.

It therefore stands to reason that water caps on its own would suffice in achieving DD (salinity) reduction. However, water caps also have associated opportunity costs of production. If one instrument suffices in achieving the desired outcome in water resource use or in salinity reduction, the question then becomes, which instrument could achieve the objective most cost-effectively. A comparison between DD and surface water caps are made in the following section, and will illustrate the effectiveness of each instrument under different treatments.

8.6 SEPARATE INSTRUMENTS: DEEP DRAINAGE v SURFACE WATER CAPS

The efficacy of deep drainage (DD) and water instruments to independently achieve DD reduction, under each treatment, is presented in this section. The total cost (TC) of each instrument under Treatment One (status quo), Treatment Two (no water trade, with alternative irrigation systems – AIS) and Treatment Three (with water trade and AIS) is examined. The opportunity cost is relative to the annual profit where full water allocations are received and with no constraint on DD, under the given treatment. Where the drainage or water cap is zero, all producers must grow only dryland crops to satisfy the constraint. This causes the TC function under each instrument to converge at the origin.

8.6.1 Treatment One

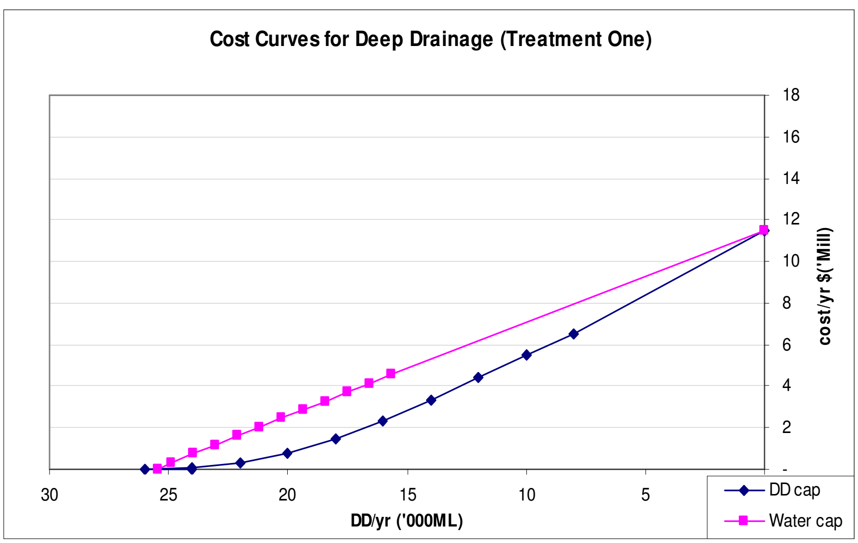

Figure 8.28: Total cost (TC) of deep drainage reductions under Treatment One, comparing instruments.

In Figure 8.28, the TC functions of achieving DD reductions under Treatment One (Status Quo), are presented. As the DD target becomes stringent, the shadow price of DD with surface water caps (pink line) appears to be fairly constant, since the TC function is linear. In contrast, when DD caps are imposed (blue line), the shadow value appears to increase at a slightly increasing rate.

Under a water cap, only irrigators sourcing surface water are forced to forgo water use, whereas under a DD cap both users of surface and groundwater are affected. Since the constraint on DD causes the most inefficient irrigators to forego water use regardless of the water source, the DD cap has a cost-advantage of up to $2Mill/yr. However, water caps have an additional benefit of generating extra environmental flows since it reduces only extractions from the river. On the other hand, under a DD cap, in-river extractions are not reduced as significantly (Figure 8.29).

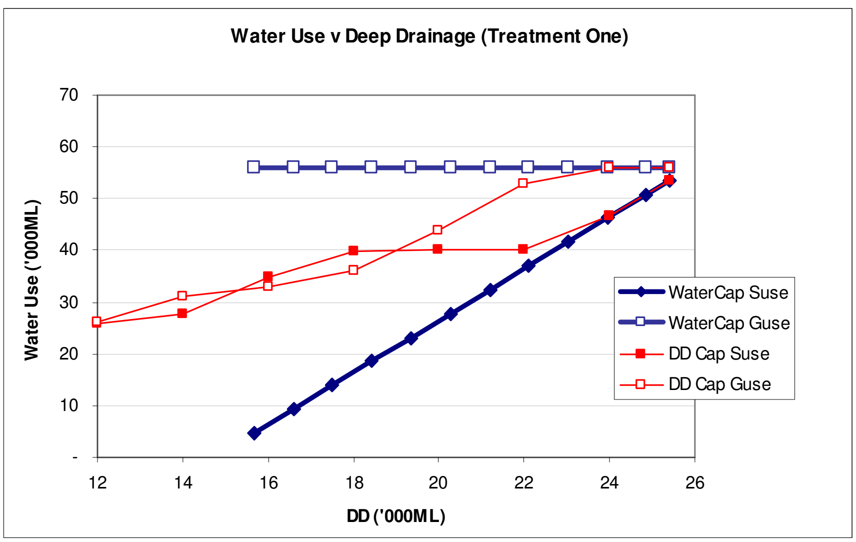

Figure 8.29: Water use under each instrument, Treatment One.

It can be seen that with water caps (Figure 8.29, blue lines), only surface water use (Suse) is reduced while groundwater extractions (Guse) remain unchanged. However, with DD caps (red lines), some reduction in groundwater use occurs as DD constraints become stringent. This is because DD caps impinge on overall water use, such that both surface and groundwater users are affected. While this allows the least-cost way of reducing DD, it also means that there is less environmental flow provision under DD caps for most target levels. Furthermore, groundwater extractions are reduced to a level below the estimated sustainable extraction rate according to the groundwater Water Sharing Plan. Water caps allow groundwater extractions to be maintained at the sustainable rate of use, while at the same time generating greater environmental flows (Figure 8.30).

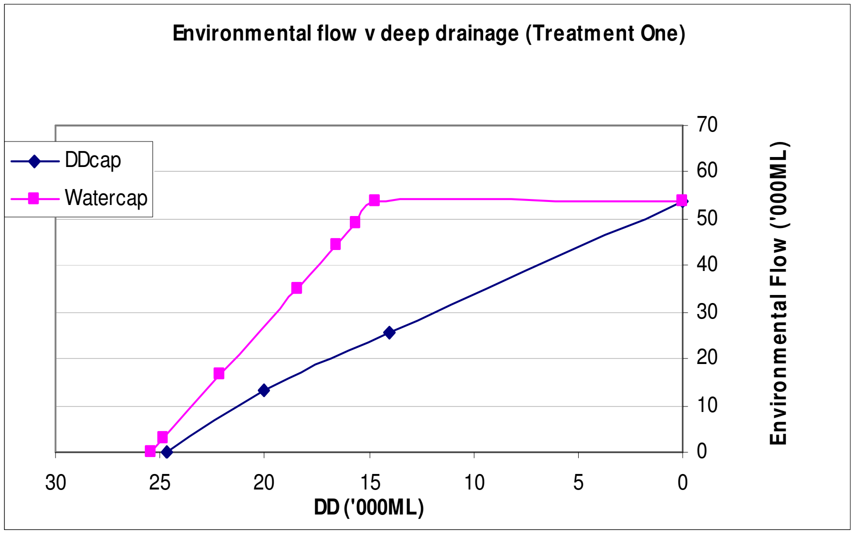

Figure 8.30: Environmental flows v deep drainage, Treatment One.

The above figure shows the environmental flow generated for a given drainage occurrence, under each instrument. It can be seen that, for all drainage levels, water caps have a significant advantage over DD caps in generating environmental flows. If the catchment manager has the dual-objective of providing for environmental flows and DD reduction, water caps could achieve this objective more cost-effectively than DD caps for most target levels.

8.6.2 Treatment Two

The results under Treatment Two (no trade, with alternative irrigation systems (AIS)) are similar to results under Treatment One, except the cost differential between the two instruments is diminished (Figure 8.31). The TC function under a water cap is again fairly linear, which implies that the shadow price on the DD constraint is constant where water cap is imposed. Where a DD cap is used, the TC function increases at a slightly increasing rate.

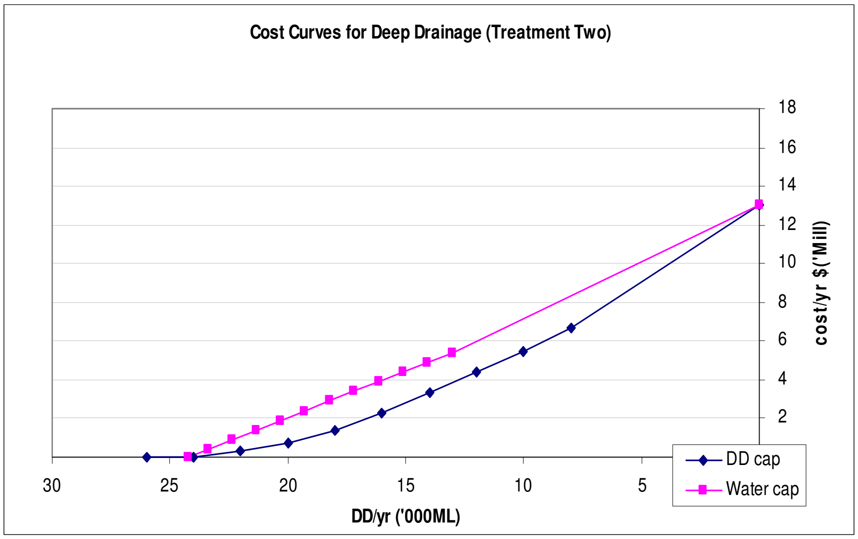

Figure 8.31: Total cost (TC) of deep drainage reductions under Treatment Two, comparing instruments.

Like Treatment One, DD caps affect overall surface and groundwater use, and allow for the least-cost means of achieving DD reduction. However, it also reduces the rate of groundwater use to below the estimated sustainable level of extractions, according to the groundwater Water Sharing Plan. This can be seen in Figure 8.32.

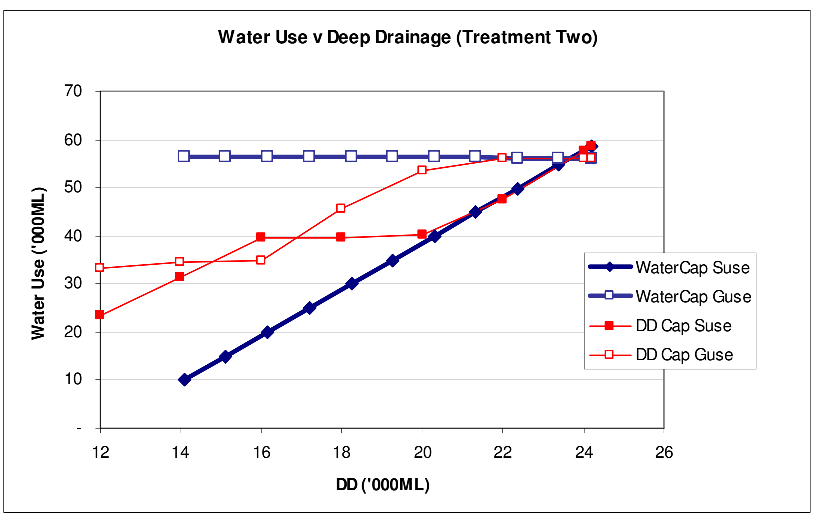

Figure 8.32: Water use under each instrument, Treatment Two.

The two instruments appear to have similar impacts on surface (Suse) and groundwater use (Guse) initially, at lax levels of DD constraint. However, for drainage constraints below 20,000ML, DD caps leads to significant reductions in groundwater use. Again, this brings groundwater extractions below the sustainable level, as well as resulting in smaller reductions in surface water extraction. Similarly to Treatment One, for a given DD occurrence, the level of environmental flow provision is significantly greater under water caps than under DD caps, for most DD target levels (Figure 8.33). Where the objective is to generate environmental flows and significant DD reduction, a water cap could achieve this aim more efficiently.

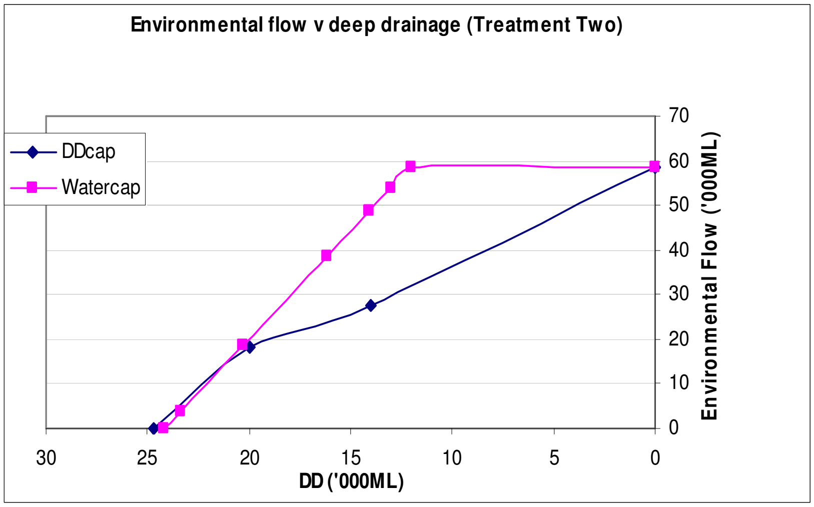

Figure 8.33: Environmental flows v deep drainage, Treatment Two.

8.6.3 Treatment Three

The TC curves under Treatment Three are relatively steeper than under previous treatments, essentially due to the increased shadow value of DD as water is shifted to its highest value use with trade (Figure 8.34).

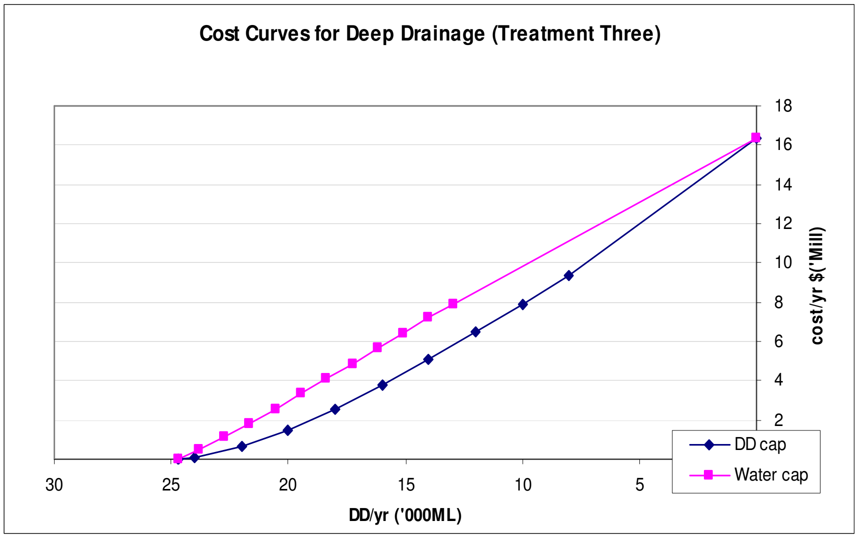

Figure 8.34: Total cost (TC) of deep drainage reduction under Treatment Three, comparing instruments.

While the overall TC is greater with trade, the discrepancy in TC under water quantity and DD instruments are not significantly different; DD caps are also more cost-effective, and can reduce the opportunity cost by approximately $2Mill, under Treatment Three. However, the most distinguishing difference is the change in water use under the two instruments (Figure 8.35).

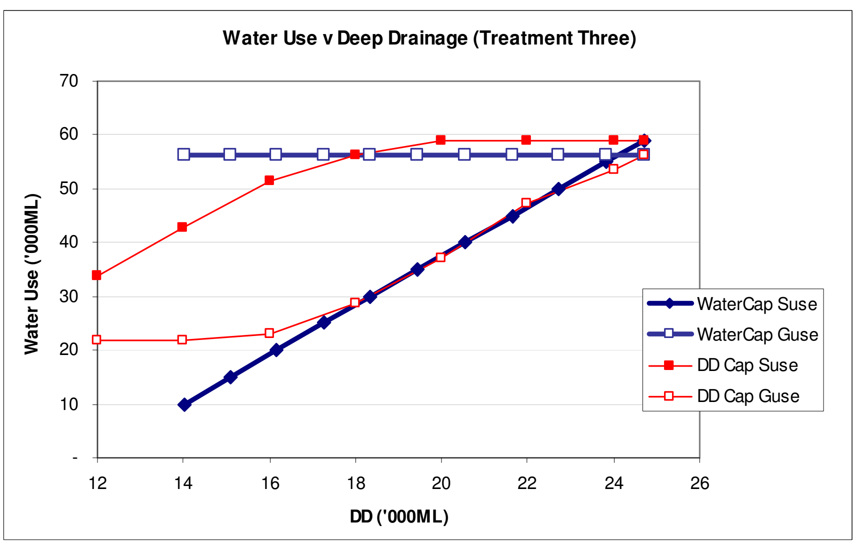

Figure 8.35: Water use under each instrument, Treatment Three.

Under DD caps, groundwater use (Guse) appears to be the main source of water that is reduced to meet the drainage constraint. This is while surface water use (Suse) is relatively unaffected until DD targets fall below 17,000ML. This means that under a DD cap, in-stream environmental flows are not increased even at very tight levels of DD constraints, because surface water extractions are not reduced. Instead, the level of groundwater use – which is set to the estimated sustainable extraction level – is reduced to a level that is far below its full capacity. The use of DD instruments therefore imposes unnecessary costs on the irrigators, while at the same time it does not generate greater environmental flow benefits. For all levels of TC incurred, the environmental flow generated is greater under water caps than under DD caps (Figure 8.36).

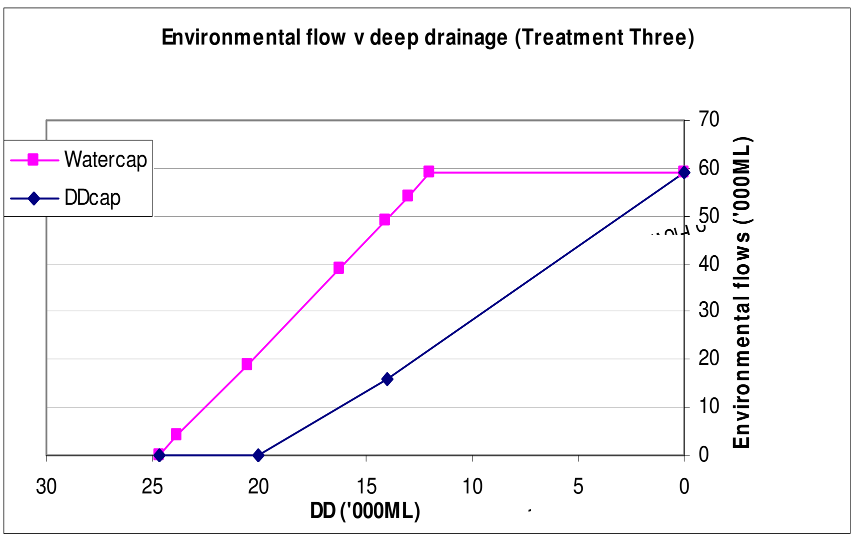

Figure 8.36: Environmental flows v deep drainage, Treatment Three.

8.6.4 Shadow Price of Deep Drainage

The shadow price of DD under each treatment is presented in Figure 8.37 (Treatment One), Figure 8.38 (Treatment Two) and Figure 8.39 (Treatment Three). The shadow values obtained represent the marginal cost of reducing DD at the catchment level. It can be seen that as AIS (Treatment Two) and water trade (Treatment Three) are introduced, the shadow value of drainage increases. Also, as the DD constraint approaches zero the shadow prices appear to increase and then plateaus out as the drainage target becomes stringent. This explains the shape of the TC functions for DD reduction observed in the above sub-sections, which increases at an increasing rate but gradually becomes linear.

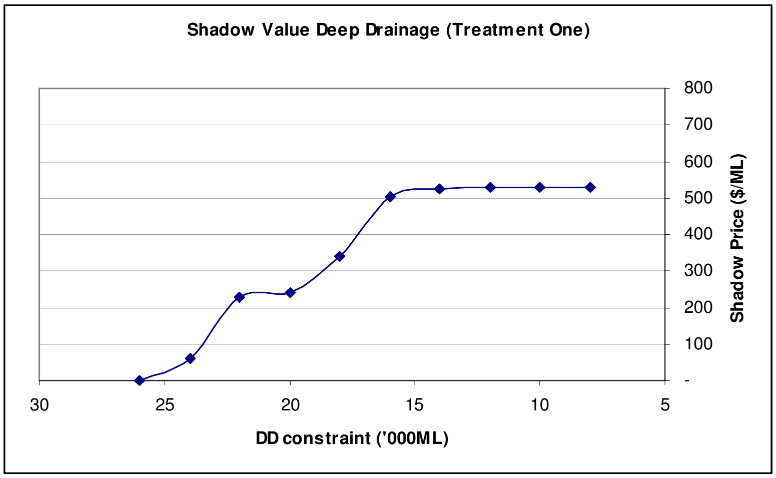

Figure 8.37: Shadow price of deep drainage under Treatment One.

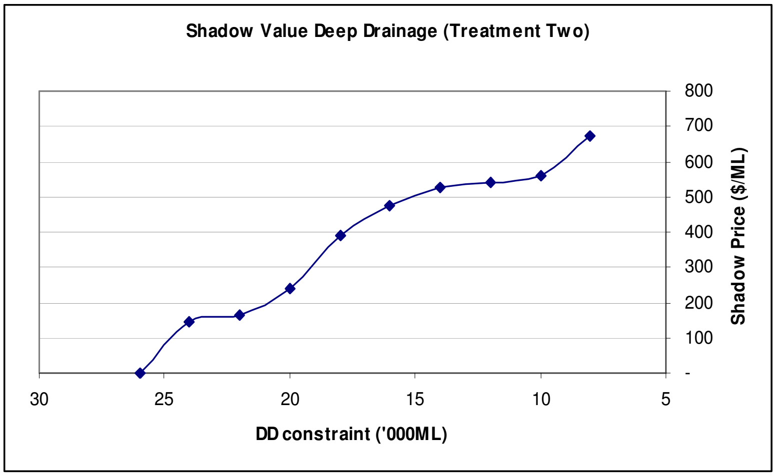

Figure 8.38: Shadow price of deep drainage under Treatment Two.

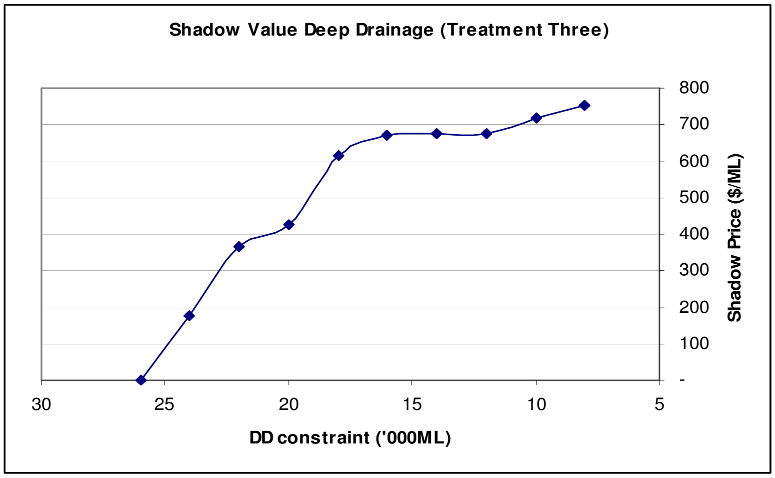

Figure 8.39: Shadow price of deep drainage under Treatment Three.

If a price-based rather than a quantity-instrument is considered for controlling drainage, these shadow prices for DD will be useful for determining appropriate DD prices. Both price and quantity instruments should theoretically achieve the same outcome, such that the results in previous sections should also hold if there is an efficient DD pricing system in place. For catchments which have chronic salinity problems and very high levels of DD, it may be optimal to use price or quantity instruments to target DD directly, since it has a greater cost-advantage over water instruments. However, the cost-advantage of DD instruments can easily be outweighed by the extra administrative cost of setting up such a system. While DD pricing would provide the correct conservation signals and achieve DD reduction, a significant knowledge gap remains regarding the actual occurrence of DD across a large landscape.

8.6.5 Summary of Separate Water and Deep Drainage Instruments

The discussion above indicates that the choice of a preferable instrument should depend on the catchment manager's objectives. If it is realised that the level of salinity contributions is in fact excessive, it may be worthwhile creating separate DD instruments to control its occurrence. Drainage instruments have the advantage of providing the least-cost means of achieving salinity targets, by forcing the most inefficient irrigators to forego wateruse, rather than causing only surface water users to relinquish allocations at higher cost. However, DD caps also have significant opportunity costs, because groundwater must also be reduced to a sub-optimal level of water use, and lead to unnecessary income losses. This is while in-river extractions are not reduced, such that less environmental flows are generated for a given DD target. In a broader context, for salinity instrument to be preferred, it would have to be insured that the benefit to downstream users from reduced water salinity must be greater than the cost incurred by upstream users in mitigating salt loads.

Alternatively, reduction of surface water allocation can reduce drainage effectively without affecting groundwater use. Given that the overall drainage occurrence and total water use is comparable under both scenarios, there is no significant difference between the effects of both instruments. Therefore, if deep drainage is not at a critical level, it may be more efficient to adopt instruments that regulate surface water use rather than deep drainage. This could provide additional benefits in the form of greater environmental flows and fresh flushes to dilute saline runoff, and avoid excessive administrative costs in creating a dual instrument. Also, given the knowledge gap regarding drainage (Vervoort 2007, pers. comm.), the economic cost for setting a 'wrong' target is likely to be high.

A factor which may affect the results relates to the technical configuration of the biophysical model. The SWAT is configured in such a way that surface water cannot be stored in on-farm storages, so that when river water is available it is pumped directly onto the field. Irrigation using surface water therefore only occurs when the opportunity arises, which is more infrequent than groundwater. Given the uncertainty of surface water supply, irrigation volume and frequency is lower, hence drainage per hectare is lower. As a result, to achieve the same drainage reduction as DD caps (which affect both surface and groundwater use), a more stringent surface water cap becomes necessary. If configurations could be made in SWAT to store surface water in on-farm storages, the security of surface water supply would increase and the frequency and volume of irrigation would also increase. Surface water users would enjoy the same security of supply as groundwater users, so that irrigation volume and DD would be very similar between the two water sources. The reduction in either water source would therefore lead to a similar fall in drainage, and translate into very similar DD cost functions between drainage and water instruments. This further strengthens the conjecture that water caps on its own would suffice in achieving the dual target of environmental flows and salinity reduction.

This concludes the comparison between water and drainage instruments, under the three treatments. For the following section, the focus moves to the results in Treatment Four, which consider the impact of increased competition for surface water from an agent that is external to the Mooki basin.

8.7 EXTERNAL WATER TRADING

The impact of an external agent entering the regional water market is assessed in this section. It is assumed that a mine begins operations in the Mooki region and enters the water market to compete for water at various (exogenous) market prices. For a given water price, P_w, irrigators could choose to trade internally to other irrigators who will use water for crop production, or externally to the coal mine. It is assumed that only surface water could be traded. For simplicity, the gain from external water trade is calculated in terms of net benefit to the external agent, based on its derived demand for water. The water price is parameterised in the interval from zero to $160/ML and irrigators can only profit from the quantity that is traded at the given price, both internally and externally.

The coal mine's derived demand for water is assumed to be $W^* = 12{,}522 - P_w/15.48$, based on the quadratic production function of $Coal\text{=-}0.1721W^2 + 4318.2W$ and coal price of $P_y = \$45$ per ton (ABARE 2006). This was obtained through econometric estimation using data from a large Australian coal mining company (see Section 7.3.5). The impact of an external buyer is compared to where only internal water trade is possible, to evaluate the degree of competition for resources. It is assumed that the annual allocations are 59,000ML for every year in the planning period. The results reported are therefore based on the outcome for one year, which is representative of all seasons.

8.7.1 Treatment Four – Base Case

Under the Base Case scenario of Treatment Four, the price of water is assumed to be zero and water trade between the HRUs in the catchment is costless. The solution essentially represents the theoretical optimal water allocation for maximum basin profit given annual surface water allocations of 59,000ML. There are no additional requirements for environmental water flow except for the stipulated level in the surface Water Sharing Plan. The volume of water sold ("internal sell") exactly equals the volume of water demanded ("internal buy"), of 37,500ML (Table 8-10). The profit from cropping alone is close around $40Mill where the market price for water is zero, such that irrigators can trade water at zero cost.

Table 8-10: Outcome under Scenario 4.1.

Pw ($/ML)	Profit/yr ($Mill)	Internal buy ('000 ML)	Internal sell ('000 ML)
0	40.15	37.5	37.5

8.7.2 Treatment Four – Internal Trade Only

Where there is only internal trade, the volume of water traded remains the same until P_w increases to above $115/ML (Table 8-11). Above this price, the market supply begins to diverge from demand.

Table 8-11: Internal trading only.

Pw ($/ML)	Profit crop only ($Mill)	Internal buy ('000 ML)	Internal sell ('000 ML)
55	38.09	37.5	37.5
70	37.52	37.5	37.5
85	36.96	37.5	37.5
100	36.40	37.5	37.5
115	35.60	37.5	39.6
130	34.38	37.2	45.0
145	33.69	34.7	46.1
160	32.48	19.1	51.8

This reflects the shadow price of water shown in Section 8.3.4, of $111.45/ML, which represents the market clearing price of water. At higher prices, there is a net inflow of water into the market, leading to excess supply. The internal demand for water remains relatively inelastic and does not begin to fall until P_w increases above $145/ML. An interesting observation is that the market-clearing price of around $111.45/ML is similar to the current temporary trade value for the Lower Namoi regulated systems of $100-$120/ML (WaterExchange 2007). There are no head-dams in the Mooki unregulated system, such that there is no risk of stranded assets when trade is opened between the two regions.

8.7.3 *Treatment Four – Internal and External Trade*

In this scenario, an external agent (coalmine) enters the regional water market and competes for water resources. The results are presented in Table 8-12.

Table 8-12: Internal and external trading.

Pw ($/ML)	Profit crop only ($Mill)	Internal buy ('000 ML)	External buy ('000 ML)	Internal sell ('000 ML)	External net benefit ($Mill)
55	38.09	37.5	-	37.5	-
70	36.32	34.9	11.1	46.0	953
85	35.68	33.6	12.5	46.1	1,213
100	35.17	33.6	12.5	46.1	1,213
115	34.67	33.6	12.5	46.1	1,212
130	34.17	33.6	12.5	46.1	1,212
145	33.68	31.8	12.5	46.1	1,212
160	32.48	19.1	12.5	51.8	1,212

For water prices below $55/ML, no water is sold to the coal mine. This suggests that water has a greater value in production below this market price. However, between the price range of *$70/ML<* P_w*<$130/ML,* there is increased competition for water between internal and external users. A volume of 3,600ML of water that was traded internally is instead sold to the external user. Furthermore, the presence of an external buyer presents a more profitable alternative to cropping, such that a further 8,600ML of water becomes sold in the water market to meet external demand. Above a price of $160/ML, there is excess supply in the market and the external agent merely soaks up some of the excess supply without infringing on internal demand, placing a market value to the excess water which has no value internally.

From the above results, it can be concluded that in order to compete with internal users when the price is below $70/ML, the external buyer needs to pay a premium to meet its full demand, since internal users have a high value for water at this market price. However, offering a price above $85/ML would allow its full demand to be met without the need to pay a premium. This is because irrigators find it more profitable to sell their allocations than to use it for crop production, and market supply increases relative to where only internal trading exists. For water prices above $160/ML, there is excess supply and there is no competition with internal users; the limiting factor then becomes the external user's demand. However, the maximum water demanded by the external user across the price range considered is only 12,500ML/yr, or 21% of total basin water supply. This suggests that, while external competition for water will affect some irrigators, it should not pose a significant competition for the regional irrigation sector. In fact, it may be profitable for some producers to sell allocations at a premium to the coalmine, without compromising the integrity of agricultural production in the region. For example, selling some water at $85/ML would lead to a 2.7% drop in annual profit from cropping ($1Mill), while representing a net benefit of $1billion for the mining industry in the region[11]. This is also considering the sizeable area in the catchment that appears to be more suited to dryland cropping.

8.7.4 Summary of External Water Trade

While there have been some concern of the impact of increased competition for water on the regional irrigation sector, it appears that the overall impact of a coal mine in the Mooki basin will be relatively small. The maximum demand for water by the external user only comprises a fairly small portion of what is available, and leads to a reduction in the annual profit from cropping of 2.7%. This is while the net benefit that the water represents for the coal mine exceeds $1billion. These results indicate that the Namoi region would accrue net gains from the coalmine. However, these results are dependent on the assumptions made regarding the coal mine's demand for water. While the derived demand was based on empirical data from the coalmining industry, the actual volume of water required depends on the size of the coal reserve in Gunnedah, which is currently unknown. This conclusion also does not consider the secondary effects on employment and other industries, nor the environmental externalities created from mining operations. The results of this analysis may vary once these external effects are factored into the costs of coal production. Nevertheless, the direct impact of increased competition for water from external users, based on the assumptions of the model, is not likely to jeopardise the regional irrigation industry.

[11] The net benefit is calculated based on the area under the coalmine's demand function for water.

8.8 SPATIALLY DELINEATED RESULTS

In previous chapters, it was highlighted that the advantage of using a GIS-based biophysical model, was to provide spatially differentiated results. This is possible using the SWAT model, which delineates the Mooki basin into HRUs as unique combination of soil type and landuse. The location of each HRU in the Mooki is presented in Figure 8.40, which was produced using the GIS program linked to SWAT. For each HRU, the optimal landuse, activity, resource use, and shadow values have been determined through the economic optimisation model. The solutions obtained for each scenario under each treatment could be implemented on ground since each HRU could be identified using this graphical representation of the basin. Although due to technical limitations GIS layers for each simulation outcome were not produced, the solutions from the economic optimisation are included in Appendix E and could be used to create layers to superimpose on the figure below (Figure 8.40). This can be useful for policy analysis, since the desired outcomes of various policies and its impacts could be assessed with a high degree of spatial detail.

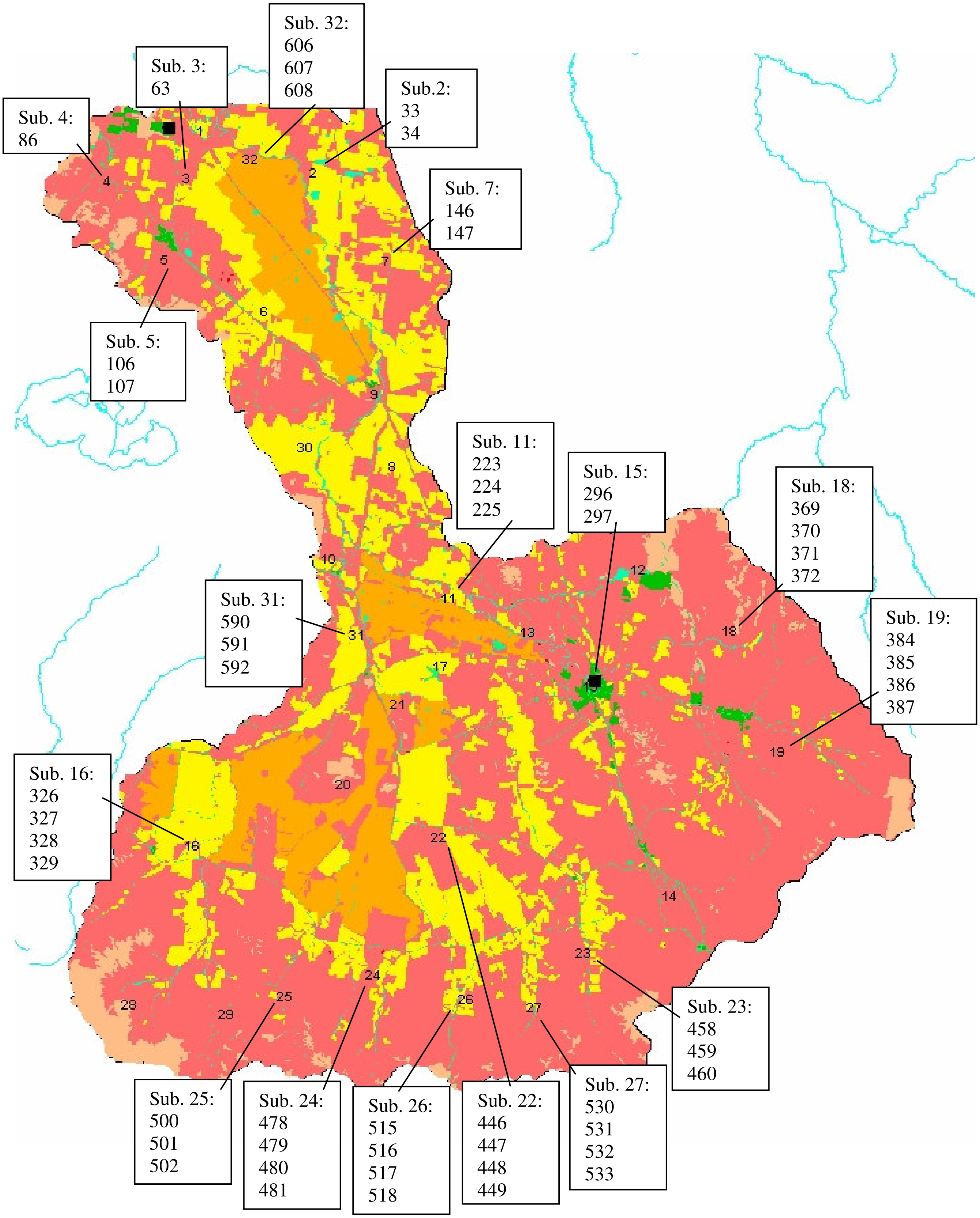

Figure 8.40: Location of the HRUs delineated in SWAT.

8.9 SUMMARY OF CHAPTER 8

Simulation of a water allocation model of the Mooki basin was developed, using an interdisciplinary approach to analyse the impact of various environmental policies on a spatially explicit scale. The objective was to determine how environmental targets pertaining to environmental flow provision and salinity reduction could be achieved at the least-cost. This involved an examination of the changes in resource use and production activities in each irrigation area, under each policy instrument and constraint. Additionally, the economic impact of a dual-instrument, and the use of separate instruments to meet environmental objectives were assessed. This was conducted in order to shed light on the usefulness of current basin policies such as Water Sharing Plans and end-of-valley salinity targets in achieving environmental goals, and the associated economic trade-offs. Furthermore, the consequences of increased competition for water from agents external to the irrigation district were analysed, to evaluate the potential impact on the stability of the irrigation industry. The value that water efficient technologies and water trade represents for the case study basin was evaluated under each policy setting, by comparing the outcomes to the status quo. In addition, consideration was given to the shadow values associated with water resources and the conjoint pollution, in the form of deep drainage that induces salinity, to explain the outcomes under each scenario. A general assessment of the inter-temporal nature of groundwater use, given stochastic surface water supplies, was also discussed at the beginning of the chapter.

From the results obtained, the findings and suggestions are as follows:

1. Overall, it was found that the use of alternative irrigation systems (AIS) and the presence of a water market would improve the economic profit generated by irrigators in the Mooki basin. Under the present surface and ground Water Sharing Plan stipulations of water allocation and environmental flow requirements, AIS on its own could reduce the occurrence of deep drainage (DD) by 5% and an increase in annual profit by 4%, relative to the status quo situation where only furrow irrigation is used. Of the two irrigation technologies

considered, pivot and drip, it appears that pivot systems provide the greatest benefit to irrigators, while drip systems are not as economically viable as pivot due to its high capital outlay. In the presence of a water market, overall catchment profit could increases by 10% and DD could fall by 3%, relative to the status quo. This is due to the increase in surface water use due to the mobility of water allocations.

2. Groundwater carry-over rules are useful where surface water is stochastic since groundwater allocations can be banked or borrowed, subject to its two-year and three-year extraction limits. However, where surface water is deterministic and constant, there is little need for banking or borrowing since the optimal rate of groundwater use should be at its annual sustainable extraction level. For an ephemeral river system like Mooki, where surface water is stochastic, groundwater carry-over rules are useful to hedge against uncertainty. However, the optimal strategy is to utilise annual allocations almost fully each year due to the limits on carry-over allowances.
3. The early system of area-based irrigation licensing has led to an inefficient allocation since it does not match water to its highest value use. This is observed through the results under each treatment, which shows that under the status quo surface water is used in the upstream irrigation areas, Caroona and Breeza, whereas its value is highest in downstream Ruvigne. It is optimal to create a fully functional water market in the Mooki region to encourage the trade of surface water downstream, since upstream regions have high conservation value while water has higher productive value downstream.
4. Where surface water is reallocated from extractive use towards environmental use, the overall opportunity cost is reduced significantly where a water market is in place. The use of AIS also reduces the economic cost incurred, since irrigation water is used more efficiently. However, it is not as effective in mitigating the costs incurred by irrigators compared to where water market is in place. Nevertheless, where environmental flow requirements are high and much of the extractive flows are redirected towards environmental purposes, the ability for AIS or water trading to alleviate the economic burden is limited. The benefit of

additional environmental flows must justify the economic cost incurred by the regional irrigation industry.

5. Considering the high level of estimated salt load from the Mooki basin, there is a cause for concern over its contribution to the salinity level of the downstream Barwon-Darling system. It appears that, where a dual-instrument is implemented to target environmental flows and salinity reduction, there is no significant difference in the marginal cost of meeting these targets at stringent levels. It appears that dual-instruments may be simultaneously set without excessive costs. However, it also suggests that one instrument is sufficient to achieve the dual-objective of salinity reduction and environmental flow provision.
6. Where water caps and DD caps are imposed separately, each instrument can generate some environmental flows and drainage reduction since the level of surface water use is reduced. However, the most distinctive difference is the source of water that is affected by the instrument. Water caps only impinge upon surface water use, while drainage caps lead to reductions in both surface and groundwater use. Each instrument has its advantage: the general trend observed under all treatments show that water caps generate greater environmental flows and maintains groundwater use at its sustainable rate of extraction. On the other hand, drainage caps can achieve DD reductions at the least-cost, but causes groundwater use to drop to levels below sustainable levels and produce lower environmental flows. In the case of the Mooki basin, water caps on their own can achieve environmental flow provisions and salinity reduction more effectively.
7. Irrigators in the Mooki basin have expressed concern over the impact of increased competition for water from a potential coalmine in the region. However, the economic impact from the water competed away for coal production is around 2.7% per year. Compared to an increase in value of $1billion from coalmining over the same period, there is a substantial net gain from the new mine for the irrigation district. This has not factored in other flow-on effects such as employment or environmental damage.

Chapter 9.

CONCLUSION, DISCUSSION AND SUGGESTIONS FOR FURTHER RESEARCH

In this study, the aim was to develop an economic modelling framework that can help assess the economic impact and effectiveness of various environmentally-oriented water policies. The model was developed based on the case study of the Mooki basin in the Namoi Valley, but can also be transferred to other catchments to examine the impact of various policies on a catchment-specific level. The empirical results were used to determine the cost of various environmental targets and also the least-cost means of achieving these objectives. In this chapter, a synopsis of the thesis provided, followed by some conclusions and recommendations drawing from the results of the empirical study, and some limitations which leads to a discussion of future work. The conclusions and discussion is based on the empirical case study of the Mooki, which derive from the review of water policies affecting the basin, the available data and literature that formed the foundations of the economic modelling, and the results of the analysis.

9.1 GENERAL OVERVIEW OF THE THESIS

At the start of this thesis, the main objective was to provide an overview of the water economy in Australia. Much attention was given to exploring the problems in water allocation that has arisen in Australia due to its geographical characteristics, and from a policy perspective. At the crux of the issue is level of river diversions in the Murray-Darling Basin, which has long been at an unsustainable rate. This was driven by earlier policies inherited from the ‘expansionary phase’ of the Australian water economy, which has created a situation of water over-allocation. As the water economy approached a ‘mature phase’, the focus fell on demand-side management. Furthermore, the capital and operational costs of water infrastructures have not been recouped. This formed the impetus for the cost-recovery process, the purpose of which was to incorporate the true cost of water services in the water price. This was but one of the myriad of Government water policies initiated since the early 1990s, in recognition of the need for a sustainable approach to water management. This includes the cornerstone agreement in 1994, the COAG Water Reform Framework that led to a succession of intergovernmental water agreements. A discussion of the various arrangements was provided in this chapter, including the Murray-Darling-Basin Agreement, the National Water Initiative, Living Murray Initiative, and the recently proposed Commonwealth National Plan for Water Security (although part of this National Plan was to replace the intergovernmental arrangements under the MDB Agreement with one Federal Government minister). These policies have the objective of reallocating water more efficiently amongst competing uses, through the use of market-based instruments which has been widely adopted in most Australian catchments (e.g. the creation of water markets). Other policies to mitigate salinity were also implemented through Murray-Darling Basin Commission initiatives, including end-of-valley salinity targets. However, the effect of various environmentally-oriented policies on the economic performance of the affected industries is uncertain, and needs to be understood. This forms the impetus for this thesis.

A review of water economics literature was then provided, to highlight the different aspects of water management and the difficulties involved. Firstly, a review of studies using a GIS-integrated framework was conducted. This was to emphasise the advantage of creating an integrated management approach, which would enable better allocation of scarce water resources. This was then followed by an evaluation of price-based and quantity-based instruments that have been promoted in natural resource management. Other complexities in water management relating to groundwater hydrology, particularly return flows and deep drainage, was also explored. These earlier chapters set the foundation for the remainder of the thesis, which focused on creating a framework for the efficient management of water on a basin-scale, in light of the complexities and current water resource policies discussed thus far.

The selected modelling approach was a combined linear programming (LP) and dynamic programming (DP) framework, due to the inter-temporal nature of groundwater use and the static nature of production decisions. The use of market-based instruments for water and deep drainage to achieve such outcomes was also discussed. The theoretical solution obtained using the LP and DP framework was then analysed, to illustrate how the efficiency criteria are met through the chosen economic modelling framework. The use of the model was focused on the impact of various environmental targets, in particular environmental flows and salinity reduction, on the regional irrigation industry. Following from this objective was an evaluation of the effectiveness of market-based instruments to achieve efficient outcomes, and the consequence of increased competition for water from external users to the catchment. Characteristics specific to the case study, the Mooki basin in the Namoi Valley, was analysed in order to customise the modelling assumptions. These include specific government policies which affect the Mooki, in relation to the institutional arrangements discussed in earlier chapters. These assumptions were then used to create the specific model in this thesis. A summary of the findings of this empirical study is given below.

9.2 SUMMARY OF THE SIMULATION RESULTS AND ANALYSIS

The following summary is based on the results of the simulation scenarios under each treatment considered in this study. In general, it is indicated that the water resources in the basin are scarce and that earlier policies on water distribution have translated into an inefficient allocation of water in the basin. There is scope for a more efficient management of water in the Mooki basin, as indicated by the discrepancy in resource allocation and production between the status quo and alternative treatments, which were conjectured to improve allocative efficiency.

9.2.1 Summary Statistics

Of the irrigation technologies considered (furrow, pivot and drip), on average, pivot systems generate the highest returns and represent the most profitable investment for the Mooki basin. Pivot systems can achieve a lower irrigation rate, lower deep drainage occurrence, and higher yield per hectare irrigated. The yield increase offsets the capital investment required. In contrast, drip systems are the least cost-effective, although they achieve the highest water savings and deep drainage reduction. This is because the yield increase is insufficient to cover the capital outlay, which is double the cost of pivot systems.

For any irrigation system, groundwater consistently produces better yields and profit compared to its surface water counterpart. This is due to the reliability of its supply, which ensures the full irrigation is delivered to the crop at the scheduled time of irrigation. Groundwater could therefore produce relatively higher profits, despite higher costs of pumping. While yield and profit is higher compared to irrigation with surface water, the water use and drainage is also higher since more water can be applied.

Where water efficient technologies are used, the productivity of water resources is increased. This is indicated by the increase in annual profit by 4% and a fall in deep drainage by 5%, where pivot and drip irrigation systems are used. When water markets exist, the annual profit increases by 9% and deep drainage falls by 3%. This is due to the mobility of surface water allocations, so that the productivity of water is significantly improved.

9.2.2 Inter-temporal Resource Use

The use of groundwater banking and borrowing through the carry-over rules is dependent on the certainty of surface water supply. When it is deterministic, and constant, groundwater use is close to annual allocations without any banking or borrowing. However, when surface water supply is stochastic, the rate of groundwater extraction moves in the opposite direction of annual surface water supply. This was an expected result, since groundwater banking is beneficial where it is used to compensate for the shortfall in the alternative water supply.

9.2.3 Production Activity Changes

The initial allocation of water according to irrigation area leads to an inefficient outcome, whereby water supply is not used where it has the highest value. As a result, where water is inefficiently allocated, irrigators in these upstream areas of the Mooki (Caroona and Breeza) are the most affected by water caps and the overall opportunity costs incurred is also inflated. Where water trade is possible, almost all surface water is traded downstream for irrigated production in Ruvigne. As water is reallocated to its highest value use, introducing a water cap in the presence of water trade allows environmental flows to be sourced at the least opportunity cost. This leads to an efficient outcome, whereby irrigated production in the upstream irrigation areas which have high conservation value is reduced, while the basin profit is increased from greater production in downstream Ruvigne. Where DD caps are imposed to mitigate salinity contribution, under all treatments, much of the impact occurs in Ruvigne where groundwater is used as the primary water source. Drainage caps thus cause groundwater extraction to be reduced below its sustainable level, which leads to a sub-optimal outcome because the full capacity of groundwater resources is not used.

9.2.4 Application of Environmental Flow Policies

The economic impact of environmental flows can be minimised simply by encouraging the use of water efficient technologies or by encouraging water trade. Where a water market exists, the opportunity cost is reduced by around $3.3Mill per year or around 9%. Using water efficient technologies could also reduce the economic impact of environmental flows, by around $1.5Mill per year or 4%. However, at stringent environmental flow targets the ability for alternative irrigation systems and water trade to mitigate the economic cost is reduced, since these mechanisms can only do so much to keep the opportunity costs low. It was also found that the additional cost to the catchment manager to source an extra unit of environmental flow is relatively constant, as indicated by the constant marginal cost of foregoing surface water. This is because of the assumption that irrigators reduce the area under irrigation rather than the rate of irrigation, such that the marginal value of water is reflected in the area of cotton it can produce.

9.2.5 Application of Deep Drainage Policies

The contribution of salt load from the Mooki to the end-of-valley target for the Namoi Valley is 7%. While this figure is relatively low, proportional to the area of the Mooki the salinity contribution should only be 1%. This is despite the lack of incentive for irrigators to internalise the impact of salinity into their production decisions, since salinity-inflicted productivity loss within the Namoi is minimal. A cap on deep drainage could be used to reduce salinity contribution to downstream river systems, and it is expected that a drainage cap could be imposed simultaneously with a water cap without much increase in the costs borne by irrigators. However, it also appears that a single instrument is sufficient to provide conservation signals with regards to water use and thereby a reduction in deep drainage. For the Mooki case study, it seems that a cap on surface water extractions is more appropriate than drainage caps, since for the same opportunity cost a greater provision of environmental flows and some salinity reduction is achieved. Also, groundwater use is maintained at its sustainable level. On the other hand, deep drainage caps have a cost-advantage over water caps, since it can achieve salinity reduction at the least-cost by forcing the least efficient irrigators to forego water use, regardless of the water source. However, since both surface and groundwater users are affected, the level of groundwater use could be reduced to a sub-optimal level whereby the full capacity of the sustainable recharge is not exploited. This is while in-stream water extractions are not reduced as significantly as water caps, thereby generating relatively less environmental flows.

9.2.6 Competition from an External Water Consumer

Increased competition for water resources from an agent external to the regional irrigation sector will cause the annual profit from cropping to fall by 2.7%. This is while the water competed away by the external agent, in the form of a coal mine, generate a value of $1billion. Competition between the internal and external users occurs across water prices of $70-145/ML, with the external user demanding at most 12,500ML. Of this, 3,600ML is directly competed away from internal buyers, while 8,600ML of water that was used for cropping where there was only internal trade, is instead sold to the external buyer. Above $160/ML, there is excess supply and the external agent merely adds value to the excess water that has not value internally.

9.3 CONCLUSIONS AND DISCUSSION

Conclusions of the thesis are drawn from previous chapters are as follows:

1. It is worthwhile for irrigators in the Mooki to invest in water efficient irrigation technologies and to participate in water trading, as a way of mitigating the impact of increasingly stringent environmental policies. However, if the environmental targets are excessively stringent these adjustment mechanisms could only do so much to reduce the economic burden. While environmental protection is important, it should not be set too stringently even if water efficient technologies are utilised and water trading is in place.
2. The groundwater carry-over rules are beneficial for irrigators in the Mooki, considering the ephemeral nature of the surface water supply. Where supplies are stochastic, it is optimal to hedge some groundwater allocations from year to year. Given the in-river supplies for the Mooki are known to be extremely variable, carry-over rules are particularly important to ameliorate the economic burden of surface water scarcity.
3. Water trade within Upper Namoi (UN) will result in surface water being traded downstream to Ruvigne, which was found to be the most productive irrigation area in Mooki. Encouraging water trade in UN would therefore lead to an optimal outcome where water is traded away from upstream irrigation areas of Breeza and Caroona, which can be left for environmental conservation. Considering the shadow price of surface water in UN is almost at par with the market in regulated Lower Namoi (LN), there is the potential for trade between these systems, which may circumvent the ‘thin market’ situation that is often cited as a problem in Australian water markets. However, if groundwater entitlements for the Mooki are further reduced below Water Sharing Plan stipulations, then the shadow value of surface water in this area is likely to increase. Under such circumstances, opening water trade between UN and LN may lead to water being traded to UN. This would conflict with the catchment authorities’ objective to protect the environment in UN where there is high conservation value. Furthermore, due to the salinity of the aquifer system in Namoi, increased return flows as a result of

water being traded upstream may accentuate the salinity problem by compounding the amount of salt load carried downstream. Given these implications, and also considering the significant cuts in entitlement already imposed, it is appropriate that groundwater entitlements are not further reduced.

4. For most irrigators in the catchment, pivot and drip irrigation systems are shown to be a worthwhile investment even where the full cost of the investment is borne by the irrigators. This implies that there is no need for subsidisation of these technologies as there is sufficient incentive for efficient irrigators to make the investment. If the cost of water efficient irrigation systems is subsidised, as intended in the Commonwealth Plan for Water Security, it would inflate the value of irrigation enterprises in UN and distort the allocation of water, precluding an efficient distribution through water trade. Also it will increase the structural adjustment required to retire irrigation areas that are inefficient. The priority should be increasing the security of water supply to persuade irrigators to adopt irrigation technologies that require large capital expenditure. Producers need the assurance that their investment can be recouped in the long run.
5. Salinity may not be a problem within the Namoi, but the downstream impact on the Barwon-Darling catchment should be taken into account when making production decisions upstream for a socially optimal outcome to be achieved. However, drainage capping may not be the best instrument to aid in this objective. Without the opportunity for trade, capping resource use leads to reductions only at a farm level without necessarily increasing overall basin efficiency. Even with trade, the overall effects on water use and drainage are very similar regardless the instrument used to control resource use; hence there is little incentive to create separate instruments. While drainage caps have a cost advantage over water caps, by allowing only the most inefficient irrigators to sacrifice water, there are significant disadvantages in terms of information and administration expenses, which can outweigh any perceived benefits. Furthermore, its impact on groundwater use, which is scheduled to be reduced considerably in many zones, will lead to a sub-optimal outcome. In this sense, the author concludes that a water market on its own may provide the best, if not optimal, means of

contending with conjunctive pollution, it the form of deep drainage associated with irrigation.

6. Although a ‘sustainable’ rate of groundwater extraction has been estimated in the groundwater Water Sharing Plan, the existence of such a sustainable use is debatable. Since the groundwater systems are in a state of equilibrium that is dependent on the recharge and discharge rates, when water is extracted from the confined aquifer, the recharge and discharge rates shift to a new equilibrium. This means that the recharge and discharge rates will not remain the same; recharge would increase while discharge would decrease due to the draw-down effect as water is pumped. The implication of this is that, users both upstream and downstream of the point of extraction will experience a reduction in groundwater resources. The ‘sustainability’ of groundwater use then becomes a question of the acceptable economic trade-off between upstream-downstream uses. In this thesis, it was assumed that the annual groundwater extractions stipulated in the groundwater Water Sharing Plan could be sustained indefinitely. However, as a future study, the trade-off between groundwater use in the Mooki and neighbouring irrigation areas sourcing the same aquifer system could be established. This is needed to be able to determine the opportunity cost of the groundwater resource to irrigators in other irrigation districts sharing the same hydrological system.
7. The entry of an external water user in the form of a coalmine should not pose a significant threat to the regional irrigation industry. This can be concluded considering the relatively minor change in overall benefits from cropping and the small volume of water demanded by the coalmine. Based on the assumptions of the derived water demand, the water competed away for coalmining represents significantly greater value to the mining industry relative to cropping. This is evidenced by the increase in water sold into the water market in the presence of an external buyer, which provides a more profitable avenue for water users in the Mooki. Irrigators could therefore stand to gain from the introduction of coalmining in the region by selling some of the basin surface water supplies, and at the same time maintain the integrity of the irrigation industry.

8. A GIS-linked economic optimisation model could be used to provide greater transparency and reduce the information cost of accurate estimations of resource use across a large landscape. From the information provided by the integrated modelling approach, specific HRUs may be identified, for example, for buying-back entitlements to provide for environmental flows or to reduce salinity contribution. It is a useful tool that can be adopted cost-effectively. A national initiative is already in place to improve access to, and availability of spatial information in Australia (Geoscience Australia 2007), which reduces the set-up cost of an integrated system of resource management. Other than the requirement for trained technical staff to operate GIS programs and develop GIS-layers, the fixed and variable costs of setting up integrated studies can be relatively low. This can then become a low-cost option to make transparent water information and management, in line with the National Plan for Water Security objective.
9. More research in the field of groundwater hydrology is required, as it remains a field for which there is limited understanding, and that has significant implications for the management of water resources. It is particularly important that accurate assessment of feedback-mechanisms relating to return flows are established on a catchment-specific basis, since the role of return flows vary from basin to basin. In the case of the Mooki, it is expected that return flows will have greater negative than positive externalities and should be minimised, due to the salinity of its shallow aquifer. This may not be the same for other catchments, which may have fresher return flows that may improve groundwater quality and contribute to water supply. The net impact of altering the hydrological interrelationships should be carefully assessed on a case-specific basis prior to the implementation of policies affecting water use.

9.4 LIMITATIONS

There are several limitations of this study that need to be acknowledged.

1) It is important to note that the accuracy of the economic analysis is dependent on the accuracy of the GIS data, and the assumptions made with respect to the biophysical parameters in SWAT. The assumption for parameter values, e.g.

percolation and soil conductivity or crop growth, will have implications for the policy outcomes. Much effort has been put into ensuring the parameter values are as accurate as possible, however it is important that results are further verified through ground-truthing to confirm the findings on-ground.

2) The assumption made with respect to the initial allocation of water resources has significant implications for the relative outcome under different treatments. A starting point that is close to an efficient water allocation will underestimate the value of water trade and water efficient technologies. On the other hand, an initial allocation that is very far from an efficient water allocation will overestimate the value of trade and alternative technologies. Again, much attention was given to the accurate distribution of water allocations according to actual use, but due to the fact that this information is private and that the irrigators are reluctant to disclose it only inferences on individual water extractions could be drawn.
3) A limited number of production activities were considered for the purposes of this study. Alternative irrigation activities may occur in the Mooki basin, such as irrigated wheat or sorghum, which have not been analysed. The focus was the economic impact on the irrigated cotton industry, so the scope of the analysis was confined to one irrigated crop.
4) In this study, the research was limited to a cost-side analysis. Ideally, the social benefit accrued from various environmental flow and deep drainage targets could be incorporated to determine the distribution of water between extractive and non-extractive uses that maximises social benefit. This would have required that expensive and time consuming methods of non-market valuation be carried out to gauge the willingness to pay by the public for various environmental targets. This was beyond the scope of the thesis.
5) Technological limitations precluded the incorporation of return flows on downstream water supply. While this was not a significant issue for the Mooki, since return flows were considered detrimental to water quality, it constrains the transferability of the modelling framework to other basins where return flows have significant positive externalities. A suggested framework to incorporate the feedback mechanism from return flows is provided in Appendix D, and with

access to technical expertise and computer resources this robust model could be created.

6) Some limitations of the modelling process are related to the functioning of the SWAT model:

   a) Changes in water quality are not included in the SWAT model. While factors relating to the hydrological movement of water in a basin are well captured, the relationship between water use and water quality are not simulated. There are intricacies in the soil hydrology which might suggest a non-linear relationship between deep drainage and groundwater salinity. In this sense, deep drainage may only need to be reduced slightly to achieve a large drop in salinity, or conversely a greater reduction in drainage may be required to achieve a small drop in salinity.

   b) Linearity was assumed for the demand functions and input requirements per hectare. This assumption could be relaxed by allowing for non-linear input functions. However, the functioning of the SWAT model is such that relationships between inputs and outputs are determined using physically based equations rather than regression equations of crop growth. This complicates the process of determining the marginal rate of substitution between various inputs e.g. labour and capital.

   c) On-farm storages to pump and contain passing river flows were not modelled in SWAT. As a result, surface water use was more irregular since simulations of water were only applied to the field if there is water passing at the scheduled irrigation event. If on-farm storages are included in the modelling, surface water would become readily available and the irrigation frequency is expected to become similar to where groundwater is used.

9.5 SUGGESTIONS FOR FURTHER RESEARCH

1. Due to time and computational constraints, it was not possible to produce GIS layers of the solutions. The next step for this thesis would be to integrate a GIS program into the economic model, to extrapolate the optimisation solutions graphically and seamlessly.

2. The interregional competition model used in this thesis could be expanded into a spatial equilibrium model, and include the Lower Namoi to determine the value of inter-regional trade between these areas. Groundwater trading between users in the Mooki and neighbouring irrigation areas, sourcing the same aquifer system, could also be simulated. This is to determine the opportunity cost of the groundwater resource to irrigators in other irrigation districts sharing the same hydrological system.
3. Alternative deep drainage reduction mechanisms such as growing perennial crops or salt interception plants could be included as alternatives to meet drainage targets.

While efficient water management is crucial in the current state of Australia's water economy, it is equally important that policies implemented with the aim of improving the distribution of water result in net benefits to society. From the results of this thesis, it appears that there is significant scope for improving the water use efficiency in the Mooki basin. This is also likely to be the case for many catchments in the Murray-Darling Basin, for which the situation of oversupply and inefficient use has long been highlighted. In this thesis, a spatially-explicit, integrated economic modelling framework for catchment management has been developed for the purposes of improving the allocative efficiency of water. This allows for catchment policies to be specifically designed for particular irrigation areas, rather than imposing blanket policies that may impose unjustified costs on irrigators.

While there are a number of economic studies which attempt to capture the biophysical component in water management, the use of a GIS-based model in this thesis means there is the advantage of a higher degree of accuracy and spatial applicability. The results from this analysis are directly applicable to the case study basin, and at a high level of spatial detail. The modelling framework is flexible enough to be transferred to other catchments given data availability, and could be used to examine the influence and effectiveness of environmentally-driven catchment policies.

This thesis is an attempt to improve the effectiveness of natural resource management through an interdisciplinary framework, which utilises the strength of GIS in economic analysis. The results demonstrate how advances in computational technology can be exploited to enhance existing economic modelling of natural resource problems, allowing a more in-depth understanding of these issues on a case specific level. The ultimate contribution of this research is to help improve the accessibility and reliability of information available at the fingertips of decision-makers, and help guide future policy directions towards a socially desirable outcome.

APPENDIX

Appendix A: Groundwater Hydrology

The definition of groundwater is water in the saturated zone of earth materials under pressures greater than that of the atmosphere (Neitsch et al. 2001 p. 159). The groundwater contains regions of high conductivity, made up of coarse-grained particles that allow water to move easily, and regions of low conductivity, made up of fine-grained particles that restrict water movement. There are confined and unconfined aquifers, which are defined as "a geologic unit that can store enough water and transmit it at a rate fast enough to be hydrologically significant" (Dingman 1994 in Neitsch et al. 2001). The following figure illustrates the two types of aquifers.

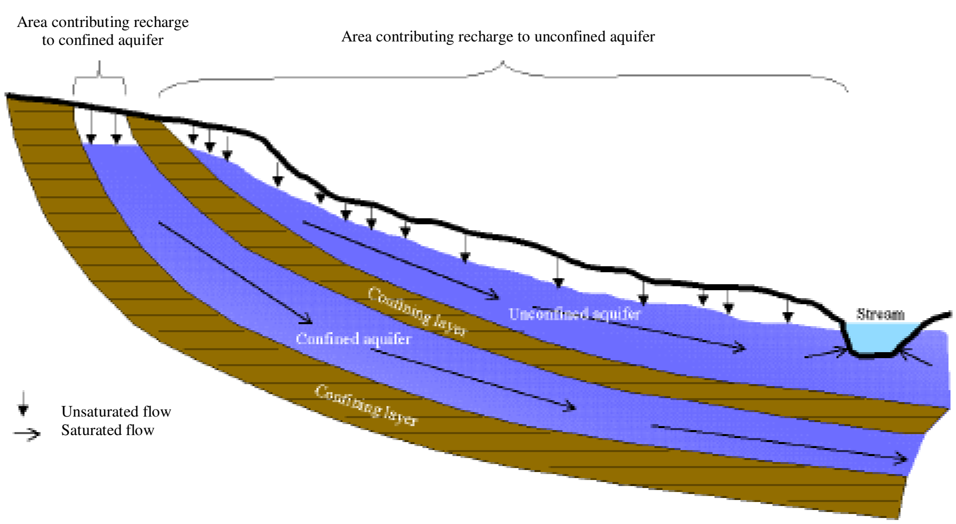

Figure A 1: Unconfined and confined aquifers (source: Dingman 1994 in Neitsch et al. 2001 p. 160).

Recharge to unconfined aquifers occurs via percolation to the watertable from a large portion of the land surface, while recharge to confined aquifers from the surface only occur at the upstream end where the aquifer is exposed at the surface and flow is not confined. Topography of an area affects the recharge and discharge of a groundwater body significantly. Recharge is defined as the portion of groundwater flow that is directed away from the watertable, and discharge being defined as the flow that is directed towards the watertable at or near surface water bodies (e.g. river).

The lag in time that recharge enters the shallow aquifer, however, depends on the height of the watertable and the hydraulic properties of the groundwater zones. The time delay cannot be directly measured and must be estimated through an iterative process of altering the lag value and comparing the simulated variations in watertable with observed values (Neitsch et al. 2001, p.162). A recent study of the Namoi's river-aquifer connectivity to be quite high, hence it is assumed the time lag between deep drainage and recharge to be minimal. This has implications for the impact of deep drainage and salinity contribution, since shallow aquifers in the Mooki are very saline, and has the potential to increase soil and deep aquifer salinity.

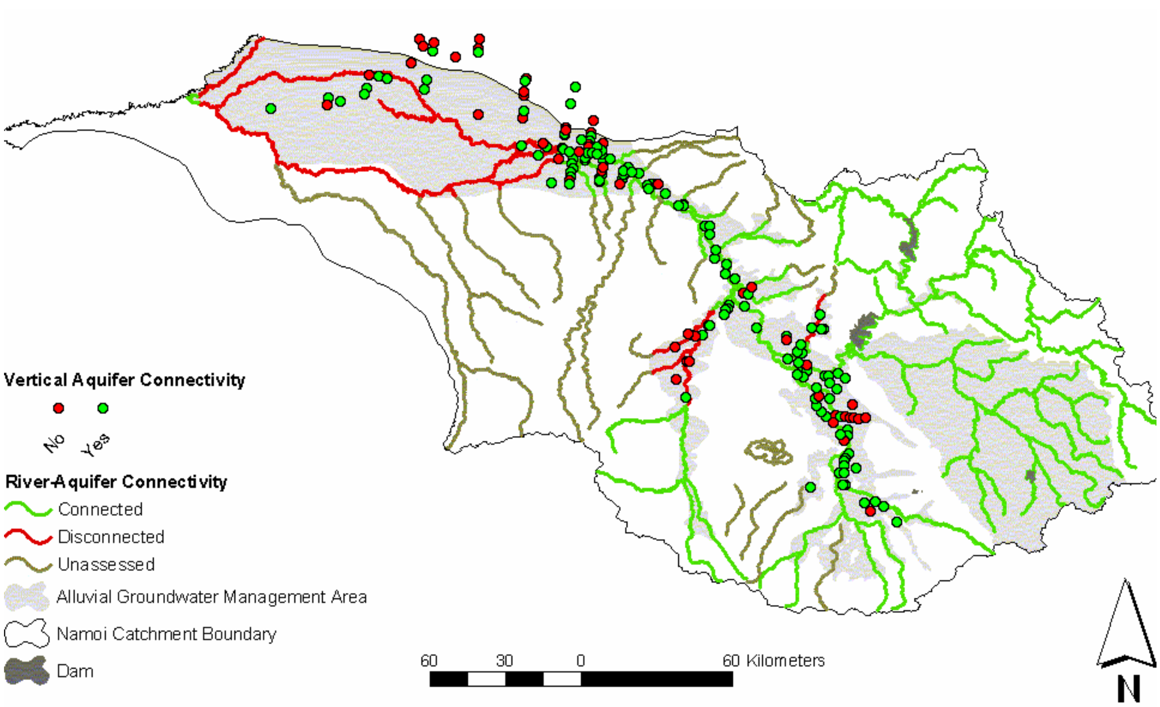

Figure A 2: River-aquifer connectivity in the Namoi (source: Ivkovic 2005, pers. comm.)

Appendix B: Recharge Levels Determined in the Groundwater Water Sharing Plan (source: Aquillina 2003 p.1 and 5).

Upper Namoi Zones	Description	Recharge ML/yr
1	Boramil Ck Groundwater Source	2,100
2	Cox's Ck (Mullaley to Boggabri) Groundwater Source	7,200
3	Mooki Valley (Breeza to Gunnedah) Groundwater Source	17,300
4	Namoi Valley (Keepit Dam to Gin's Leap) Groundwater Source	25,700
5	Namoi Valley (Gin's Leap to Narrabri) Groundwater Source	16,000
6	Tributaries of the Liverpool Range (South to Pine Ridge Road) Groundwater Source	14,000
7	Yarraman Ck (East of Lake Goran to Mooki River) Groundwater Source	3,700
8	Mooki Valley (Quirindi – Pine Ridge Road to Breeza)	16,000
9	Cox's Ck (up-stream Mullaley) Groundwater Source	11,400
10	Warrah Ck Groundwater Source	4,500
11	Maules Ck Groundwater Source	2,200
12	Kelvin Valley Groundwater Source	2,000

Appendix C: Method to Determine Accurate HU Irrigation Scheduling

Cotton Irrigation Scheduling – by Heat Units

Heat-units (HU) refer to the amount of energy received by plants, for which a minimum amount must be received to reach certain stages of growth. Irrigation events could be scheduled according to the HU received, since the amount of irrigation water required by the crop depends on the stage of growth. For cotton, the first irrigation should occur halfway between squaring (emerging cotton flower bud) and flowering, and the final irrigation should occur when 60% of cotton bolls are open. For heavy soils in northern NSW, the recommended practice is to stop irrigation at 20% open bolls or mid-March. Irrigation should be at even intervals from the first irrigation, which is approximately every 162 HU received (every 12-14 days) (Milroy et al. 2002). The heat-units required to reach each developmental stage in cotton's growth was based on data from Myall Vale in Namoi. The minimum day degrees (HU) required and the dates at which each stage occurred are shown in Table A1. HU scheduling was then devised based on this information and tested for its yield response in SWAT.

Table A1: Minimum heat units required for cotton development (source: CCCRC 2005).

From planting to:	Minimum day degrees required	Crop stages for Myall Vale
Emergence	80	12th October
5th True leaf	330	9th November
1st Square	505	27th November
1st Flower	777	19th December
Peak Flower	1302	25th January
Open boll	1527	11th February
60% open	2050	25th March

However, HU irrigation scheduling generated poor yield response functions. Crop growth for all HRUs was low and no distinct relationship could be drawn between the irrigation level and yield. It was concluded that perhaps too much water was available in the soil, so the plant had full access to moisture and little irrigation was required. This was related to the timing of irrigation in the HU scheduling, which was probably occurring when the plant did not need it, and only few of the irrigation events were occurring at the correct time. This was most likely due to miss-timing in the growth stages, because the same HU irrigation scheduling was applied to all HRUs regardless the soil type. Therefore, advice was sought from an agronomist and the recommendation was that field capacity of the soils (soil water holding capacity) should be taken into account in order to set an appropriate heat unit schedule. The greater the field capacity the less frequent and less irrigation is required and vice versa.

Campbell (2006, pers. comm.) suggested using separate scheduling for each soil type according to its field capacity. The advice was to group the HRUs into like soils and observe the amount of time it takes for a full field capacity to be used up. Irrigation should then be timed to occur with about 10% field capacity to prevent water stress. Accurate HU scheduling could be devised in the following manner:

1. Alter the input file in SWAT to produce simulation output in daily-time steps;
2. Determine which groups of soil should be aggregated into like soils (heavy clay or permeable soils) by checking soil input files;
3. Looking at the output files, take the sum of evapotranspiration (E_t) over the period it takes to use up total water capacity. Then time irrigation to occur at around 90% of soil water capacity to prevent stress;
4. However, since SWAT only allows irrigation scheduling by HU or date, not E_t, the HU associated with 90% soil water capacity need to be determined. The HU received over the number of days it takes to use up total water capacity is found using the following equation:

   $$GDD = \sum Y_{Days}\left(T_{mi} - T_b\right)$$

   This reads as: heat units received = sum over Y days of the difference between mean temp, T_{mi}, of day y and base temperature, T_b, for cotton (12°C). This allows the HU received to be associated with the soil water capacity.

5. Once GDD is obtained, irrigation events in SWAT could be scheduled according to the amount of HU received for every time soil water capacity is down to 90%.

However, due to limited computer capacity, it was extremely difficult to obtain the output required from SWAT through brute-force. This was due to the computer memory requirement in producing daily-step files for all HRUs. The second-best option was to use date scheduling, which generated reasonable yield responses to irrigation and was also computationally inexpensive.

Cotton Irrigation Scheduling – by Date

The most basic irrigation scheduling is by date, following a set irrigation path. The problem with date-scheduling is that irrigation takes place regardless of when rainfall has occurred. Irrigation would take place on the specified date even if a rainfall event has just occurred, and no irrigation takes place even if crops are water stressed and there no irrigation is scheduled. In reality, irrigators often use neutron probe readings of soil moisture to determine when irrigation should occur. It would therefore be more realistic to schedule according to HU, however for the reasons above, it was not possible to extract exact HU-scheduling for all HRUs. The yields produced using date-irrigation was reasonable, and was adjusted to a discrete distribution of cotton yields for north-eastern NSW to better correspond with the yields obtained in the Mooki basin.

Appendix D: Developing an Optimiser to Incorporate Return Flows

In order to incorporate return flows into the decision making process, there needs to be a feedback mechanism in SWAT to account for the impact of upstream landuse on downstream water supply. The suggested framework for finding an optimum is to enumerate the entire range of possible outcomes or use the differential evolution algorithm, suggested in Figure A 3.

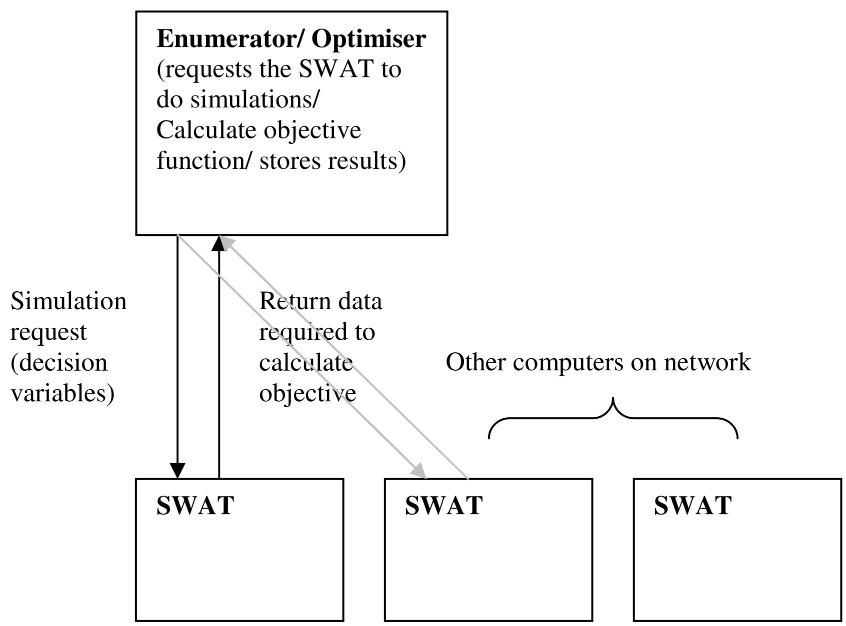

Figure A 3: Optimiser/enumerator schematic (source: Neal 2005, pers. comm.).

Following the optimisation problem in this thesis, each of the 53 HRU has nine possible production activities. For the purposes of confining the problem to a finite space in the optimiser, it is assumed that once a production activity is chosen, all of the area of the HRU must be produced under the selected activity. This leads to nine possible outcomes for each HRU.

Without hydrological links between each HRU, the optimisation problem is quite manageable, with just $9 \times 53 = 477$ combination of outcomes. However, the complexity arises where the impact of return flows on downstream water supply are taken into consideration. Given there are 53 HRUs, the decision made at each HRU affects the return flows downstream and hence influences the water supply. Given there are nine possible outcomes for each of the 53 HRUs, the total combination of HRUs and decision variables are $9^{53} = 3.76\times10^{50}$.

Advances in computational power means it is possible to simulate every one of the 3.76×10^{50} scenarios – eventually. Each SWAT simulation takes approximately 5 minutes. Hence to find the optimal combination by enumeration (calculating every possible combination) would require $5mins \times 3.76\times10^{50} = 1.88\times10^{51}\ minutes$. In other words, it would take $1.88\times10^{51}/(60mins\times24hrs) = 1.31\times10^{48}\,days$ for *one* standard computer. If a computer network of 40 computers was available fulltime, approximately $1.31\times10^{48}/40 = 3.28\times10^{46}$ days would be required for complete enumeration.

If stochastic optimisation techniques were used to find "near optimal" solutions, then complete enumeration may not be required, reducing the computation load. One example is the Differential Evolution algorithm (Neal 2005, pers. comm.). The objective to be maximised would be the profit from all farm activities less a costly weight multiplied by the amount by which constraints are exceeded. For example:

Objective = Profit - weight1×excess salinity - weight2×water over-consumption

The steps required are as follows:

1. Set SWAT up to run simulations and return the information required to calculate the objective function above, which will require coding in FORTRAN (programming code in SWAT).
2. Create/modify code for either:

   a) An *enumeration* algorithm that will request SWAT to do simulations and store results.

   b) An *optimisation* algorithm that will request SWAT to do simulations and store results (e.g. the Differential Evolution algorithm).

3. Buy supercomputer time, or parallelise the problem so it can be run over a network of computers.

While the use of an enumerator or optimiser would be the first-best option for the simulation problem considered in this thesis, limitations on technical expertise and time mean that it is not possible to develop an appropriate simulator within the available timeframe.

Appendix E: Solutions from the economic optimisation.

REFERENCES

ACIL Consulting (2002), *Economic Impacts of the Draft Water Sharing Plan, An Independent Assessment for the NSW Department of Land and Water Conservation*, Sydney.

Alley, W.M., and Leake, S.A. (2004), 'The Journey from Safe Yield to Sustainability', *Ground Water* 42(1), 12-16.

Aluwihare, P., Crean, J., Young, R. (2001), *On-farm impacts of flow-sharing options for the Mooki River, Final Report to the Namoi Unregulated River Management Committee*, NSW Agriculture Economic Services Unit, Orange.

Aluwihare, P., Jones, R., Letcher, R. (2005), *The farm level impacts of water sharing plans in the Namoi Valley: a stochastic dynamic programming analysis*, paper presented to the 49th Annual Conference of the Australian Agricultural and Resource Economics Society, Coffs Harbour, 9-11 February 2005.

Ancev, T. and Vervoort, W. (2007), 'The National Plan for Water Security: Taking over the Role of a Market?', *Connections: Farm, Food and Resource Issues,* The University of Melbourne, Melbourne.

Ancev, T., Odeh, I.O. (2005), 'Use of Spatially Referenced Data in Agricultural Economics Research', paper presented to the 49th Annual Conference of the Australian Agricultural and Resource Economics Society, Coffs Harbour, 9-11 February 2005.

Ancev, T., Stoecker, A.L., Storm, D.E., and White, M.J. (2006), 'The Economics of Efficient Phosphorus Abatement in a Watershed', *Journal of Agricultural and Resource Economics* 31(3), 529-548.

Ancev, T., Vervoort, W., Odeh, I.O.A. (2004), *Economic Evaluation of Catchment Management Options in the Irrigated Cotton Areas of the Upper Murray-Darling Basin in New South Wales, Australia*, Selected Paper, Annual Meeting of the American Agricultural Economics Association, Denver, Colorado, 1-4 August 2004.

Anthony, D. (1998), 'Irrigation in the cotton industry', *Proceedings from the 1998 National Conference and Exhibition*, Brisbane, 19-21 May 1998.

Anthony, D. (2003), *Water Use Efficiency in the Cotton Industry - Moving Ahead*, [www.atse.org.au/uploads/Anthony03.pdf], accessed August 2006.

Aquillina, J. (2003), *Water Sharing Plan for the Upper and Lower Namoi Groundwater Sources 2003 Order,* Department of Land and Water Conservation, Sydney.

Arthington, A.H. (1996), 'The effects of agricultural land use and cotton production on tributaries of the Darling River, Australia', *GeoJournal* 40(1), 115-125.

Asseng, S., Pracilio, G., Dolling, P., and Wong, M., (2003), 'Deep drainage under cropping in Western Australia', *Solutions for a better environment: Proceedings of the 11th Australian Agronomy Conference*, Geelong, 2-6 February 2003.

Australian Bureau of Agricultural and Resource Economics (ABARE) (2005), *Australian Commodity Statistics 2005*, Canberra.

Australian Bureau of Agricultural and Resource Economics (ABARE) (2006), *Australian Commodities no. 06.3*, Canberra.

Australian Bureau of Statistics (ABS) (2005), *Historical Selected Agriculture Commodities, by State (1861 to Present)*, Sydney.

Australian Water Resources Council (AWRC) (1975), *Review of Australia's Water Resources 1975*, Australian Government Publishing Service, Canberra.

Bateman, I.J., Jones, A.P., Lovett, A.A., Lake, I.R. and Day, B.H. (2002), 'Applying Geographical Information Systems (GIS) to Environmental and Resource Economics', *Environmental and Resource Economics* 22(1-2), 219-269

Beare, S., Heaney, A. and Mues, C. (2001), 'Modelling salinity management options in the Murray-Darling Basin', *Salinity economics - a national workshop*, Orange, 22-23 August 2001.

Beare, S.C., Bell, R., and Fisher, B.S. (1998), 'Determining the value of water: the role of risk, infrastructure constraints, and ownership', *American Journal of Agricultural Economics* 80(5), 916-940.

Bennett, J. (2005a), 'Australasian environmental economics: contributions, conflicts and 'cop-outs'', *Australian Journal of Agricultural and Resource Economics* 49, 243-261.

Bennett, J. (2005b), 'Realising Environmental Demands in Water Markets', in *The Evolution of Markets for Water, New Horizons in Environmental Economics*, ed. Bennett, J., Edward Elgar Publishing, Cheltenham,165-179.

Bennetton, J., Cashin, P., Jones, D. and Soligo, J. (1998), 'An economic evaluation of bushfire prevention and suppression', *Australian Journal of Agricultural & Resource Economics* 42(2), 149-75.

Bernardo, D.J., Whittlesey, N.K., Saxton, K.E., and Bassett, D.L. (1987), 'An Irrigation Model for Management of Limited Water Supplies', *Western Journal of Agricultural Economics* 12(2),164-173.

BHP Billiton (2006), 'Energy Coal CSG (Summary)', *BHP Billiton Sustainability Report 2006,* [sustainability.bhpbilliton.com/2006/documents/performance/EnviroSummaryPdfs/ECL_2006.pdf], accessed January 2007.

BHP Billiton (2007), ‘Resource Use - Water’, *BHP Billiton Sustainability Report 2006,* [sustainability.bhpbilliton.com/2006/environment/ourPerformance/resourceUse/water.asp], accessed January 2007.

Bjornlund, H. (2003a), ‘Efficient Water Market Mechanisms to Cope with Water Scarcity’, *Water Resources Development* 19(4), 553-567.

Bjornlund, H. (2003b), ‘Farmer participation in markets for temporary and permanent water in southeastern Australia’, *Agricultural Water Management* 63, 57-76.

Bockstael, N.E. (1996), ‘Modeling Economics and Ecology: The Importance of a Spatial Perspective’, *American Journal of Agricultural Economics* 78(5), 1168-1180

Borthwick,D. (2006), ‘Taking Environmental Water Policy Forward Under the NWI’, *Outlook 2006*, Canberra.

Boyce (2005), *Australian Cotton Comparative Analysis, 2004 Crop*, Cotton Research and Development Corporation, Narrabri, October 2005.

Bredehoeft J. (2006), ‘On Modeling Philosophies’, *Ground Water* 44(4), 496-499.

Bureau of Meteorology (BOM) (2007), *Drought*, [www.bom.gov.au/lam/climate/levelthree/c20thc/drought.htm], accessed July 2007.

Carey, J., Sunding, D.L., and Zilberman, D. (2002), ‘Transaction costs and trading behaviour in an immature water market’, *Environment and Development Economics* 7, 733-750, Cambridge University Press, United Kingdom.

Caswell M.F. (1991), ‘Irrigation technology adoption decisions: empirical evidence’, In: Dinar, A. and Zilberman, D. (ed), *The Economics and Management of Water and Drainage in Agriculture*, Kluwer Academic Publishers, Nowell, 229–250.

Caswell, M.F., Lichtenberg, E. Zilberman, D., (1990), 'The effects of pricing policies on water conservation and drainage', *American Journal of Agricultural Economics* 72(4), 883-890.

Centre for Agricultural and Regional Economics (CARE) (2003), *A Socio-Economic Analysis of the Impact of the Reduction in Groundwater Allocations in the Namoi Valley. The Impact of the Namoi Groundwater Sharing Plan*, University of New England Institute for Rural Futures, Armidale.

Clapp, J.M., Rodriguez, M. and Thrall, G. (1997), 'How GIS Can Put Urban Economic Analysis on the Map', *Journal of Housing Economics* 6, 368-386.

Commonwealth of Australia (2003), *Parliamentary Debates, Senate Official Hansard no. 10*, Thursday, 11 September Fortieth Parliament, First Session – Sixth Period, [www.aph.gov.au/hansard/senate/dailys/ds110903.pdf], accessed July 2007.

Cooperative Research Centre for Irrigation Futures (CRCIF) (2005), *Irrigation in Perspective: Irrigation in the Murray and Murrumbidgee Basins*, CSIRO Land and Water, Commonwealth of Australia.

Cotton Australia (2007), 'Biotechnology', *Fact Sheets*, [www.cottonaustralia.com.au/factSheets/resources/biotechnology2.pdf], accessed May 2007.

Cotton Catchment Communities Cooperative Research Centre (CCCCRC) (2005), *Tools: Catchment Development Temperatures from October 1*, [www.cotton.crc.org.au/Tools/Agronomy/avg_wea.htm], accessed December 2006.

Cotton Catchment Communities Cooperative Research Centre (CCCCRC) (2007), *Namoi Catchment*, [cotton.pi.csiro.au/content/Catchments/MyCatchments/NamoiCatchment.aspx], accessed July 2007.

Council of Australian Governments (COAG) (1994), *Attachment A - Water Resource Policy, Council of Australian Governments' Communiqué 25 February 1994*, Canberra.

Council of Australian Governments (COAG) (2004), *Council of Australian Governments Meeting, 25th June 2004*, COAG, Canberra.

Council of Australian Governments (COAG) (2005a), *About COAG*, [www.coag.gov.au/], accessed March 2007.

Council of Australian Governments (COAG) (2005b), *Commonwealth-State Ministerial Councils, A Compendium*, The Department of the Prime Minister and Cabinet, Canberra.

Council of Australian Governments (COAG) (2006), *Supplementary Intergovernmental Agreement on Addressing Water Overallocation and Achieving Environmental Objectives in the Murray-Darling Basin*, COAG, Canberra.

Cox, J. and Warner and R. (2007), *Water Trading – Panacea or Placebo*, IPART, Sydney.

Crase, L. and Jackson, J. (1998), 'A Statistical Analysis of the Characteristics of Irrigation Farmers' Responses to Reduced Irrigation Water: a Case Study of Irrigation Farmers Facing Water Policy Reform in the Murray LWMP Area', in Crase, L., O'Reilly, L., and Dollery, B. (2000), 'Water Markets as a Vehicle for Water Reform: the Case of New South Wales', *Australian Journal of Agricultural and Resource Economics* 44(2), 299-321.

Crase, L., O'Reilly, L., and Dollery, B. (2000), 'Water Markets as a Vehicle for Water Reform: the Case of New South Wales', *Australian Journal of Agricultural and Resource Economics* 44(2), 299-321.

Davidson, B.R. (1969), *Australia wet or dry? The physical and economic limits to the expansion of irrigation*, Melbourne University Press, Victoria.

Department of Agriculture, Fisheries and Forestry (DAFF) (2006), *Murray-Darling Basin Policies and Programs*, Canberra.

Department of Environment and Heritage (DEH) (2007), *Australian Natural Resources Atlas Australia, Land*, Commonwealth of Australia, Canberra.

Department of Infrastructure, Planning and Natural Resources (DIPNR) (2004a), *Understanding Your Water Account*, Tamworth.

Department of Infrastructure, Planning and Natural Resources (DIPNR), (2004b), *Water Licensing under the Water Management Act 2000*, Tamworth.

Department of Infrastructure, Planning and Natural Resources (DIPNR), (2004c), *Water Sharing Plan for the Phillips Creek, Mooki River, Quirindi Creek and Warrah Creek Water Sources*, Tamworth.

Department of Infrastructure, Planning and Natural Resources (DIPNR) (2005b), *Available Water Determinations Register*, [www.wma.dnr.nsw.gov.au/wma/DeterminationSearch.jsp?selectedRegister=Determination], accessed December 2005.

Department of Land and Water Conservation (DLWC) (1998), *Stressed Rivers Assessment Report*, April, Sydney.

Department of Land and Water Conservation (DLWC) (2001), *Submission to IPART on bulk water pricing 2001/02 – 2003/04*, Sydney.

Department of Land and Water Conservation (DLWC) (2002), *Water Quality in the Namoi Catchment 2000-2001*, Sydney.

Department of Land and Water Conservation (DLWC) (2003), *Information for licence holders about the 10-year water sharing plan for the Upper and Lower Namoi aquifers report no. 1*, Sydney.

Department of Natural Resources (DNR) (2007), *Water Management Act Registers*, [www.wma.dnr.nsw.gov.au/wma/WALStatisticsSearch.jsp?selectedRegister=WAL Statistics], accessed June 2007.

Department of the Environment and Water Resources (DEWR) (2007), *Salinity*, [csd.unl.edu/general/glossary-letter.asp?Definition=D], accessed April 2007.

Department of the Prime Minister and Cabinet (DPMC) (2006), *National Water Initiative*, [www.pmc.gov.au/water_reform/nwi.cfm], accessed March 2007.

Dingman, S.L. (1994), 'Physical hydrology', in Neitsch, S.L., Arnold, J.G., Kiniry, J.R., Williams, J.R., *Soil and Water Assessment Tool Theoretical Documentation*, Texas.

Doss, C.R. and Taff, S.J (1996), 'The Influence of Wetland Type and Wetland Proximity on Residential Property Values', *Journal of Agricultural and Resource Economics* 21, 120-129.

Dwyer, G., Loke, P., Appels, D., Stone, S., Peterson, D. (2005), 'Integrating rural and urban water market in south east Australia: Preliminary analysis', paper presented at the *OECD Workshop on Agriculture and Water: Sustainability, Markets and Policies,* Adelaide, 14-18November.

Easter, K. W., Rosegrant, M.W. and Dinar, A. (1998), 'The Future of Water Markets: A Realistic Perspective', in Easter, K. W., Rosegrant, M.W. and Dinar, A. (eds), *Markets for Water: Potential and Performance*, Kluwer Academic Publishers, Boston.

Foley, J.P. and Raine, S.R. (2001), 'Centre Pivot and Lateral Move Machines in the Australian Cotton Industry', *National Centre for Engineering in Agriculture Publication* 1000176/1, USQ, Toowoomba.

Freebairn, J. (2003), 'Policy Forum: Water Pricing and Availability, Principles for the Allocation of Scarce Water', *The Australian Economic Review* 36(2), 203-12.

Freebairn, J. (2005), 'Principles and Issues for Effective Australian Water Markets', in Bennett, J. (ed) *The Evolution of Markets for Water, Theory and Practice in Australia*, Edward Elgar, Cheltenham, 8-23.

Gaffney, M. (1997), 'What Price Water Marketing?: California's new frontier', *The American Journal of Economics and Sociology* 56, 475-521.

Geoscience Australia (2007), *Australian Spatial Data Directory*, [asdd.ga.gov.au/], accessed June 2007.

Godden, D. (1997), *Agricultural and Resource Policy, Principles and Practice*, Oxford University Press, Melbourne.

Goesch, T. (2001), 'Delivery Charges for Water: Their Impact on Interregional Trade in Water Rights', *Australian Commodities December Quarter*, 626-34.

Goesch, T. and Beare, S. (2004), 'Water Rights and Trade, Meeting the Reform Agenda', *Australian Commodities* 11(1), Canberra.

Goesch, T. and Heaney, A. (2003), 'Government purchase of water for environmental outcomes', *ABARE eReport*, November 2003.

Greenville, J. and MacAulay, T.G. (2006), 'Protected Areas in Fisheries: a Two-Patch, Two-Species Model', *Australian Journal of Agricultural and Resource Economics* 50(2), 207-226.

Gretton, P. and Salma, U. (1997), 'Land Degradation: Links to Agricultural Output and Profitability', *Australian Journal of Agricultural and Resource Economics* 41, 209-225.

Grismer, M.E. and Gates, T.K. (1991), 'Hydrologic aspects of saline watertable management in regional shallow aquifers', *The Economics and Management of Water and Drainage in Agriculture*, (ed.) Dinar, A. and Zilberman, D., Netherlands, Kluwer Academic Publishers.

Gunnedah Shire (2006), *Shire Profile*, [www.infogunnedah.com.au/shire_profile/main.html], accessed April 2006.

Haisman, B. (2005), ‘Impacts of Water Rights Reform in Australia’, in *Water Rights Reform,* eds Bruns, B.R., Ringler, C. and Meinzen-Dick, R., International Food Policy Research Institute, Washington D.C.

Hameed, T. and O'Neill, R. (2005), ‘River Management Decision Modelling in IQQM’, paper presented at MODSIM, Melbourne, 12-15th December.

Hamparsum, J. (2004), *FutureWater Australia*, Irrigation Australia, Adelaide.

Hartwick, J.M. and Olewiler, N.D. (1986), *The Economics of Natural Resource Use*, 2nd ed., Harpercollins College Div, New York.

Hatchett, S.A, Horner, G.L. and Howiit, R.E. (1991), ‘A Regional Mathematical Programming Model to Assess Drainage Control Policies’, 465-87, in Dinar, A. and Zilberman, D. (eds), *The Economics and Management of Water and Drainage in Agriculture*, Norwell, Mass and Dordrecht: Kluwer Academic.

Heady, E.O. and Hall, H.H. (1968), ‘Linear and Nonlinear Spatial Models in Agricultural Competition, Land Use, and Production Potential’, *American Journal of Agricultural Economics* 50(5), 1539-1548.

Heady, E.O., Madsen, H.C., Nicol, K.J., and Hargrove, S.H. (1973), ‘National and Interregional Models of Water Demand, Land Use, and Agricultural Policies’, *Water Resources Research* 9(4), 777-791.

Heaney, A. and Beare, S.C. (2001), ‘Water Trade and Irrigation, Defining Property Rights to Return Flows’, *Australian Commodities* 8(2) 339-348.

Heaney, A., Beare, S., and Bell, R. (2001), ‘Evaluating improvements in irrigation efficiency as a salinity mitigation option in the South Australian Riverland’, *Australian Journal of Agricultural and Resource Economics* 45(3), 477-493.

Heaney, A., Beare, S., Goesch, T. (2002), 'Environmental flows and water trade', *ABARE Current Issues 02.3*, March 2002.

Heaney, A., Thorpe, S., Klijn, N., Beare, S. and Want, S. (2004), 'Water charges and interreginal trade in the southern Murray Darling Basin', *ABARE Conference Paper* 04.14.

Horan, R.D. (2001), 'Differences in Social and Public Risk Perceptions and Conflicting Impacts on Point/Nonpoint Trading Ratios', *American Journal of Agricultural Economics* 83(4), 934-941.

Howard, J. (2007), *A National Plan for Water Security*, Commonwealth of Australia, Canberra.

Huffaker, R. and Whittlesey, N. (2000), 'The allocative efficiency and conservation potential of water laws encouraging investments in on-farm irrigation technology', *Agricultural Economics* 24, 47-60.

Independent Pricing and Regulatory Tribunal (IPART) (2001), *Department of Land and Water Conservation Bulk Water Prices, from 1 October 2001, Determination no. 3*, December 2001.

Independent Pricing and Regulatory Tribunal (IPART) (2004), *Bulk Water Prices from 2005/06 Issues Paper*, IPART, Canberra.

Independent Pricing and Regulatory Tribunal (IPART) (2005), *State Water Corporation and Water Administration Ministerial Corporation, Bulk Water Prices Determination for 2005-06, Report no. 8 and 9*, Canberra.

Independent Pricing and Regulatory Tribunal (IPART) (2006), *Bulk Water Prices for State Water Corporation and Water Administration Ministerial Corporation from 1 October 2006 to 30 June 2010*, *Determination No. 4 and 5*, September 2006.

Industry Commission (1992), *Water Resources and Waste Water Disposal, Report No. 26*, 17 July, Canberra.

Jakeman, A.J., Letcher, R.A., Newham, L.T.H. and Norton, J.P. (2005), 'Integrated catchment modelling: issues and opportunities to support improved sustainability outcomes', Paper presented at the *29th Hydrology and Water Resources Symposium*, Engineers Australia, Canberra, 21-23 February 2005.

Jobling, J. (2000), *Land and Water Conservation – Groundwater Licences*, Hansard Paper session 552, Questions & Answers Paper No. 54, [www.parliament.nsw.gov.au/prod/lc/qalc.nsf/0/CA25707400260AA3CA25706E001E3688], accessed May 2007.

Jolly ID, Williamson DR, et al. (2001), 'Historical stream salinity trends and catchment salt balances in the Murray-Darling Basin, Australia', *Marine and Freshwater Research* 52, 53-63.

Khanna, M., Yang, W., Farnsworth, R. and Onal, H. (2000), 'Cost-effective targeting of land retirement to improve water quality with endogenous sediment deposition coefficients', *American Journal of Agricultural Economics* 85(3), 538-553.

Lavitt, N. (1999), *Integrated Approach to Geology, Hydrogelogy and Hydrogeochemistry in the Lower Mooki River Catchment*, PhD dissertation, School of Geology University of New South Wales.

Legras, S. and Lifran, R. (2006), 'Dynamic taxation schemes to manage irrigation-induced salinity', *Environmental Modeling and Assessment* 11, 157-167.

Letcher, R.A. and Jakeman, A.J. (2002), *Experiences in an integrated assessment of water allocation issues in the Namoi river catchment*, Technical Report Working Paper, ICAM, ANU.

Lindo Systems (2007), *What's Best! 9.0*, [lindo.com/products/wb/wbm.html], accessed July 2007.

Mallawaarachchi, T. and Quiggin, J. (2001), 'Modelling socially optimal land allocations for sugar cane growing in North Queensland: A linked mathematical programming and choice modelling study', *Australian Journal of Agricultural and Resource Economics* 45, 385-409.

Marshall, J., Eveleigh, R., Hickman, M. (2002), 'Dryland Cotton Production', *Crop Yields*, *Cotton Cooperative Research Centre*, Narrabri.

Mawhinney, W. (2005), 'Case study, catchment water quality and cotton: northern NSW', *WaterPak 2005,* 268.

McCown R.L., Hammer G.L., Hargreaves J.N.G., Holzworth, D.P. and Freebairn D.M. (1996). *Agricultural Systems* 50, 255-271.

Milroy, S., Goyne, P. and Larsen, D. (2002), 'Irrigation Scheduling of Cotton', *Cotton Information Sheet*, Australian Cotton Cooperative Research Centre, Narrabri.

Murray, M. (2004), 'Auscott takes mechanical approach to water use efficiency', *Irrigation and water resources*, *Irrigation and water resources*, 26-29.

Murray-Darling Basin Commission (MDBC) (1995): *An Audit of Water Use in the Murray-Darling Basin,* Murray-Darling Basin Ministerial Council, Canberra

Murray-Darling Basin Commission (MDBC) (2001), *Integrated Catchment Management in the Murray-Darling Basin 2001-2010*, *Delivering a Sustainable Future*, Murray-Darling Basin Ministerial Council, June 2001.

Murray-Darling Basin Commission (MDBC) (2003), *Water Audit Monitoring Report 2001/02, Report of the Murray-Darling Basin Commission on the Cap on Diversions*, Murray-Darling Basin Ministerial Council, June 2003.

Murray-Darling Basin Commission (MDBC) (2004), *The Cap, Providing Security for Water Users and Sustainable Rivers*, Murray-Darling Basin Ministerial Council, Canberra.

Murray-Darling Basin Commission (MDBC) (2005a), 'Approved End-of-Valley Salinity Targets', *Basin Salinity Management Strategy Operational Protocols*, Murray-Darling Basin Ministerial Council, Canberra 2005.

Murray-Darling Basin Commission (MDBC) (2005b), *The Living Murray Business Plan,* Murray-Darling Basin Commission, Canberra.

Murray-Darling Basin Commission (MDBC) (2006a), *About MDB Initiative*, [www.mdbc.gov.au/about/murraydarling_basin_initiative__overview], accessed May 2007.

Murray-Darling Basin Commission (MDBC) (2006b), *Basin Salinity Management Strategy 2001-2015*, Murray-Darling Basin Commission, Canberra.

Murray-Darling Basin Commission (MDBC) (2006c), *Basin Tour – Irrigation*, [www.mdbc.gov.au/about/tour_the_basin/irrigation], accessed May 2007.

Murray-Darling Basin Commission (MDBC) (2007a), 'Progress Report (Water Recovery)', *The Living Murray*, [www.thelivingmurray.mdbc.gov.au/programs/water_recovery/progress], accessed June 2007.

Murray-Darling Basin Commission (MDBC) (2007b), *Review of Cap Implementation 2005/06, Publication No. 12/07,* Murray-Darling Basin Commission, Canberra.

Murray-Darling Basin Commission (MDBC) (2007c), 'The Living Murray Progress Report', *Murray-Darling Basin Commission*, [thelivingmurray.mdbc.gov.au/progress], accessed March 2007.

National Action Plan for Salinity and Water Quality (NAPSWQ) (2001), *Australia's Salinity Problem*, [www.napswq.gov.au/publications/salinity.html], accessed April 2006.

National Competition Council (NCC) (1998), *Compendium of National Competition Policy Agreements (second edition 1998), Part One – Competition Policy Agreements*, Commonwealth of Australia, Canberra.

National Land and Water Resources Audit (NLWRA) (2000), *Water in a dry land, issues and challenges for Australia's key resource*, Department of the Environment and Water Resources, Canberra.

National Land and Water Resources Audit (NLWRA) (2001), *Australian Water Resources Assessment 2000. Surface water and groundwater availability and quality*, National Land and Water Resources Audit, Canberra.

National Water Commission (NWC) (2005), 'Regional Water Resource, Assessment Upper Namoi Alluvium', *Australian Water Resources,* Australian Government, [www.water.gov.au/RegionalWaterResourcesAssessments/SpecificGeographicRegion/TabbedReports.aspx?PID=NSW_GW_004&Menu=Level1_7], accessed July 2007.

Natural Resource Management (NRM) (2004), *Salinity*, Australian Government, [www.nrm.gov.au/publications/salinity/index.html], accessed March 2007.

Neitsch, S.L., Arnold, J.G., Kiniry, J.R. and Williams, J.R. (2001), *Soil and Water Assessment Tool Theoretical Documentation Version 2000*, Grassland, Soil and Water Research Laboratory, Agricultural Research Service, Texas.

New South Wales (NSW) Farmers' Association (2003), 'Watertight Trading Rights', *The Primary Report, An Analysis of Issues Affecting the Rural Economy by NSW Farmers' Association*, May 2003.

New South Wales (NSW) Irrigators' Council (2001), *Fact Sheets, Namoi Valley*, [www.nswirrigators.org.au/pdf/catchment_profiles/Namoi.pdf], accessed May 2007.

New South Wales Department of Primary Industries (NSW DPI) (2006a), *Farm Budgets and Costs,* [www.agric.nsw.gov.au/reader/sumcropbud], accessed December 2005.

New South Wales Department of Primary Industries (NSW DPI) (2006b), 'Salinity tolerance in irrigated crops', *Salinity*, [www.agric.nsw.gov.au/reader/wm-plants-waterquality/dpi389plantsalinetolerance.htm], accessed January 2007.

Nordblom, T., Hume, I., Cresswell, H., Glover, M., Hean, R., Finlayson, J. and Wang, E. (2007), 'Minimising costs of environmental service provision: water-yield, salt-load and biodiversity targets with new tree planting in Simmons Creek Catchment, NSW, a dryland farming/grazing area', paper presented at the 50th Australian Agricultural and Resource Economics Society Annual Conference, Queenstown, 13-16 February, 2007.

O'Halloran, J. (2005), 'Case study, Linear move irrigators: Auscott Midkin, Moree', *WaterPak 2005*, 222.

Oliver, Ml (2007), 'River Murray: The Big Picture', *MurrayCare,* [www.murrayusers.sa.gov.au/big_picture2.php], accessed May 2007.

Paris, Q. (1991), *An Economic Interpretation of Linear Programming*, Ames, Iowa State University Press.

Pigram, J.J. (1986), *Issues in the Management of Australia's Water Resources*, Longman Cheshire, Melbourne.

Powell, R., Thompson, D., Chalmers, L. and Gabbott, A., (2003), *A socio-economic analysis of the impact of the reduction in groundwater allocation in the Namoi Valley, final report*, Department of Transport and Regional Services, Canberra.

Productivity Commission (PC) (2003), *Water Rights Arrangements in Australia and Overseas: Annex B, Commission Research Paper*, Productivity Commission, Melbourne.

Productivity Commission (PC) (2004), 'Modelling Water Trade in the Southern Murray-Darling Basin', *Productivity Commission Staff Working Paper*, Melbourne.

Quiggin, J. (2001), 'Environmental Economics and the Murray-Darling River System', *Australian Journal of Agricultural and Resource Economics* 45(1), 67-94.

Raine, S.R., Foley, J.P., and Henkel, C.R. (2000), 'Drip Irrigation in the Australian Cotton Industry: a Scoping Study', *National Centre for Engineering in Agriculture Publication* 179757/2, USQ, Toowoomba.

Randall, A. (1981a), 'Property Entitlements and Pricing Policies for a Maturing Water Economy', *Australian Journal of Agricultural Economics* 25(3), 195-220.

Randall, A. (1981b), *Resource Economics, an Economic Approach to Natural Resource and Environmental Policy*, Grid Publishing, Columbus, Ohio.

Renwick, M.E. and Green, R.D. (2000), 'Do Residential Water Demand Side Management Policies Measure Up? An Analysis of Eight California Water Agencies', *Journal of Environmental Economics and Management* 40(1), 37-55.

Rolfe, J. (2005), 'Potential Efficiency Gains from Water Trading in Queensland', in *The Evolution of Markets for Water, New Horizons in Environmental Economics*, ed. Bennett, J., Edward Elgar Publishing, Cheltenham, 119-138.

Rosenberger, R.S. and Loomis, J.B. (2003), 'A Primer on Nonmarket Valuation: Benefit Trasnfer, in *A Primer on Nonmarket Valuation,* eds Champ, P.A., Boyle, K.J., Brown, T.C. Kluwer Academic Publishers, 445-82.

Scheierling, S.M., Young, R.A. and Cardon, G.E. (2004), 'Determining the Price-Responsiveness of Demands for Irrigation Water Deliveries versus Consumptive Use', *Journal of Agricultural and Resource Economics* 29(2), 328-345.

Sexton, M. (2006) [online], 7.30 Report 10th May 2006, *Doubts surround Murray River target*, Australian Broadcasting Corporation.

Silburn, M. and Montgomery, J. (2005), 'Deep drainage under irrigated cotton in Australia: a Review, *WaterPak 2005*, 32-40.

Smith, D.I. (1998), *Water in Australia: resources and management*, Oxford University Press, Melbourne.

Smith, P. and Richards, A. (2003), *How much does it cost to pump?*, AGFACTS, NSW Agriculture.

Smith, S. (2000), 'New Water Management Legislation in NSW: A Review', *Briefing Paper 8/2000*, Research Papers, Parliament of New South Wales.

State Water (SW) (2005), *About State Water*, [www.statewater.com.au/index.htm], accessed June 2005.

Strang, M. (2006) [online], ABC Rural, 20th February 2006, *Gunnedah Water concerns after coal exploration announcement*, Australian Broadcasting Corporation.

Takayama, T. and Judge, G. (1971), 'Spatial Price Equilibrium and Linear Programming', *American Economic Review*.38, 496-509.

Tanaka K. and Wu, J. (2004), *Evaluation of Conservation Policies for Reducing Nitrogen Loads to the Mississippi River and Gulf of Mexico*, Selected Paper, Annual Meeting of the American Agricultural Economics Association, Denver, Colorado, 1-4 August 2004.

Tennakoon, S.B. and Milroy, S.P. (2003), 'Crop water use and water use efficiency on irrigated cotton farms in Australia', *Agricultural Water Management* 61, 179-194.

Thomson, N.J. (1979), 'Cotton', in J.V. Lovett and J.K. Lazenby (ed), *Australian Field Crops*, Vol. 2, Angus and Robertson.

Tinbergen, I. (1950), *On the theory of economic policy*, Elsevier, North Holland.

Trewin, D. (2006), *Water Use on Australian Farms 2004-05*, *report no. 4618.0,* Australian Bureau of Statistics, Sydney.

Triantafilis, J., Huckel, A.I. and Odeh, I.O.A. (2003), 'Field-scale assessment of deep drainage risk', *Irrigation Science* 21, 183-192.

Troy, A. and Wilson, M.A. (2006), 'Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer', *Ecological Economics* 60(2), 435-449.

Tsur, Y. and Dinar, A. (1995), *Efficiency and Equity Considerations in Pricing and Allocating Irrigation Water*, Policy Research Working Paper, The World Bank, Agriculture and Natural Resources Department, Washington, May 1995.

University of Nebraska Lincoln (UNL) (2007), *Glossary of Selected Geologic Terms*, Conservation and Survey Division, School of Natural Resources, University of Nebraska Lincoln, [csd.unl.edu/general/glossary-letter.asp?Definition=D], accessed April 2007.

Varela-Ortega, C., Sumpsi, J. M., Garrido, A., Blanco, M., Iglesias, E. (1998), Water pricing policies, public decision making and farmers' response: implications for water policy, *Agricultural Economics* 19, 193-202.

Voinov, A. (1997), *PLM-Patuxent Landscape Model, Project Description*, The University of Vermont, Burlington, [www.uvm.edu/giee/PLM/Project.html#Bingham], accessed March 2007.

WaterExchange (2007), *WaterExchange,* [www.waterexchange.com.au/], accessed January 2007.

Weber, M.L. (1999), 'Markets for Water Rights under Environmental Constraints', *Journal of Environmental Economics and Management* 42, 53-64.

Weinberg, M., Kling, C.L., and Wilen, J.E. (1993), 'Water markets and water quality', *American Journal of Agricultural Economics* 75, 278-291,

Whittaker, G., Confesor, R. Jr., Griffith, S.M., Färe, R. (2007), 'A Hybrid Genetic Algorithm for Multi-Objective Data Envelopment analysis', Paper presented at the Centre for Applied Economic Research *Policy choices for Salinity Mitigation: Bridging the Disciplinary Divides*, Coogee, 1-2 February 2007.

Whitten, S.M., Khan, S., Collins, D., Robinson, D., Ward, J. and Rana, T. (2005), *Tradeable recharge credits in Coleambally Irrigation Area: Report 7 Experiences, lessons and findings*, CSIRO and BDA Group.

Wilson, S. and Ivey, R. (2001), *Dryland salinity in the Namoi, Gwydir and NSW Border Rivers. What is it costing you?*, Murray-Darling Basin Commission, Canberra.

Wong, C.M., Williams, C.E., Collier, U., and Schelle, P. (2007), *World's top 10 rivers at risk*, WWF International, Gland.

Young, M. and McColl, J. (2005), 'Defining Tradable Water Entitlements and Allocations: A Robust System', *Canadian Water Resources Association* 31(1), 65-72.

Young, R.A. (2005), *Determining the Economic Value of Water, Concepts and Methods*, RFF Press, Washington D.C.

Zilberman, D. and Lipper, L. (2002), The economics of water use, in Bergh, J. C. J. M. van den (ed.), *Handbook of environmental and resource economics*, Edward Elgar Publishing Ltd, Cheltenham, 141-158.

Zilberman, D., Chakravorty, U. and Shah, F. (1991), 'Efficient management of water in agriculture', in Dinar, A. and Zilberman, D. (ed), *The Economics and Management of Water and Drainage in Agriculture*, Kluwer Academic Publishers, Nowell, 221-246.