UNIVERSITY OF SOUTHERN QUEENSLAND

An assessment of the economic value of using seasonal climate forecasting in water resources management

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Abstract

Australia's water resource systems are suffering from excessive diversion of surface flows with adverse effects on the riverine environments now becoming clearly evident. The capacity of water managers to achieve current reform aims whilst minimising impacts on rural communities will be improved with the aid of new technologies and decision-making processes. Seasonal climate forecasting (SCF) based on the relationship between the El Niño-Southern Oscillation phenomenon (ENSO) and streamflow is a technology that may play a part to improving the management of river-flow regimes providing benefits to both extractive and non-extractive (environmental) users of water.

This research uses a case study to test the use of SCF information in managing access to one component of irrigation water supply in the Border Rivers catchment in the northern part of the Murray-Darling Basin in eastern Australia. The aims were two-fold including developing an appropriate methodology and modelling framework that is transferable across a range of locations and evaluating the efficacy of seasonal climate forecasting information. A modelling approach tested water access rules by simulating both economic and environmental outcomes. These outcomes were analysed using a trade-off analysis based on the production possibility frontier (PPF) in conjunction with the Pareto principle whereby the SCF information would be considered efficacious if its use improved environmental outcomes without economic costs or visa versa.

Although seasonal climate forecasting has progressed significantly in recent years, there appears to be of little use of seasonal climate forecast information in catchment water management decision-making. Forecast accuracy, or the perceived lack of forecast accuracy, is cited as a key impediment to the uptake of forecast information in decision-making, despite the efforts of researchers to statistically validate forecast systems.

The research findings indicate that the use of SCF information was sufficiently accurate to improve economic outcomes without negatively impacting on environmental outcomes. In addition, an improvement in forecasting accuracy would further improve economic outcomes without major impacts on environmental outcomes. The increase in economic outcomes from using seasonal forecasting information are small relative to the total regional gross margin produced by the case study area in the absence of the SCF based water access rules for irrigation. This suggests that the study findings may not be of sufficient scale to convince decision-makers to adopt the information to assist in managing water access.

Certification of dissertation

I certify that the ideas, results, analyses, software and conclusions reported in this dissertation are entirely my own effort, except where otherwise acknowledged. I also certify that the work is original and has not been previously submitted for any other award, except where otherwise acknowledged.

John William Ritchie

Date 15/1/2010.

ENDORSEMENT

Supervisor

Dr Geoff Cockfield

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1. Introduction

This thesis considers whether or not weather forecasting techniques can be used to improve outcomes from the management of water resources. I develop a forecast-based water allocation model and test this in a case study region to assess the outcomes of using seasonal climate forecast information in setting water access rules for a part of the irrigation water supply. Both economic and environmental outcomes, in line with the aims of the post-1995 water reform process in Australia (ARMCANZ 1995), are assessed.

This chapter reviews the drivers of water reform and identifies the underlying environmental and policy problems. A brief explanation of the mechanisms and aims of the policy response and challenges follow with an introduction of the rationale for the use of seasonal climate forecast information as one policy response. Finally, the research aims and hypothesis are identified.

1.1. The Origins of the Policy Problem

Until the early 1990s, the intention of water policy in Australia was to provide cheap and abundant water to nearly all users. The development of infrastructure for irrigated agriculture, the largest sectoral user of water by 79 percent of the volume stored (Schofield et al. 2003, p. 8), was seen as the means to manage the variable climate and the resulting intermittent water supplies (Tisdell et al. 2002). Water storage infrastructure (e.g. dams, weirs) provided a reliable water supply, thus ensuring more consistent and higher agricultural yields. The consequence was higher and more stable incomes for agricultural producers, which had subsequent positive economic effects for these regions.

The main beneficiaries of this policy have been farmers by enabling them to convert from dryland to irrigated farming with flow-on benefits for input industries and output-processing industries (Godden 1997). The cheap water policy ignored the willingness-to-pay of users and was justified (implicitly) on market failure and regional development grounds (Godden 1997; Productivity Commission 1999; Tisdell et al. 2002). There is however, increasing competition for water from domestic suppliers, manufacturing, mining and the government, acting as an ‘agent’ for the natural environment.

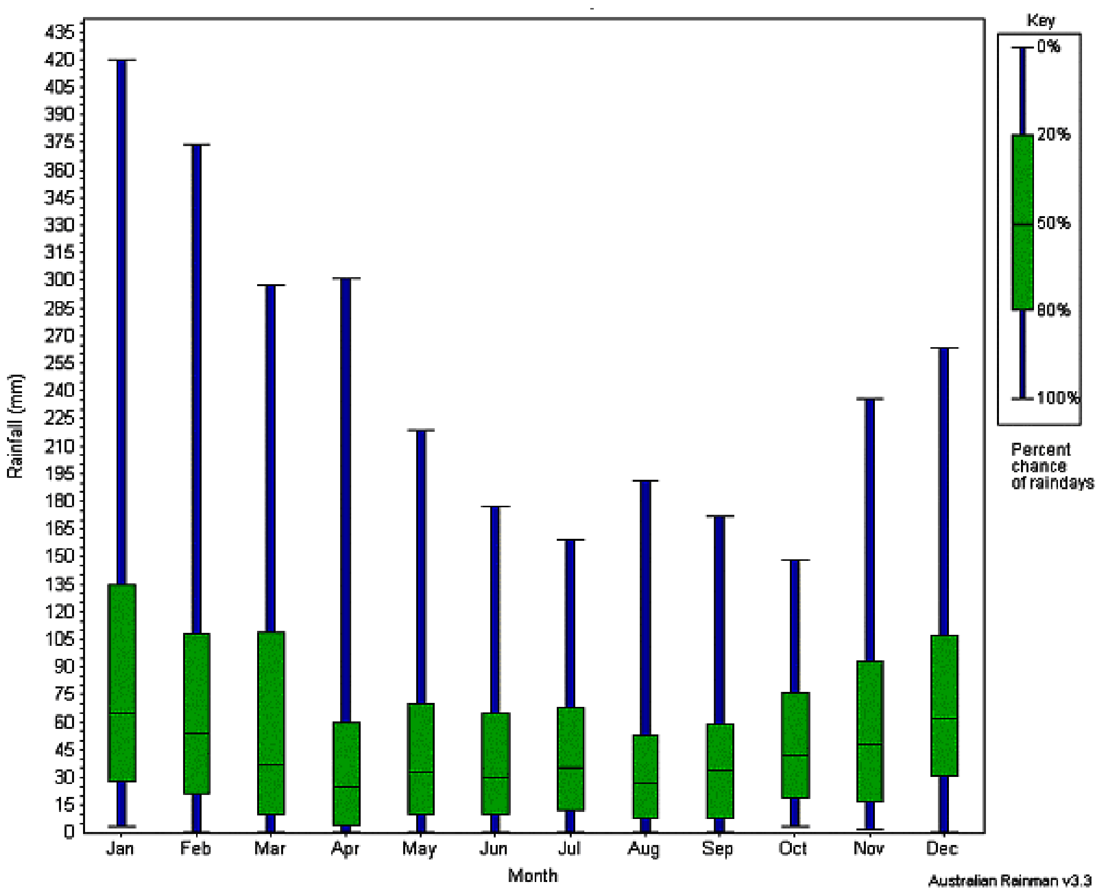
Water is managed at a range of scales – basin, catchment, scheme, river reach and farm level - by a multitude of organisations and individuals. Of the 24,909 gigalitres of water consumed by the Australian economy in 2000-2001, agriculture was overwhelmingly the largest user consuming 66.9 percent or 16,660 gigalitres. Other users included the household sector at 8.8 percent, water supply, sewerage & drainage services at 7.2 percent, the electricity and gas supply industry at 6.8 percent, manufacturing 3.5 percent, mining at 1.6 percent and a range of smaller users at 5.2 percent (Australian Bureau of Statistics 2004, p.17). With urbanisation, there is demand for secure domestic supplies, while some argue that water might add more net value to mining and manufacturing production than to agriculture (Roberts et al. 2006). The actual and potential trade-offs between consumptive uses are however, not considered in this thesis. The concern here is with the trade-off between water used in irrigated agriculture and that retained for environmental flows.

The policy problem here is how to manage the apparent over allocation of water for consumptive users, in this case irrigators, which has led to environmental degradation (Quiggin 2001; Roberts et al. 2006). Returning more water to river systems may improve environmental outcomes, but it could also reduce national and regional outputs. All of the potential trade-offs are brought into sharp relief by the prospect of an overall reduction in supply due to climate change (Murray-Darling Basin Authority 2009). The overarching mechanism for the management of water is government allocation amongst competing users. Under the Australian constitution, the allocation of water resources is the responsibility of state governments (The Australian Government 2005). This was clear, for example in the Victorian *Irrigation Act* of 1886 which defined control of water by the state and led to the centralised role of state authorities in the allocation of water (Tisdell et al. 2002). While in practical terms, as noted by Lloyd and Howell (1993), management of water is influenced by all three levels of government, the final allocation amongst users is made by the states.

States' allocations have historically been carried out in a largely administrative ad hoc manner, on a first-come, first-serve basis with little allowance made for the needs of the environment (Johnson and Rix 1993). Indeed, the role of state governments in water management has tended to be one of infrastructure developer and owner/operator of large-scale urban and rural supply schemes (including irrigation) (Tisdell et al. 2002). When combined with pricing structures that have not reflected the scarcity and true value of water, there has been over-allocation of water to consumptive uses. The results of this are now becoming evident.

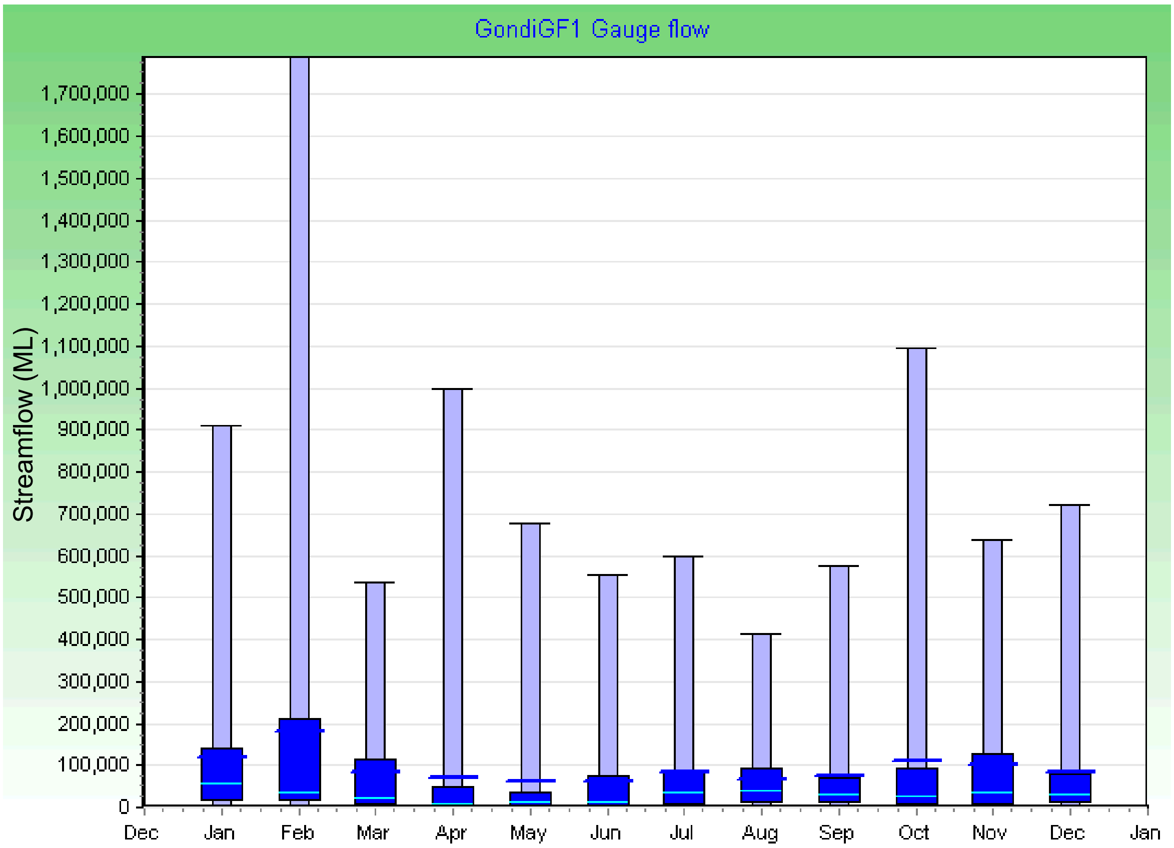
Australia is the driest inhabited continent and has amongst the highest variability, both spatially and temporally, of rainfall and streamflow (Arthington et al. 2003; Chiew et al. 1998). Finlayson and McMahon (1991) showed that Australia has an annual streamflow coefficient of variation of more than 70 percent considerably more than most other continental areas. North Africa has 31 percent, Europe 29 percent, North America 35 percent and, the exception is southern Africa at 78 percent. These characteristics are one of the reasons behind the high level of water infrastructure development, which has led to Australia having the world's highest per capita water storage capacity (Arthington et al. 2003). Despite this infrastructure development, Australia's water supply continues to be variable, as is production from the major user of water, irrigated agriculture (Podbury et al. 1998).

A simple example of the variability in rainfall for Goondiwindi, located in the centre of the study area, is given by Figure 1.1 which displays the variability of rainfall around the 50 percentile or median. For instance in January the median monthly rainfall over the historical record 1879-1998 is 65 millimetres. The box plot highlights that in 80 percent of years at least 28 millimetres of rain is received while in 20 percent of years 135 millimetres falls. Of the outliers the lowest on record for the month is 3 millimetres whilst the highest rainfall received on record in January was 420 millimetres. When considering each of the months it is evident that even at the monthly scale there is considerable variation between months over the year and within months. This variability is also reflected in streamflow, as shown by Figure 1.2 which illustrates the box plot for monthly streamflow at Goondiwindi.



(Source: Australian Rainman).

Figure 1.1: Probability of monthly rainfall for Goondiwindi (1879-1998).



(Source: Watershed, http://www.nrw.qld.gov.au/watershed/index.html, site: 416201A Macintyre River at Goondiwindi)

Figure 1.2: Probability of Monthly gauged streamflow of the Macintyre River at Goondiwindi 1950-1993.

In addition to this variability, the impact of future climate change on streamflows and consequently water availability, is a likely complication. Warming of the earth's climate system is expected to lead to reductions in rainfall and further increases in both variability and, potentially, intensity (Allen Consulting Group 2005; New South Wales Government 2007). This impact is not expected to be localised with climate change modelling indicating that severe droughts and floods will occur in many of the major river basins of the world as a result of climate change (Tarlock 2000).

In the Border Rivers catchment, the impact of climate change on water availability has been studied by the CSIRO in the Sustainable Yields Project (New South Wales Government 2007). The study estimated impacts on water availability under a range of scenarios including future climate at current levels of water infrastructure development, as well as future climate and future water infrastructure development. A number of global warming scenarios from low to high warming were also considered. The conclusion of the study was that runoff is more likely to decrease than increase, although a number of scenarios do indicate the potential for an increase in runoff (New South Wales Government 2007).

The most likely estimate (median) climate 2030 scenario indicates a 9 percent reduction in annual runoff with water availability reduced by 10 percent and subsequently a reduction in end-of-system flows by 12 percent and total diversions by 2 percent. Under the more extreme scenarios, runoff ranges from a 28 percent reduction to a 20 percent increase for the high global warming scenario. A 9 percent decrease and 5 percent increase in average annual runoff is projected for the low global warming scenario. In addition to the impacts on runoff and water availability (i.e. volume), the study concluded that changes were also likely for the frequency of events with periods between rainfall events increasing (New South Wales Government 2007).

Consideration of future water supply availability is placed into context by examining the history of water resource development and the current issues. Randall (1981) describes the transition of the Murray-Darling Basin from an expansionary phase to a mature water[1] economy through the 1960s and 1970s. At the same time, there has been a shift in the awareness and values of some sections of society away from developing natural resources at any cost, where the aim was to exploit natural resources. Now the impacts of resource use on the environment are questioned, leading to a shift in the thinking of governments away from 'developmentalism' (Godden 1997).

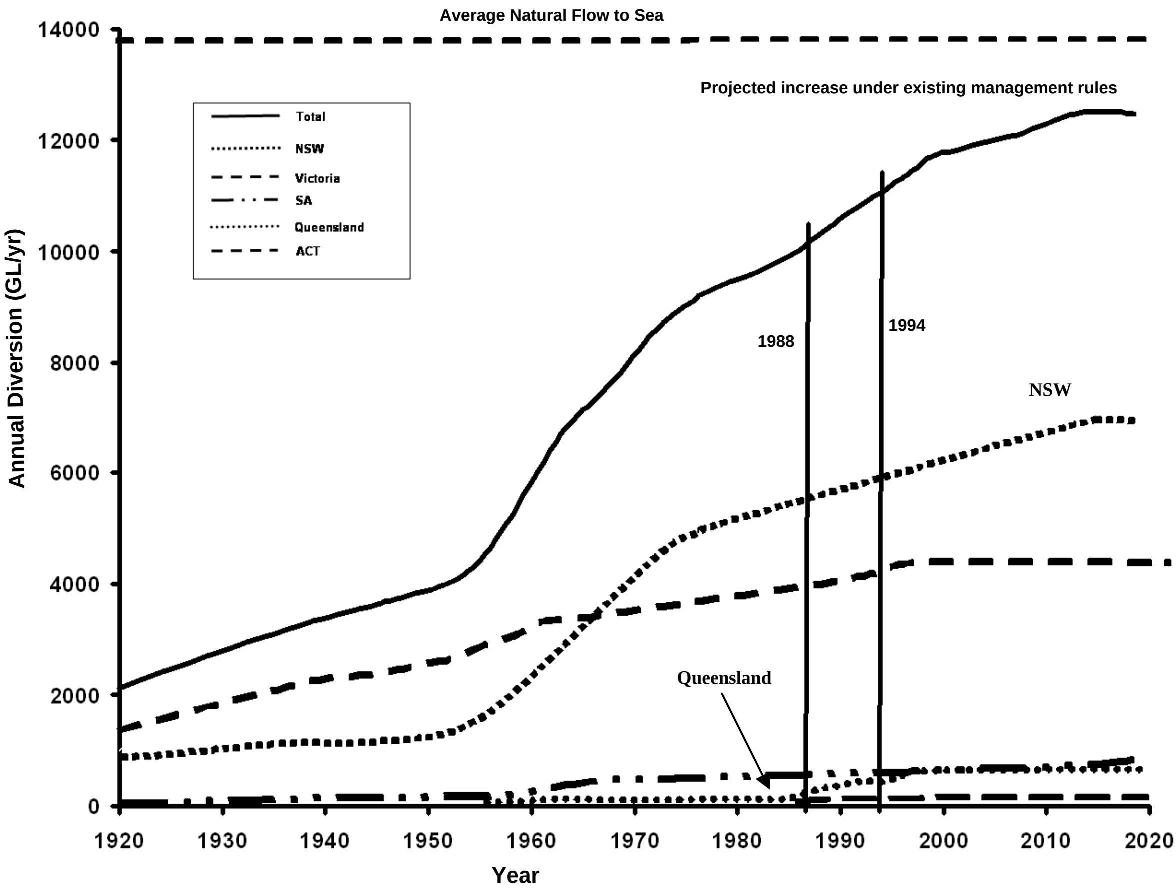
By the 1970s there was general awareness of the pollution and salinity problems in the Murray river and The Murray-Darling Basin which led to a failed inter-governmental agreement on management of the system in 1981 (Walker 1994). The externalities involved, and other mature water economy problems, led to increasing costs to government, through the need to replace aging infrastructure. In an environment where governments were pressured to reduce public debt, these were catalysts for the signing of the Murray-Darling Basin Agreement in 1987 (MDBMC 1995). Subsequently the Murray-Darling Basin Commission (MDBC) was established in 1988 to provide advice to the cabinet level decision-making body for the Basin, the Murray-Darling Basin Ministerial Council that was created in 1985. The MDBC initially administered three programs which were developed between 1988 and 1993: the Salinity and Drainage Strategy; the Natural Resource Management Strategy; and the Nutrient Management Strategy (Walker 1994). These were followed by the revised Murray-Darling Basin Agreement that was given legal status by the *Murray-Darling Basin Act* in 1993. In parallel with this, research in the late 1980s and early 1990s concluded that both high extraction rates and methods of water use were producing considerable damage to riverine environments (MDBMC 1995).

There is now strong evidence that Australia's riverine environment is becoming increasingly degraded and that a major cause of this is the expansion of water infrastructure development for consumptive purposes, while there is a limited understanding of potential environmental risks (Quiggin 2001; Tisdell et al. 2002; Schofield et al. 2003). This degradation is manifest in a range of physical and aesthetic problems including increased loss of habitat and species, poor water quality and decreases in environmental amenity (Quiggin 2001). In the Murray-Darling Basin, the most productive agricultural area in Australia, diversions have been growing rapidly since 1950 (Figure 1.3). Fostering regional economic development

[1] Quiggin draws on Randall (1981, p.73) defining the expansionary and mature phases of water economies, see Appendix A .

was a key driver for this development (Tisdell et al. 2002; Godden 1997; Productivity Commission 1999).

The significant decline in the river health and the environment was eventually acknowledged. In 1995 the Murray-Darling Basin Ministerial Council audit of water use in the Basin concluded that regulation and water diversions had "reduced the variability and changed the seasonality of flows in some parts of the Basin" (MDBMC 1995, p.4).



(Source: MDBMC 1995)

Figure 1.3: Growth of water diversions in the Murray-Darling Basin.

These changes resulted in a number of environmental problems and symptoms including:

- increases in "drought-like flows[2]" at the mouth of the Murray River from 1 in 20 years to 1 in 6 years;
- salinity of the Murray River at Morgan had increased by 10.5 EC[3] from the 1988 level of 537 EC since 1988;
- a reduction in the frequency of inundation of floodplain wetlands which has negatively affected the ability of these wetlands to filter and recycle nutrients, and to provide breeding areas for native flora and fauna;
- increased potential for blue-green algae outbreaks; and
- decline in native fish populations. For example, by the early 1990s the commercial native fish catch in South Australia had decreased by more than 80 percent since the 1950s (MDBMC 1995).

The conclusion from the MDBC audit report was that the likely impacts of consequential environmental damage would include loss of agricultural productivity, significant social adjustment to this production decline and a significant impact on river ecology and biological diversity.

1.2. Addressing the policy problem

The findings of the 1995 audit galvanised the change in direction of water policy in the 1990s through the Council of Australian Governments (CoAG) and National Competition Policy (NCP) reforms and the Murray-Darling Basin Agreement. In 1995, the Agricultural and Resource Management Council of Australia and New Zealand[4] (ARMCANZ) adopted a strategic reform framework aimed at addressing the sustainability problems of the water industry (ARMCANZ 1995; Quiggin 2001). At the same time, the Murray-Darling Basin Commission began the process of setting an annual cap on diversions in the Basin. The direction of these reforms was reiterated in the development of the National Water Initiative which aims to continue and amend the water policy reforms (Council of Australian Governments 2004).

ARMCANZ (1995, p. i) identified the goal of water reform as achieving the "highest and best value of the limited resource for community benefit whilst ensuring that use of the resource is ecologically sustainable". Highest-valued use was defined as including economic returns from consumptive uses and the value to society from environmental and other in-stream water use. The CoAG, NCP and Murray-Darling Basin Commission reforms, which explicitly and implicitly link to ARMCANZ, have two broad aims:

1. to redress certain historical approaches to water policy and alleviate some of the environmental damage which has resulted largely because of past policy direction; and

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[2] Drought-like flows refer to flows less than 4,600 GL/year of the Murray river over the Barrages (MDBMC 1998).

[3] EC = Electrical conductivity

[4] ARMCANZ was a Ministerial Council set up by the Commonwealth and state Governments to further co-operation and collaboration. It consisted of Australian Commonwealth, state, territories and New Zealand ministers responsible for agriculture, land and water resources, and rural adjustment policy issues, (see: http://www.mincos.gov.au/).

2. put in place a series of policies/procedures/management systems that will facilitate the most efficient use of the water that remains for extractive users[5].

These reforms are encouraging the water industry towards ecologically sustainable water use, combined with maximising the economic value of allocated water for extractive uses.

To achieve these goals there is a need to first understand the relationship between the environmental and economic outcomes from using water for consumptive purposes. This is central to the policy problem as it is generally accepted that increasing use for consumptive[6] purposes has translated into poorer environmental outcomes (Quiggin 2001). There remains however some uncertainty as to the form of this relationship. For example is it linear in that increasing use for consumptive purposes produces an increasing loss of environmental outcomes, i.e. environmental health, which is constant? An alternative may in fact occur where up to a point the use of water for consumptive purposes produces only minor or what may be considered acceptable decreases in environmental outcomes to society. Once a certain point or range of use is reached the decreases in environmental health may increase dramatically and reach a point that society is unwilling to accept. There are of course a considerable number of forms that this relationship may take with the underlying competitive relationship an accepted fact.

Determining the nature and extent of this trade-off relationship is a complex task involving not only knowledge of what environmental health means, what effects it and how to measure impacts on health. In addition mechanisms or methods need to be developed to explore how the relationship between resource use and environmental health changes over different levels of use. While these represent on-going research questions there is a general assumption that a trade-off exists and without attempting to answer the above questions we can make some explorations into the trade-off relationships.

Second to ensure society is able to make decisions on what levels of outcomes are acceptable we must be able to measure these outcomes and assess the costs and benefits of allocating water to production of varying levels of each of these outcomes. Therefore, assessment of the costs and benefits of alternative river management arrangements is critical to the ability of governments to gain acceptance of, and implement, new arrangements. For example the support by stakeholders such as the irrigation community of any new management regime to fulfil the policy aims outlined above will rest on the size of any impacts, such as decreased profits, and whether those impacts can be minimised if they are negative. These costs (negative impacts) and benefits (positive impacts) have a number of components covering the spectrum of social, economic and environmental.

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[5] Water extracted for use from rivers by pumping or gravity channels where the water is used to attain an economic return (definition developed from (National Water Commission 2008) and (Tisdell et al. 2002).

[6] Consumptive use: “Use of water for private benefit consumptive purposes including irrigation, industry, urban and stock and domestic use” (Kollmorgen et al. 2007, p. iv)

Social impacts in themselves relate to a broad range of factors but often are taken to include distributional effects measured using indicators such as employment impacts and social cohesion (United States Environmental Protection Agency 2000; Murray Darling Basin Commission 2004a). Economic costs and benefits to be considered include the costs or benefits to the immediate users of water and the flow on impacts to the regional and broader economy. Methods for assessing the economic costs have been developed over decades and a considerable number of tools and approaches exist, each with strengths and weaknesses and varying levels of complexity. At the most basic level where output from one industry increases without commensurate decreases in output from another this can be taken to represent a benefit to the regional economy.

Environmental costs on the other hand relate to the impacts on riverine health for example changes in riverine habitat and species, water quality and environmental amenity (Quiggin 2001). In considering environmental costs one should consider the concept of ecological sustainability which is clarified in the *National Principles for the Provision of Water for Ecosystems* – "to sustain and where necessary restore ecological processes and biodiversity of water dependent ecosystems" (ANZECC 1996, p.5).

The ability to achieve this and measure that we are achieving this hinges on a common and agreed understanding of the environmental needs of riverine ecosystems. Again you need to understand the relationships before you are able to measure. Currently, the science of identifying environmental needs, and the development of water allocations and flow regimes that maintain or improve riverine health, are in their infancy. Allan and Lovett (1997) stated that insufficient understanding of critical ecological processes and their relationship to flows is an impediment to the development of environmental flow regimes. Arthington et al. (1998) and Land and Water Australia (2009) note that progress has been made but that significant gaps in knowledge and conflicting views on the most appropriate methodologies for developing of environmental flow regimes still remain. The implication of this is that new management rules which are part of the process to achieve acceptable levels of environmental health and ecological sustainability are unlikely in the first instance to be optimal in terms of these outcomes.

Summarising these issues it is clear that achieving the policy goals of ARMCANZ has a number of inherent problems. A useful way of summarising the problems was proposed by Schofield et al. (2003) who categorised a number of socio-economic challenges to improving environmental outcomes from river management as:

1. improving water allocation and water-trading arrangements;
2. assessing the costs and benefits of increasing allocations to the environment;
3. understanding and managing the impacts of reductions in allocations for consumptive water use;
4. developing cost-effective ways to enhance environmental flows; and,
5. improving administrative arrangements for the management of water allocations.

An important consideration in trying to address these challenges is the issue of the trade-off between outcomes from the use of water which is explicit in the ARMCANZ goal and underpins each of these challenges. Often when considering resource use the impact on each of the two outcomes, economic and environmental, are assessed separately and the trade-offs from different levels of use is implied rather than made clear. This research uses the notion of the trade-off and at first seeks to explore how the relationship between economic and environmental outcomes changes as water is moved between producing the two outcomes that are generally considered to be competing. Once the trade-off relationship has been identified a number of seasonal climate forecasting tools are introduced and the outcome of their use is compared to the outcome of the trade-off analysis to ascertain if the information is able to improve either economic or environmental outcomes without negatively impacting on the other.

In undertaking this approach this research tests the use of what might be termed a new technology, seasonal climate forecasting information, in the context of these challenges. The research however is restricted to testing whether or not this new technology can assist in achieving the desired outcomes of the water reform process by assessing the economic and environmental trade offs at an identified regional scale from using the technology to control access to a component of the irrigation water supply which will be explained in later chapters.

To implement the analysis in relation to climate forecasting for better economic and environmental outcomes, I propose a multidisciplinary approach that combines climatic, hydrologic, agronomic, economic and environmental disciplines to evaluate impacts from change to water resource management options. Scenario analysis is used to assess if the use of a new technology can improve water allocation such that production can be maintained while environmental flows are increased. This is can the trade-off effects between economic and environmental outcomes can be minimised.

1.3. The Research Problem

The concept of environmental flow management (which identifies river flow as critical in maintaining river health) has emerged as a central component of new approaches to water management (Tharme 2003). Despite the lack of appropriate scientific knowledge, there remains a need to initially develop management procedures in the short-term to achieve the longer-term goal of ecological sustainability. This was recognised by the *National Principles for the Provision of Water for Ecosystems* noting in regard to managing environmental water that "monitoring is required to ascertain how adequately the objectives of environmental water provisions are being met, and hence to enable adaptive management to be implemented" (ANZECC 1996, p. 15). More recently Arthington et al. (2006) propose that adaptive management can be used in the first instance to set environmental flow targets that are then validated and changed where necessary as new knowledge is developed. This may be achieved using a range of mechanisms including administrative arrangements, water trading and management regimes. Within this range of mechanisms lies the potential to augment management regimes with additional knowledge by forecasting water flows.

Seasonal climate forecasting (SCF) provides information on the hydrologic impact of climate variability which may assist in water management decision-making (Changnon et al. 1986; Chiew et al. 2000). It is now accepted that there is a significant relationship between the El Niño-Southern Oscillation phenomenon

(ENSO) and rainfall and streamflow variability in many parts of Australia, particularly north-eastern Australia (Chiew et al. 1998; Abawi et al. 2001). The influence of the ENSO phenomenon is not restricted to Australia and has been linked with variability in the wider regional and global climate (Latif et al. 1994). The ability to use this knowledge to forecast both rainfall and streamflow is becoming increasingly recognised (Stone et al. 1992; Chiew et al. 2000; Abawi et al. 2001). However, the adoption of this knowledge into water allocation management remains limited.

The uptake of SCF information by water resource managers in general has been extremely limited, and non-existent in the management of environmental flows (Long and McMahon 1996; Hartmann et al. 2002; Callahan et al. 1999). Examination of the impediments to using climate-forecast information for water management suggests that both forecast characteristics and institutional factors are responsible, highlighting a complex interplay between scientific and policy factors (Long and McMahon 1996; Pagano et al. 2001; Hamlet et al. 2002; Hartmann et al. 2002). Opportunities for incorporating seasonal climate forecasting (SCF) into environmental flow management do however exist. The opportunities for SCF arise from the ability to use SCF to forecast the variability of flows in coming periods and to enhance the management of rivers to meet both consumptive and in-stream[7] requirements.

Over time, two factors can lead to changes in the ability to make use of SCF information in water management decision-making: improvements in forecasting ability, and, alterations to water management regimes that permit the use of forecast information in decision-making. The approach of this study is to develop a trade-off analysis based on the concept of the Production Possibility Frontier and the Pareto Principle to test whether certain seasonal climate forecasting tools are able to be used in environmental flow management decision-making and produce desirable outcomes. This approach allows for the analysis of changes in the two variables, where only one can be easily measured, in this case consumptive values.

1.4. Hypotheses

Based on the policy and research issues outlined above, this research will test two primary hypotheses nested within the aims outlined below.

Aim 1: Develop the trade-off methodology and modelling framework for examining the impact of applying three ENSO based SCF tools to flow management on both the regional economy and the health of the riverine environment of the Border Rivers Catchment (BRC).

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[7] In-stream use: “The use of freshwater in situ (for example, within a river or stream). Can include recreation, tourism, scientific and cultural uses, ecosystem maintenance, hydroelectricity and commercial activities, and dilution of waste. The volume of water required for most in-stream uses cannot be quantified, with the exception of hydro-electricity generation” (Kollmorgen et al. 2007, p. v) (National Water Commission 2008). In this study in-stream use is restricted to ecosystem maintenance.

Aim 2: Evaluate the efficacy of using three ENSO based SCF tools in environmental flow management in the Border Rivers Catchment.

- *Hypothesis 1: The use of any of the three seasonal climate forecasting tools based on the El Niño-Southern Oscillation phenomenon to manage off-allocation water access by irrigators will increase the regional economic output of irrigated agriculture and/or produce conditions that will lead to improved riverine environmental health in the case study catchment.*

- *Hypothesis 2: An improvement in forecasting accuracy will increase the value of crop production and environmental flows to levels above those prevailing under the use of the identified forecast tools.*

The testing of these hypotheses will be undertaken within a framework that examines the trade-off between economic and environmental outcomes that are specifically defined in chapter 4. The methodology also includes consideration of sensitivity analysis of results for key parameters.

1.5. The study site

The Border Rivers Catchment (BRC) in the northern part of the Murray-Darling Basin was selected as the site for this study because it fulfilled a number of requirements that make the study possible. As noted in the introduction, a key aspect of the study is the consideration of the economic and environmental impacts of using SCF information in water resources management. The study site therefore needs to be one that has the requisite information to allow this analysis to be undertaken.

In the first instance there needed to be a hydrologic model available that can be configured to test the use of SCF in decision-making. As the BRC is a catchment where water planning activities in NSW and Queensland have been underway, such a model has been developed. In addition, to minimise the need for an overly complex economic assessment model which has to account for a large number of industries and usage methods, a catchment where the major water user is one industry, dominated by one crop, was preferred. In the BRC, irrigated agriculture accounts for some 98 percent of water diverted (see Table 3.2), irrigated cotton is the largest single irrigated crop grown and it accounts for the majority of irrigated water used. Finally water planning activities undertaken in the BRC have included the consideration of the impacts of water use on environmental outcomes. This facilitates the assessment of environmental impacts from using SCF information. The study site is more fully discussed in chapter 3 of the thesis.

1.6. Thesis structure

The remainder of the thesis is split into six chapters covering the literature review, study area, methodology, results, discussion and conclusions. In the literature review (chapter 2) the implications of the need to consider economic and environmental impacts in regard to the study aims are considered. Approaches to assessing impacts on these aspects separately and jointly are subsequently canvassed. Finally the use of SCF information in the management of agricultural and water resources is assessed. The literature review, combined with the information on the study area (chapter 3), provides the basis for the development of the methodology (chapter 4). This begins with an overview of the modelling framework followed by details of the individual model components. The final sections of chapter 4 cover the integration of the model components into the modelling framework and the development of the scenarios tested in the research.

Presentation of the results (chapter 5) is in three sections. First, results of the non-SCF scenarios are reported and second, the validity of the model against expected results is assessed. The final section covers the results of the scenarios incorporating the SCF information in decision-making. The results are then discussed in relation to their inferences for the study aims and hypotheses (chapter 6). The implications of the results for adoption by water resource managers are considered and steps required to overcome the barriers to adoption identified. The final chapter (7) concludes the thesis by summarising the contributions to the body of knowledge, identifying key implications of the methodology on results, and suggestions for future research that would improve the robustness of results.

1.7. Conclusion

Achieving the goals of water reform in Australia involves a large number of challenges and will require a considerable range of actions to be undertaken by players ranging from different levels of government to irrigators. This research seeks to test whether the use of ENSO based seasonal climate forecasting information which to date is treated as periphery information rather than necessary information in water resources decision-making, can play a small part and assist in achieving these goals. I test the use of ENSO based SCF information to ascertain whether either environmental or economic improvements can be made without leading to a decline of either of these outcomes. As these outcomes are typically seen as being competitive in nature, i.e. an improvement in one outcome is expected to lead to a decrease in the other, the analysis undertaken focuses on assessing this trade-off and how the use of SCF information impacts on the trade-off. Because forecasting ability or skill in relation to streamflow varies spatially it is necessary to conduct this test in one location, the Border Rivers catchment, using a methodology that can be transferred to other locations to conduct similar tests. In testing the hypothesis if the use of SCF information improves the indicators of riverine environmental health without negatively effecting irrigated production or improves irrigated production without negatively effecting the indicators of riverine environmental health the SCF information will be deemed to be efficacious and may assist in overcoming some of the barriers to the use of this information in water resources decision-making.

2. Literature Review

Recognition that the continuing increase in the development of water resources to increase production of irrigated agriculture has produced increasing environmental degradation has lead to changes in the way water is to be managed in future. In particular the emphasis has become one of improving water allocation and management regimes to minimize degradation. A key aspect of this is the acknowledgment that the environment is a legitimate user of water and that environmental outcomes from water management need to be measured to assist in improving management regimes. Considerable challenges remain in implementing arrangements that give effect to this recognition.

This chapter examines the issues involved in the implementation of water management plans using forecast information for allocation decision-making, potentially leading to improved environmental flow outcomes. A preview of the literature suggests a model whereby the decision process involves a consideration of trade-offs amongst competing users. However, there are a number of methodological problems to assessing these trade-offs, most notably defining environmental goals and valuing environmental impacts. To overcome these constraints, I adopt an approach where the environmental outcomes are assessed using methods consistent with current water planning and management arrangements and incorporate this into a framework for assessing the trade-off between environmental and economic outcomes from water use in irrigation. I briefly examine the development of seasonal climate forecasting in Australia and its potential for use in water resources management. While this potential is acknowledged in other research, so too is the lack of adoption of this information in decision-making. Leading into the following chapter where the methodology is developed, I consider the scale of the modelling approach to be taken which ensures the study objectives are met, including the need for a flexible or rapid assessment method and explore barriers to systematic use of the assessment information. This approach will assist in overcoming at least some of the barriers to adoption, either through this research or in future periods, when improvements are made to current forecasting tools or new forecasting methods arise.

2.1. Water Allocation: Assessing the trade-offs

The 1994 Council of Australian Governments (CoAG) and later Natural Water Initiative (NWI) agreement (National Water Commission 2009), and other decisions since the mid-1990s, seek to have the environment recognised as a legitimate user of water. States, which have the primary power for allocating water, are in the process of providing water allocations (where allocation has a broad definition) to gain environmental outcomes (Arthington et al. 2003). As noted in the ARMCANZ agreement, the goal of water reform is to achieve the "highest and best value of the limited resource for community benefit whilst ensuring that use of the resource is ecologically sustainable" (ARMCANZ 1995, p. i). Highest-value use is defined to include economic returns from consumptive uses and the value to society from environmental and other in-stream water use. One outcome from the pursuit of this goal is conflict between consumptive users and governments (and others), who press for re-allocation of water for environmental (in-stream) purposes because of the apparent trade-off between these uses. The potential for trade-offs between environmental and economic outcomes is implicitly recognised in the ARMCANZ goal of water reform (van der Lee and Gill 1999). This situation poses a quandary for policy-makers and analysts in determining what framework and tools to use when making water allocation and management decisions.

If we assume there are two broad uses of water resources: consumptive and in-stream uses, the allocation (i.e. trade-off) problem can be presented using a Production possibility frontier (PPF). The PPF displays the combinations of outputs possible for two goods given a particular state of technology and available resources. The PPF shows what levels of output are attainable and unattainable and draws on insights into the relationship between the two outcomes (Baumol 1994; Parkin 1990). A conceptual model based on this trade-off using the PPF will be developed and used through the thesis as an assessment framework to test the research hypotheses.

2.1.1. Modelling Trade-offs: the Production Possibilities Frontier

The Production possibility frontier (PPF) stems from David Ricardo's work explaining the theory of comparative advantage. In more recent times it has been used in the context of multi-functionality in relation to agri-envionmental policy making where it is recognised that producing agricultural output has both positive and negative effects on the environment (Vatn 2001). The term multi-functionality refers to the fact that an activity can have multiple outputs and therefore may contribute to several objectives at once" (Abler 2004, p.8). Multi-functionality is described by Vatn (2001) and Harvey (2003) using a production possibility framework where the relationship between agricultural and environmental output can be joint, complementary and competitive. The PPF shows the tradeoffs between production of two or more goods by examining how the levels of output change when a fixed set of inputs or resources are transferred from producing one good to another. In depicting this trade-off, the PPF also shows how much production can be attained at different levels of inputs or resources (Doll and Orazem 1984; Parkin 1990).

The simple form of a PPF can be represented by a straight line trade-off (Figure 2.1) for two goods, or in this case outcomes; in-stream and consumptive use benefits. In Figure 2.1 for example a_{max} might represent some environmental physical output such as numbers of a species of fish and b_{max} may represent bales of cotton from irrigated production. If all water is used to produce in-stream benefits, then an outcome of a_{max} will be attained with no consumptive benefits. Alternatively, if all resources were put towards consumptive uses outcome b_{max} will result in no in-stream benefits. Along the line $a_{max}b_{max}$, the "frontier", a number of combinations of outcomes are possible, as well as any combination inside that line Moving from C to D to E shows different trade-offs between the two outcomes.

The trade-offs are the result of the physical relationships, not the value of the outcomes. That is, changing the use of the resource (in this case water), to produce more of one type of outcome or output, will result in less of the other being produced. As production moves between the levels C, D or E, in Figure 2.1, the trade-off of in-stream benefits forgone to produce more consumptive outcomes is constant because of the linear relationship. Trading-off from C to D results in the loss of Δa units of in-stream benefits to gain an increase of consumptive use benefits equal to Δb. Therefore the trade-off, called the Marginal Rate of Transformation (MRT) is equal to the ratio of Δa and Δb. A further move to E where $\Delta a = \Delta a^1$ will result in a gain of

Δb^1 of outcome B. As Δb^1 is equal to Δb, the trade-off between the two outcomes is constant.

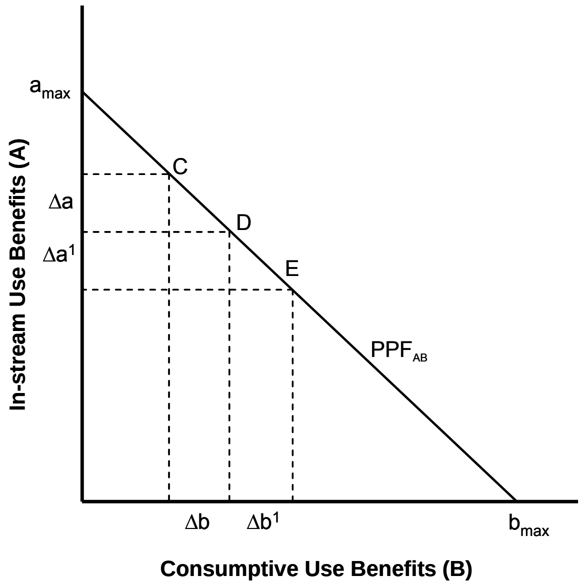


Figure 2.1: Straight line production possibility frontier.

Linear trade-off relationships in production or natural systems would be unusual. Conventionally, the PPF is drawn not as a straight line but as a convex curve as in Figure 2.2 where the MRT is increasing, reflecting increasing opportunity cost, the level of which is dependent on the slope of the curve at any particular point. The increase in opportunity cost is because the productive resource, water in this case, does not have constant productivity in producing the outcomes (Parkin 1990; Dudley 1997). To illustrate the increasing MRT, in Figure 2.2 moving from C to D to E, $\Delta a = \Delta a^1$ however $\Delta b > \Delta b^1$ and so $\frac{\Delta a}{\Delta b} < \frac{\Delta a1}{\Delta b1}$, including an increasing opportunity cost of in-stream benefits to attain higher levels of consumptive benefits (moving from left to right on the curve).

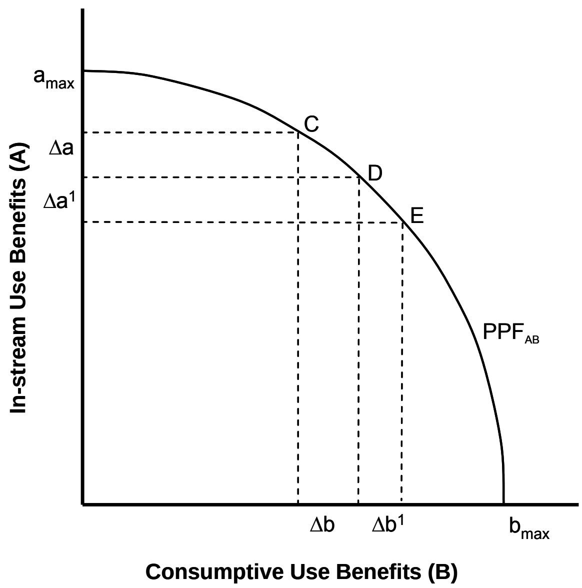


Figure 2.2: A convex PPF displays a non-constant trade-off or MRT.

2.1.2. Production possibilities frontier - consumptive and in-stream outcomes

The PPF is becoming more commonly used as a tool to assess and understand the tradeoffs between outcomes in the area of environmental and natural resource management and to inform policy decision-making (Calkin et al. 2002; Calkin et al. 2005; Scott et al. 1998). These studies are taking the standard PPF in a new direction by moving from what can be termed production - production space where the analysis assesses the trade-off between two products to an analysis that assesses the trade-off between one product in physical terms and another in economic or other non-physical terms.

Calkin et al. (2002) combine biological and economic models to simulate the PPF between timber production and the likelihood of persistence of a wildlife species. They note that the increasing requirement for resource managers to meet multiple, and often conflicting economic and ecological goals, is a key driver of trade-off analysis.

Groeneveld et al. (2005) use a similar approach to assess the trade-off between conservation outcomes and agricultural income in the context of European agri-environmental schemes that compensate farmers for undertaking voluntary conservation measures. This study developed typically convex PPFs showing a competitive relationship existed whereby more monetary benefits to farmers could be attained by decreasing the expected area of occupied habitat and visa versa. In comparing PPFs for different policy options, the study concluded that in some cases a change in policy could lead to improved environmental outcomes without negative outcomes to agriculture. This finding is congruent with the convex curve simulated by Calkin et al. (2002) which highlighted the dollar opportunity cost for likelihood of species persistence.

Scott et al. (1998) assessed the trade-off between irrigation use and the environment using a PPF identifying the change in shape of the PPF as a point where the trade-off changed. They identified a kink or threshold in the PPF at which the trade-off or opportunity cost of changes to allocation amongst the competing environmental and consumptive uses was high. At this point allocating more water to irrigation uses resulted in small gains to irrigators at a high cost to the environment and visa versa.

The trade-off depicted in Figure 2.3 extends the PPF concept to the general problem that the water reform agenda is dealing with. This concept is used by Dudley (1997) and Dudley et al. (1998) and stems from the theory of a production possibility frontier marking the boundary between production levels that can and cannot be attained. Dudley et al. (1997) describe the trade-off such that the x axis denotes increasing discounted benefits to consumptive uses (in dollar terms) of water over time. The y axis shows increasing benefits obtained from environmental uses which may be measured in dollar or non-dollar terms. If water were not a limiting factor, the amount of 'production' would settle at the zero trade-off point providing maximum benefits for each use (Dudley 1997). However, within the existing water-limited environment maximum benefits are unattainable.

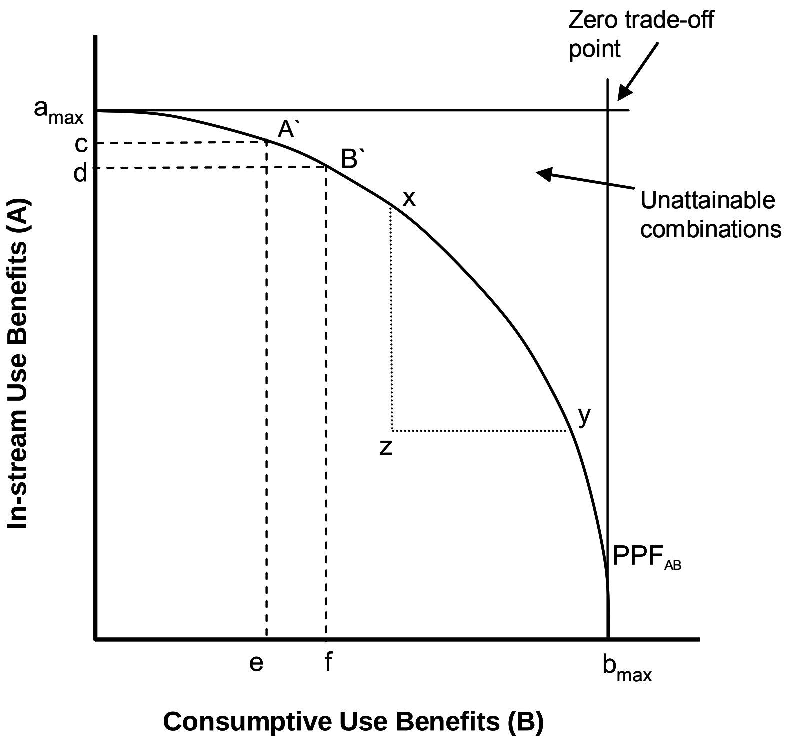


Figure 2.3: Trade-offs between environmental and in-stream benefits of water use.

When using the trade-off curve as described in Figure 2.3 inputs to production are held fixed including capital items such as on-farm water storages. The trade-off between the two outcomes is then driven by changing water allocated to the production of one output to the other. This is achieved by changing the management of water which drives or allocates water from production of consumptive benefits to production of in-stream benefits. The use of non-physical measures of output, such as monetary values, is justified because other factors such as prices remain fixed. Therefore the change in non-physical measures is driven by the change in physical output.

Points inside the curve $a_{max}b_{max}$ in Figure 2.3 (e.g. z) represent inefficient resource use (Dudley 1997; Dudley et al. 1998). That is, in this case the fixed resources could be better used to get one or more outcomes, nearer to, or preferably on the frontier From z, an increase in overall benefit to one or both users is available by moving towards the boundary line on or within the direction of the north-east quadrant. The position taken by Dudley et al. (1998) is largely based on the premise that the allocation between users is currently on the boundary $a_{max}b_{max}$. Assuming (as is conventional) that technological and management variables are held constant points on the boundary represent maximum efficiency in allocation of resources.

In keeping with this convention the PPF or trade-off curve for this study is defined as full technical efficiency, full use of available management skills, full availability and use of information (Aldy et al. 1998). As will be pointed out in a later section, this information does not include seasonal climate forecasting, which if used and found to be efficacious could result in a change to the PPF and potentially a change to the trade-off between outcomes. An efficacious use of SCF information in making water management decisions represents an improvement in technology and knowledge which could have the effect of moving the frontier outwards such as in Figure 2.4. In this case the outward movement in the frontier leads to the potential for a change in outcomes from C to E and the trade-offs as represented by the MRTs do not change because the ratio of Δa and Δb remains the same at each vector.

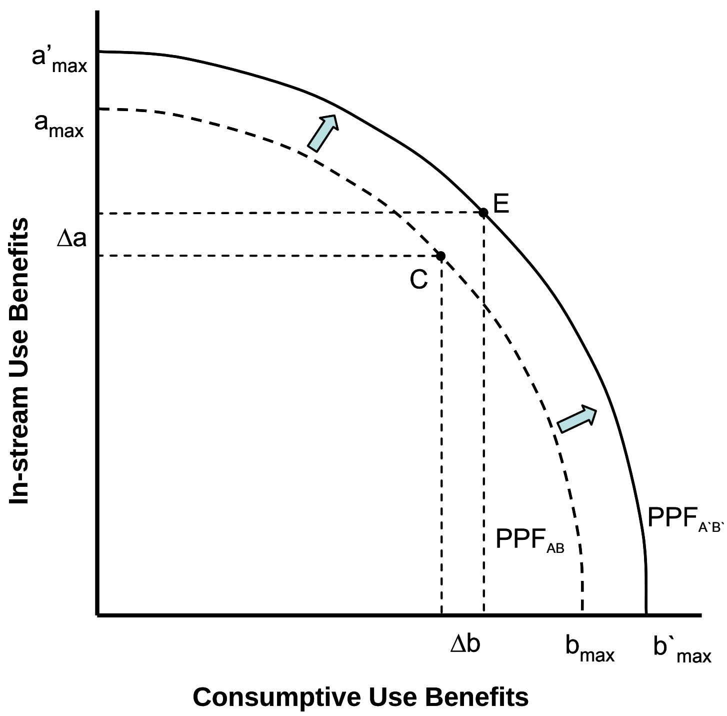


Figure 2.4: The effect of an improvement in management ability from SCF information.

An improvement in technology however may not necessarily lead to an even movement in the frontier. Depending on the relative effect on the physical relationship between the input and outcome, the result may be more of one outcome being produced (point E in Figure 2.5) or full specialisation with no change to the other (point F in Figure 2.5). For the same change in Figure 2.4 and Figure 2.5 Δb will be greater for the latter.

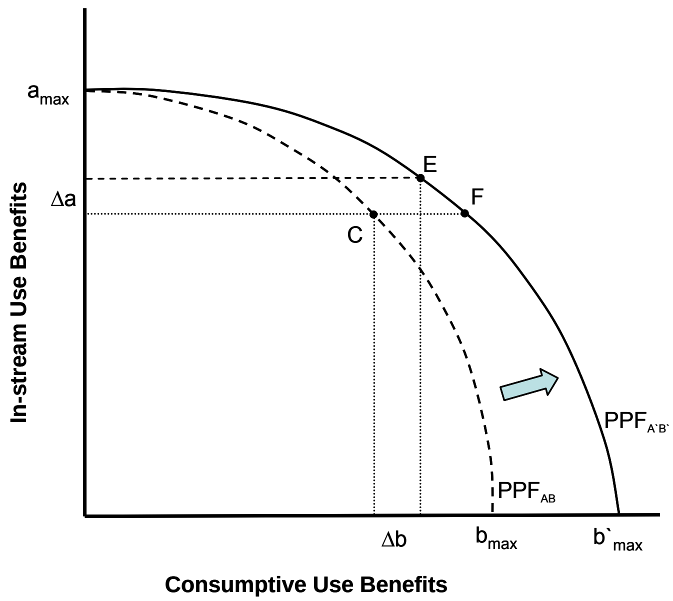


Figure 2.5: Impact of technological improvements on the PPF.

The thesis hypothesis represented diagrammatically is that the use of SCF information will result in a movement of the frontier outwards. For example, if we assume that Point C (Figure 2.6) represents the current allocation between competing uses, then the successful use of SCF could result in a move from C towards x. Points along this line (Cx) represent an improvement in environmental outcomes without any negative shift in consumptive use benefits. Alternatively, a move from C to y would also be a positive outcome from society's perspective, with a gain for consumptive uses without environmental costs. It should also be noted that any move from point C toward the north-east quadrant, such as E, represents an improvement in both types of outcomes.

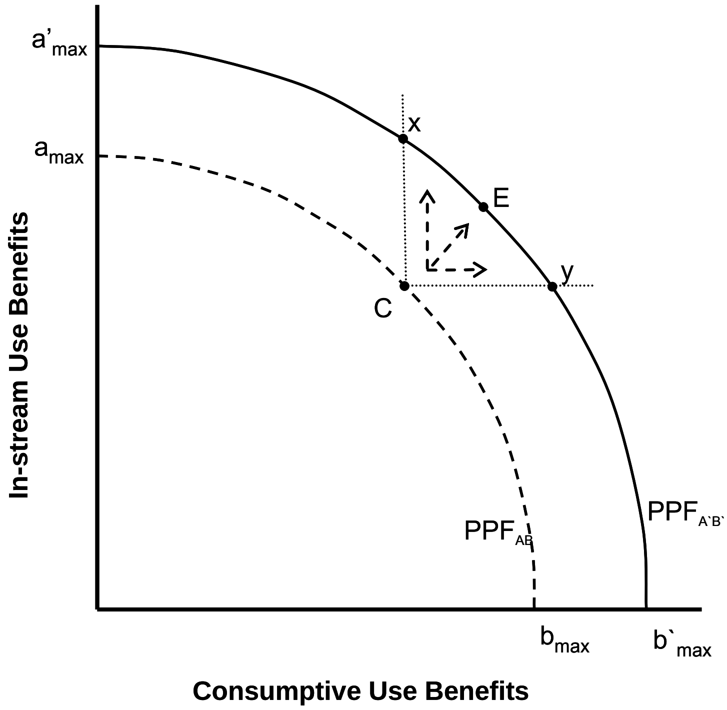


Figure 2.6: Trade-offs between environmental and in-stream benefits of water use.

2.1.3. Refining the PPF using the ARMCANZ goals

The absolute position of point C in Figure 2.6, while difficult to know with any certainty, is nonetheless important when determining whether the direction of movement from point C is such that positive outcomes are achieved or whether there are restrictions on levels of outcomes below which negative outcomes will not be acceptable for each set of users. Dudley (1997) assumed that constraints may exist restricting the potential area of trade-off available in a decision framework. Following Dudley (1997), the horizontal line *bb* in Figure 2.7 represents the minimum level of environmental benefit that society is willing to accept i.e. any allocation that results in a level of environmental benefit below *bb* will not be permitted. This level could be set by the environmental regulations that govern how water is managed. Implementation of the water reform process is a tacit recognition that point C is either very close to *bb* or may have dipped below *bb*. A constraint line (represented by line *aa* in Figure 2.7) also exists for consumptive uses. The absolute value of the consumptive use constraint is unknown but could be identified if water planning processes undertake economic impact analysis on the adoption of new water sharing or allocation rules.

In the various water resource and water sharing plans (DNRM 2003) a series of environmental flow objectives or rules have been specified which in effect represent a level against which environmental flow outcomes will be measured for acceptability. Given that the pressure for water reform has come primarily from environmental aspects, we can be certain that the current allocation arrangements result in point C being to the right of line aa. The outcome from the existence of lines aa and bb is that the acceptable trade-off area could potentially be wider than xy. However, for the purposes of this study it is assumed that line Cx tracks the same as aa and bb similarly can be represented by Cy.

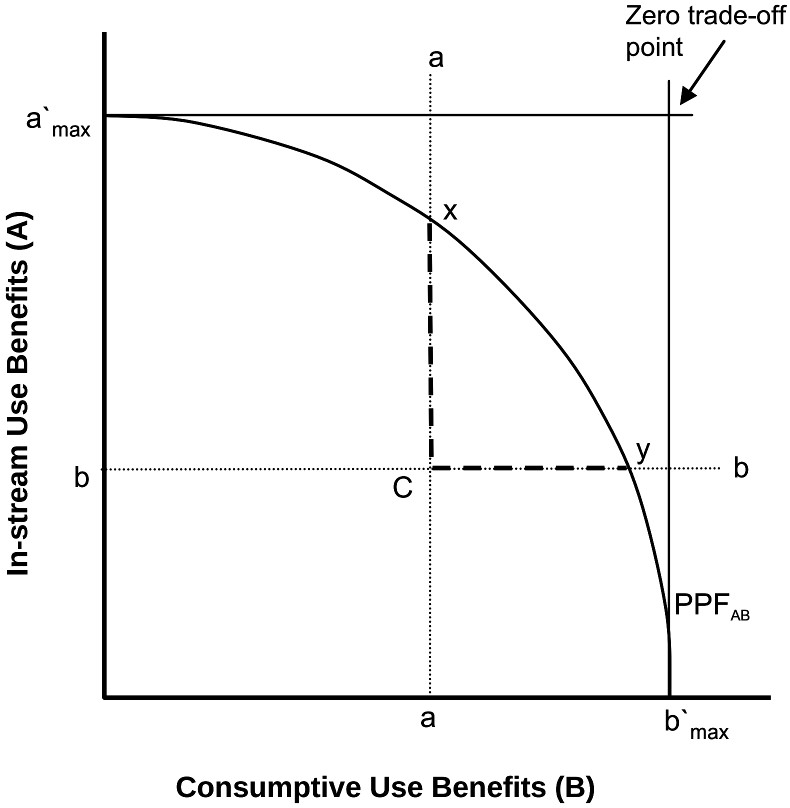
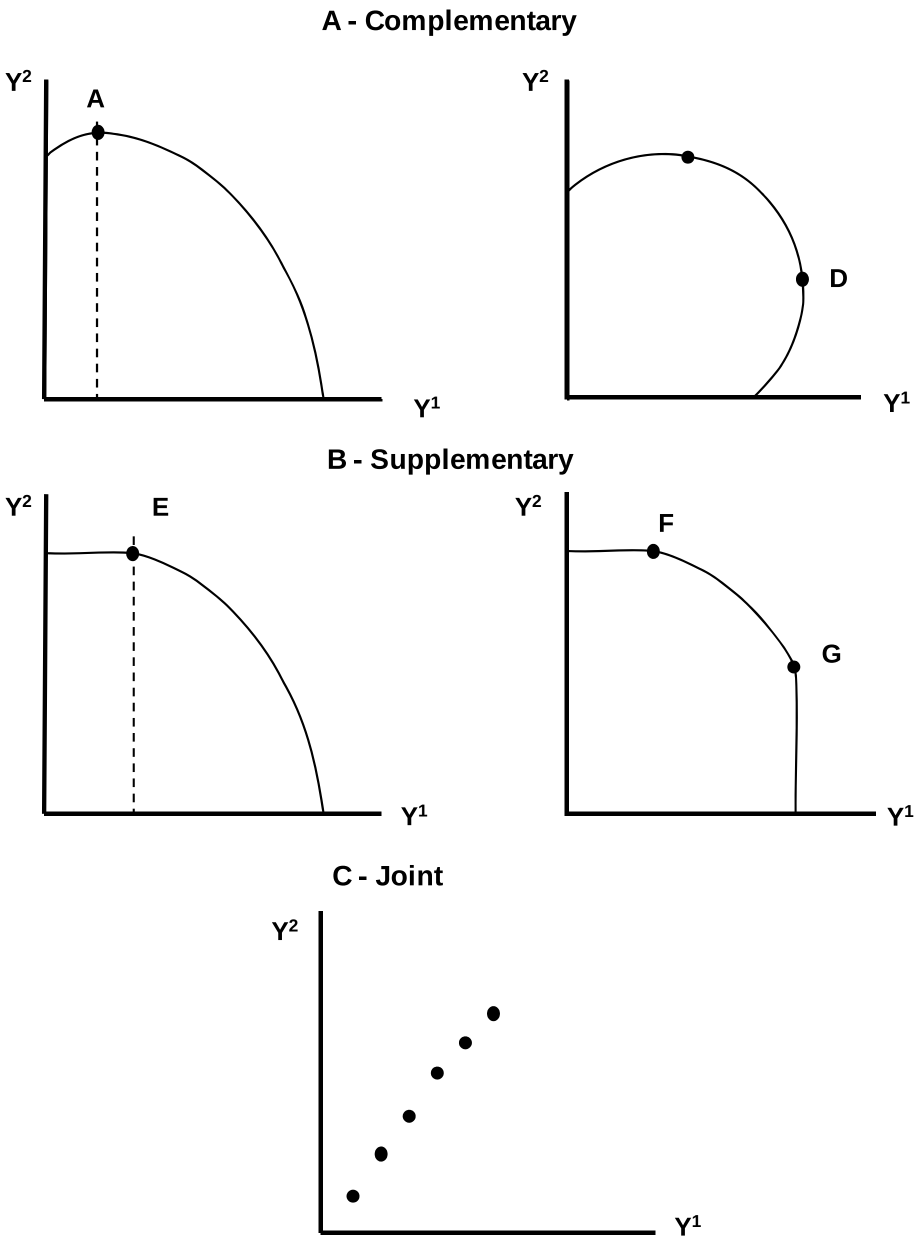


Figure 2.7: Constraints on minimum acceptable levels and direction of movement.

2.1.4. Implications of the shape of the PPF and outcome relationships

The shape of the PPF is an important assumption to make. In addition to the typical convex increasing MRT curve displaying competitive relationships (Figure 2.7), Doll and Orazem (1984), Abler (2004), Vatn (2001) and Harvey (2003) introduce a range of different curves in terms of the relationship between two products (Figure 2.8). They identify complementary, supplementary and joint products. Furthermore, they suggest that two goods may display a range of relationships over the one curve where some levels of production result in the existence of a complementary relationship and other levels where a competitive relationship occurs. In the area of natural resource management Wiggering et al. (2006), referring to environmental and productive trade-offs, noted that the shape of the PPF is influenced by the degree of joint production, technology and outcome type. For example, a case of a joint production relationship might occur when considering the relationship between revegetating land to forest for habitat and carbon sequestration. When there is no forest there is no sequestration, however as more forest is planted and habitat increases, so does carbon sequestration.

The complementary relationship, where the production of one product leads to the production of another, is also postulated as being possible by Abler (2004), Vatn (2001), Harvey (2003) and Wiggering et al. (2006). If this relationship is displayed, Doll and Orazem (1984) remark that complementary products must eventually become competitive when large proportions of an input are devoted towards the production of one product. The supplementary relationship, where the amount of one product or outcome can be increased without decreasing the production of another product, might be considered a specialised case of the complementary relationship. Calkin et al. (2005) relates joint production relationships to the compatibility of objectives, commenting that PPF analysis can be used to explore if compatibility and therefore joint production relationships, are feasible. They found compatibility and conflict between two objectives, fire threat reduction and late-seral forest structure (LSF), when LSF is varied as a proportion of total reserve area. The presence of these varied and changeable relationships between variables or outcomes add weight to the argument for building an understanding of trade-offs in resource use.



(Source: Doll & Orazem (1984)

Figure 2.8: The range of shapes production possibility curves may assume depending on differences in the relationships between outcomes.

The relationships identified by Doll and Orazem (1984), Abler (2004), Vatn (2001), Harvey (2003) and Wiggering et al. (2006) might be applied to ecological thresholds. Huggett (2005) listed a number of definitions of ecological thresholds, but suggested that the most practical was by Bennett and Radford (2003, p. 1) who proposed that "ecological thresholds are points or zones at which relatively rapid change occurs from one ecological condition to another". The existence of thresholds remains an area of ongoing research limited by available empirical data (Walker et al. 2004). However, both Huggett (2005) and Walker and Meyers (2004) identify a number of studies where the findings suggest that thresholds do exist. For example, Hanski et al. 1996, With and King 1999 and Fahrig 2001, 2002 cited in Huggett (2005) suggest that crossing a threshold of habitat loss may alter the probability of species extinction from near zero to near one.

The implication of the existence of thresholds for this study is that changing the mix of water use from in-stream to consumptive use may at some point breach a threshold whereby an alternative state is reached that is difficult, or unable to be reversed. It should be noted that in addition to the lack of empirical studies, there are also a number of uncertainties around thresholds. Huggett (2005) and Walker and Meyers (2004) refer to some uncertainties which may have implications for this study. First, a degree of uncertainty around threshold values and magnitudes, i.e. the threshold may be a range rather than a point; second, there may be more than one causal factor determining the threshold value; third, the threshold may differ between different landscape scales; and finally the response to a threshold being crossed may vary from quite short to long and may also be variable.

Assuming the existence of ecological thresholds, some inferences can be made about the shapes of the frontiers proposed by Doll and Orazem (1984) and Wiggering et al. (2006). For example, the second complementary relationship of Figure 2.8 shows a point D whereby continuing to divert say water from the production of Y2 to Y1 not only results in a steep decrease in production of Y2 but also a decrease in production of Y1. This relationship implies that D may be a threshold beyond which the response of the production system in producing outcomes Y1 and Y2 changes dramatically. Points A, E, F, and G also imply threshold points. The existence of threshold points and the hypothesised potential outcome make their identification a key aspect to consider in this study as it may impact on the ability for SCF information to be used in a policy or water management context.

Setting aside the issues of constraints in Figure 2.7 and the different product – outcome relationships of Figure 2.8, the water trade-off decision depicted in Figure 2.6 is used as the basis for developing the model until or unless thresholds are identified. It is clear from Figure 2.6 in relation to the water trade-off decision that the best outcome from the perspective of society in general is for any changes to result in movements from point *C* towards the frontier within the north-east quadrant. In an economic decision-making framework, seeking an ordering of different states or outcomes from policy changes where one outcome (and therefore allocation of resources) is preferred by society over another, falls into the sphere of social welfare (Boadway and Bruce 1984). The specification by ARMCANZ (1995, p. i) that the goal of the water reform process is to "obtain the highest valued use ….to society" introduces the notion of social welfare, hence moving closer to the possibility frontier.

Social welfare can be characterised as an aggregation of the utility of individual members in society (Weimer and Vining 1989), where utility can be thought of as the satisfaction or well-being consumers receive from a good or service (Parkin 1990). Welfare economics is concerned with analysing the desirability of changes and policies by providing a ranking mechanism and evaluating the consequences of change (Johansson 1991). The aggregation of utility is the basis for assessing the social welfare benefits or costs to society of policy or resource use change (Common 1995). Therefore an increase in social welfare can be thought of as the aggregated increase in the utility of individuals in society. Welfare economics further assumes that social welfare is maximised when a state of Pareto efficiency is achieved. A Pareto efficient state occurs when resources are efficiently allocated between uses, or in the case of two possibilities being on the frontier. Alternatively, it is when any change in resource allocation would not result in making one person better-off without making another worse-off (Johansson 1991). As utility is not restricted to monetary benefits improved environmental outcomes can lead to increased utility. This also translates to one person can be made better off by their utility increasing due to environmental outcomes improving. Conceptually this notion is displayed in Figure 2.6 evaluating whether a policy induces a change from point *C* and where this change is to, in terms of the production possibility frontier. Points on the PPF boundary are by definition Pareto optimal while points in the north-east quadrant from C are referred to as Pareto improvements.

This notion of Pareto efficiency has been extended by the Kaldor and Hicks compensation principle where a change is desirable if the winners from any change are able to compensate the losers, although actual compensation is not required to take place (Johansson 1991; Common 1995). While this study is focussed on the straight forward idea of identifying Pareto improvements, the Kaldor and Hicks compensation principle is also considered in the decision-making framework and is an important aspect when attempting to obtain a social ordering of outcomes.

2.1.5. Selection of optimal outcomes from society's viewpoint

Once a trade-off frontier is identified, the analysis can move onto identifying the point at which society's utility is maximised for the given set of outcomes. Ideally, this will be at some point on the frontier itself, however where the current state is one characterised by inefficient use of resources, such as point *C* (Figure 2.7), then society's current level of production, and therefore utility, lies somewhere within the boundary.

The identification of the point of maximum social utility is achieved using the concept of a social indifference curve (Boadway and Bruce 1984). A consumer indifference curve is a line identifying all of the combinations of the two outcomes for which a consumer is indifferent; i.e. a line made up of points at which the consumer's utility from consuming the two products, or in this case outcomes, is equal (Baumol 1994). In order to understand the implications when the PPF and indifference curves are combined, a number of properties of indifference curves should be considered. The first is that utility is equal as we move along an indifference curve, stemming from the indifference curve being a line identifying all of the combinations of the two outcomes for which a consumer is indifferent between them. Therefore the consumer is indifferent between point $A(a^1,b^1)$ and $B(a^2,b^2)$ in Figure 2.9. The second property is that assuming the consumer wants more of both of the outcomes, utility increases as the consumer moves to indifference curves further from the origin. That is, utility is higher on curve μ_2 than μ_1, and higher on μ_3 than μ_2. The final property is that indifference curves slope negatively from left to right reflecting that as the consumer seeks more of the x-axis outcome (while maintaining the same level of utility), the quantity of y-axis outcome decreases. Should the consumer attain more of the x-axis outcome while maintaining the same level of y-axis outcome, the consumer would have increased utility and moved to a higher indifference curve (Baumol 1994).

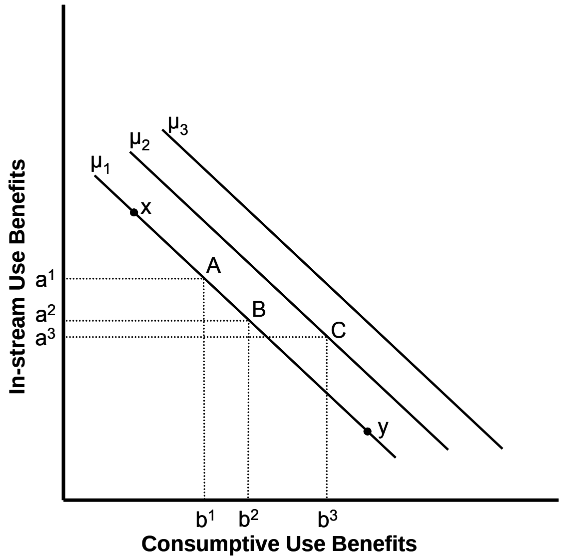


Figure 2.9: Straight line indifference curves identifying all of the combinations of the two outcomes for which a consumer is indifferent.

The slope of the indifference curve, referred to as the Marginal Rate of Substitution (MRS), is used to identify the amount of one outcome the consumer is willing to forgo to attain more of the other outcome (Baumol 1994). Similarly to the MRT for the Production Possibility Frontier, the MRS for the straight line indifference curve is constant along the curve and is calculated as the ratio of the y-axis change and the x-axis change. Therefore moving from A to B on μ_1

$$MRS = \frac{(a^1 - a^2)}{(b^2 - b^1)}.$$

The interpretation of the straight line indifference curves in Figure 2.9 leads to a conclusion that these are unlikely to occur. Consider the MRS of a movement along the indifference curve in Figure 2.9 from x to y. A constant MRS for both points means that at y the consumer is equally willing to trade away in-stream benefits in order to attain consumptive use benefits as he was at point x. In general however, it is assumed that at point y the consumer is less willing to trade off in-stream benefits to attain consumptive use benefits at y than x. Therefore, indifference curves are typically drawn curved inwards to the axes as in Figure 2.10.

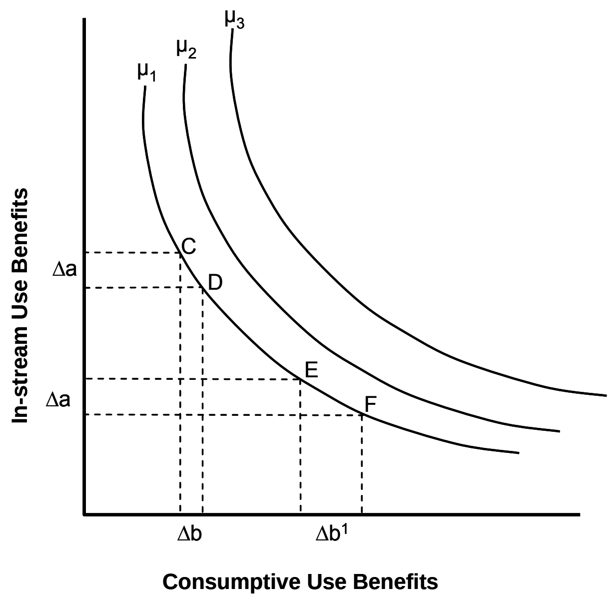


Figure 2.10: Indifference curves concave to origin reflect changes in willingness to trade-off as levels of outcome change.

The curved indifference curve has a variable MRS. A move from C to D results in a MRS of $\Delta a / \Delta b$ which is larger than the MRS of a movement from E to F being $\Delta a / \Delta b_1$. Therefore, there is a decreasing MRS as we move from left to right on the curve. In conventional consumer theory this is because of the diminishing marginal utility of increasing consumption of a good. In the case of multi-objective resource users this reflects the expectation that the willingness to trade-off one outcome for the other will decrease as we move towards high values for any of the two outcomes. That is, as we move down the curve μ_1 (in Figure 2.10), from F we expect less willingness to trade-off in-stream for consumptive benefits. Conversely, as we move up the curve from C we expect a decrease in the willingness to trade-off consumptive use for in-stream benefits (Baumol 1994).

If we assume that the assumptions regarding an indifference curve hold, then the point of maximum social utility is found by combining the Production possibility frontier and the Social Indifference curve showing utility as in Figure 2.11. In this case, the PPF $a_{max}b_{max}$ identifies the highest benefits attainable from all combinations of 'production' of the two outcomes. Therefore, as the indifference curve μ_1 is higher than μ_2 it follows that point E is the level of production that produces the highest level of utility. Point G, while having the same level of utility as E, is in the area of unattainable productive conditions.

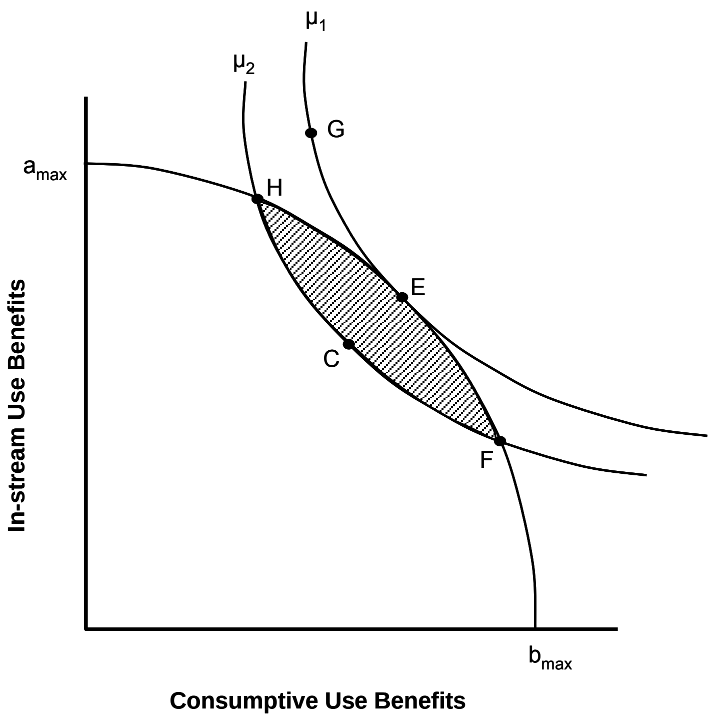


Figure 2.11: Combining the production possibility frontier and the social indifference curve to identify maximum social utility.

The level of maximum utility is also dependent on where the current level of production of the two outcomes is in relation to the frontier. Where the level of production is within the boundary such as point C in Figure 2.6, the point of maximum utility would be C on indifference curve μ_2 providing less utility than E on indifference curve μ_1. Points H and F are also on indifference curve μ_2 and therefore provide the same level of utility as point C. However, they also represent higher levels of production of either in-stream (H) or consumptive benefits (F), each representing a trade-off of the other. The shaded area therefore represents an area of higher utility than points along μ_2 while remaining within or on the PPF, and are therefore possible to achieve.

The combination of the identified production possibility frontier and the conceptual social indifference curve can be used to identify the level of outcome mix between consumptive and in-stream uses that is socially optimal (Wiggering et al. 2006). This socially optimal level by definition is contained on the frontier as points within the frontier represent inefficient resource use (Dudley 1997). Such an exercise allows the social ordering of options according to welfare economics. While this study does not attempt to identify social indifference curves, the conceptual shape when combined with the identified PPF can be used to provide insights into the preferences of society. They can therefore be used to assess the usefulness of seasonal climate forecasting information in environmental flow management.

2.2. Analytical approaches to the analysis of tradeoff.

The theoretical review to this point introduced the concept of improving or maximising social welfare by making a policy change that produced a Pareto efficient outcome. Furthermore, the production possibility frontier is a framework for assessing whether a policy change produces a Pareto improvement. It allows amongst a number of policy choices, the identification of the largest Pareto improvement and therefore the largest improvement in social welfare. The next step is to consider how this PPF framework is implemented – what analytical method is able to be used to estimate whether the outcome from any policy choice actually equates to a move in a north-easterly direction from the hypothesised starting point of C (Figure 2.6).

The problem being considered in this study is one where the outcomes cannot be measured by two single variables. Therefore the use of a PPF as typically defined will not be possible. Impacts on riverine environmental health are not able to be measured using a single variable as will be shown in Section 2.3. At best there is a relationship between riverine environmental health and the flow regime which in turn is typically described by a number of hydrologic indicators. On the other hand irrigated agriculture which is the primary consumptive user of water in the study area comprises more than the production of a single crop. In the study area, as will be explained in chapter 3 irrigated cotton is the largest single user of water, however there are other irrigated crops produced and there may well be more produced in the future. Therefore developing a PPF based on single production outputs will restrict the analytical method.

An alternative is to adopt a method similar to that used by Groeneveld et al. (2005), Calkin et al. (2002) and Scott et al. (1998) and develop a trade-off curve where the trade-off is between an outcome measured in dollar terms and another that may be measured using an indicator as opposed to a physical quantity. The actual indicators for each of the trade-off variables will be developed over the next two sections however it is important to note that in the event that the unit of measure of one or both axes is converted to dollar terms, the trade-off remains driven by the physical relationship as the prices used to convert to dollar terms are kept constant and production techniques are fixed. The monetisation of the indicator however does have the advantage that it allows additive benefits over a number of crops that use the one input that is changed.

In implementing this approach there are two variables that require estimation or valuation – consumptive and in-stream uses. These represent the x and y axes respectively. Development of a PPF requires an estimate of the response to increasing the use of water for each of these variables. This represents moving from the point of intersection of the x and y axes towards b_{max} for consumptive uses and towards a_{max} for in-stream uses. When plotted together, the responses will enable the mapping of the PPF which displays the combinations of outputs possible for two goods and therefore the trade-off of one good for the other.

The trade-off is to be made in terms of one variable that is amenable to monetary valuation, i.e. consumptive uses of water, and a variable whose valuation in monetary terms can be controversial and contestable, that is in-stream (environmental) uses. This influences the choice of analytical methods which are assessed below in terms of their ability to be used to map the PPF and thereby test the study's hypotheses.

There are a number of analytical approaches for estimating the two outcomes making up the PPF. These include two methods that can be used to estimate both outcomes in the same analysis: the analysis produces estimates of the social ranking of options. There are also a number of other methods that can be used to estimate one axis, while a different approach is then used to estimate the other axis. The following sections consider firstly the two 'stand alone' methods, multi-criteria analysis and extended benefit-cost analysis. Following this, I examine the approaches to estimate single outcomes for both consumptive and in-stream axes of the PPF.

2.2.1. Estimating net benefits using benefit-cost analysis

While the absolute level of social welfare is extremely difficult to measure, neoclassical welfare economics uses the notion of change to economic surplus as a means to measure changes in social welfare (Bennett 2000). Economic surplus accrues to both consumers and producers from the use of resources, and under perfect competition, the summation of consumer and producer surplus provides a measure of social welfare. Consumer surplus is the difference between what price a consumer pays for the good and what value that consumer places on that good (Parkin 1990). Bennett (2000) identified phrases such as "value for money" as evidence that consumers commonly obtain more value from a good than the price paid. Producer surplus is the difference between revenue and the opportunity cost of production (Parkin 1990). There is a positive change in social welfare where the summation of the change in producer and consumer surplus is greater than zero. This is analogous to attaining a Pareto improvement or a move in a north-easterly direction from the hypothesised starting point of C in Figure 2.6. On the other hand, a negative change in surplus represents a decrease in social welfare (Bennett 2000).

Benefit-cost analysis (BCA) is the most common tool used by economists to measure changes in surplus (and social welfare) and to rank alternative policy options (Ward and Beal 2000). Collins and Scoccimarro (1995) suggest that the process of analysing trade-offs between water users, e.g. consumptive users and the environment, can be approached through a benefit-cost analysis framework.

BCA adopts the rule that when assessing options where social benefits are larger than social costs, a net social benefit or Pareto improvement is achieved (Department of Finance and Administration 2008). Achievement of a social ranking of options therefore relies on estimating the net benefit of a number of options, such that the option with the highest net benefit (π) is the most preferred option. In the case of this study and the concept shown in Figure 2.2:

$$\text{Net Benefit } (\pi) = \text{Net Benefit Consumptive Use} + \text{Net Benefit In-stream Use} \quad (1)$$

Using a monetarised measure and further disaggregating, this becomes:

$$\pi = P_1 C + P_2 I \qquad (2)$$

where P = price of the outcome. C and I are the physical units of measure of each outcome; in the case of consumptive uses in this study it might be the yield of crops whereas for in-stream use some measure of environmental health units.

In its simple form, BCA however also suffers from a number of well-documented shortcomings. In particular for this study its failure to adequately account for the consumption of non-market goods. This includes people valuing the fact that the nation's rivers are healthy even if they do not derive any direct dollar benefit from this knowledge, which is the in-stream benefit outcomes on the PPF and P_2I in (2). See Bennett (2000), McMahon and Postle (2000) and The Interdepartmental

Committee on Environmental Economic Valuation (2006) for detailed examinations of BCA's inadequacies.

If we adopt the neoclassical welfare economics approach, where economic activity occurs to increase the welfare of individuals who make up society, noting that welfare constitutes consumption of both market and non-market goods, then policy changes resulting in changes to welfare need to be compared to determine if the increase in welfare outweighs what is forgone (Freeman 1994). Failure to account for consumption of non-market goods clearly decreases the usefulness of analysis outcomes and may, through a lack of information, lead to sub-optimal decisions.

In response to the need to value goods and services where values are not revealed in the market place, economists have developed a number of methods for estimating non-market values for incorporation into BCA. The rationale for use of non-market valuation stems from the neoclassical welfare economic paradigm, which holds that public goods provide welfare to individuals. Therefore, it is appropriate to derive a monetary value of the welfare changes ensuing from alterations to their level of provision and to include these values in project and policy appraisal (McMahon et al. 2000). Impinging on this is the fact that some impacts are identified as not subject to the market place. However, it also remains true that they still generate value (Bennett 2001) and therefore it is necessary to use non-market techniques to bring these impacts into analyses. Under this approach, extended BCA deals with achieving allocative efficiency where the highest valued set of outputs are produced at least cost. This is inclusive of environmental values as outputs are valued on what people would be willing to pay for products they want. At a point of allocative efficiency it is not possible to make one more person better off without making another worse off. This, by definition is a point of Pareto optimality. Should a policy change have a positive cost-benefit ratio then it is judged that a Pareto improvement will occur (Department of Finance and Administration 2008).

While the use of traditional BCA incorporating non-market environmental impacts (extended benefit-cost analysis) as a policy-development tool is increasing (Bennett 2000), there are a number of criticisms of these analyses. Bennett (2000) commented that the increased use of non-market valuation techniques (as a response to pressure to include environmental impacts) is one reason for rising dissatisfaction with BCA. Criticisms of non-market valuation techniques arise on two levels. The first is concerned with the ethics of placing monetary values on environmental amenity, as well as differences between individual preferences and values held by society (McMahon et al. 2000). The second level of criticism relates to the use of surveys and problems of bias (Lockwood and De Lacy 1992; Young 1996). Presumably it is primarily for these reasons that, despite the case for use of extended BCA as a decision tool in the water reform process, it appears to have had limited actual use to date. In addition to these issues, as will be discussed in section 2.3, there are inherent difficulties in estimating the biophysical outcomes from increasing water allocation to environmental purposes. Consequently, this makes valuation of outcomes problematic.

For this study, while BCA is a method that produces social ranking of options, it fails to identify the trade-off inherent in the decision-making process. Furthermore, while BCA does take some account of consumer preferences for both of the outcomes – economic and environmental, the tools used to undertake this; are known however to have issues which have precluded them from use in a policy setting in Australia. These tools include obtaining preferences by eliciting monetised values for in-stream benefits through surveys via contingent valuation and/or choice modelling (Environment Protection Agency 2006); Exacerbating this is the complexity of understanding all of the aspects of river health leading to the inability to accurately predict the outcomes from changes in water management. This can be difficult to communicate in surveys and to enable people to differentiate and value different outcomes. At present there is inadequate knowledge of the impacts of water management changes on river health, and therefore concrete outcomes cannot be used in surveys, and those surveyed have only weak reference points due to this uncertainty to make their valuations. Therefore BCA is considered inappropriate for this study.

2.2.2. Estimating net benefits using multi-criteria analysis

An alternative 'stand alone' approach that does not produce a monetarised ranking is multi-criteria analysis (MCA). Conceptually this method still aims to achieve a social ranking of options relying on estimating the net benefits of a number of options such that the option with the highest net benefit (π) is the most preferred option. That is,

Net Benefit (π) = Net Benefit Consumptive Use + Net Benefit In-stream Use.

However in this case instead of using equation (2) (page 41) the approach instead translates (1) into

$$\pi = W_1C + W_2I$$

where W is a weighting given to each of the outcomes.

MCA incorporates the consideration of a range of criteria in incommensurable units and therefore does not require monetary values to be assigned to environmental (or social) criteria (Hyde et al. 2004). It is a tool espoused as an alternative to Benefit-Cost Analysis (BCA) (Gillespie 2000; BTE 1999), however MCA represents a broader approach than BCA where the objective is economic efficiency of resource allocation through consideration of impacts on net social benefit (Department of Finance and Administration 2008; Gillespie 2000). In fact MCA may include BCA as a component in its analysis framework. MCA is preferred (over BCA) in some instances for a range of philosophical and ethical reasons including BCA's heavy dependence on monetary valuation, particularly or goods and services not easily valued, such as many environmental services (BTE 1999). MCA deals with these valuation issues by providing an analytical framework that does not require all benefits and costs to be valued in dollar terms for the analysis of policy trade-offs where there are multiple objectives (Flug et al. 2000; Hajkowicz 2006; BTE 1999).

The analytical process of MCA can be described in a step-wise fashion. The first step involves defining the objectives of the policy that are considered important by the decision-maker. Second, a series of criteria which are to be used to evaluate the alternatives is selected. Following this, the alternatives are scored in relation to how well they perform in relation to each criterion. The fourth step entails assigning a weighting to each criterion which in effect determines its ranking of importance. The fifth step involves aggregating the scores for the criteria to identify a preferred option. A range of techniques are used in this aggregation (Asafu-Adjaye 2000). At times a final step whereby further consultation and interaction is undertaken with stakeholders on the results. This step can lead to modification of weights and subsequent changes to preferred options.

Despite prima-facie reasons for its use, MCA has also attracted considerable criticism pertaining to its lack of theoretical underpinnings, as well as a number of practical difficulties that can influence the outcomes of analyses arbitrarily depending on the analyst's preferences (BTE 1999; Gillespie 2000). Countering this argument, Hajkowicz (2006, p. 1) argues that MCA has over time "undergone considerable methodological advancement" and that the "common methodological challenges and potential sources of error" can be overcome.

Drawing on BTE (1999), Gillespie (2000) presents a critique of MCA from an economist's perspective. While the comprehensive list of issues are not canvassed in detail here, Gillespie concludes that MCA is likely to hinder the understanding of resource allocation options leading to poor social outcomes. Amongst other issues, the critique highlighted that:

1. at the basic level the lack of an established theoretical framework to provide rigor and consistency to the practical task of carrying out a MCA introduces a risk of inconsistent results for the same task by different analysts. An example of inconsistency surrounds the most appropriate evaluation method to use. Asafu-Adjaye (2000) states that although there are as many as 50 possible evaluation methods it is ambiguous as to which methods are best performed,

2. the selection of criteria against which alternatives are evaluated can be arbitrary e.g. ease of measurement, introducing risks of failure to adequately account for all benefits and costs, double counting of costs and benefits, as well as potentially introducing implicit (inappropriate) weighting of criteria,

3. the range of scoring systems developed for use in MCA can lead to inconsistencies in results as well as to misleading results where relativities between alternatives in terms of measures for a particular criterion are obscured. For example consider the simplified case where estimates of soil erosion of 5, 20 and 50 tonnes per hectare of sediment are calculated. If these scores are ranked using an ordinal scale, the relative differences i.e. the fact that the third ranked score 50 tonnes per hectare is ten times higher than the alternative with the least erosion and the second ranked is four times higher, is obscured; and,

4. the specification of weights for each criterion by consultants, bureaucrats, politicians and even a sample of community representatives are unlikely to reflect the aggregate preferences of the whole community. Bennett (2000, p. 987) similarly notes that "MCA allows those who are dissatisfied the freedom to incorporate their own values". Ultimately this gives rise to credibility issues as the analyst can be influenced by a range of stakeholders including politicians. This is recognised by Flug et al. (2000, p.276) who note that "the most important step in the overall system analysis is the assignment by qualified, unbiased experts of numeric rating values to each attribute". This is complicated by the potentially large numbers of criteria and weights which

may lead to unintentional implicit weights producing results that are difficult to rationalise (Asafu-Adjaye 2000).

These issues raised by BTE (1999) and Gillespie (2000) are congruent with those raised by Hajkowicz (2006) who lists five potential sources of error in MCA:

1. incorrect problem structure;
2. poor performance data;
3. inappropriate capturing of decision-maker preferences;
4. incorrect application of additive utility; and,
5. duplicate or overlapping criteria.

An example of some of these issues can be shown using a MCA study undertaken to assess the best preference amongst nine reservoir storage and release policies for the Glen Canyon dam in the Colorado River Basin of the USA (Flug et al. 2000). The study included seven resource criteria – fish, vegetation, wildlife and habitat, endangered species, cultural resources, recreation and power. The weightings for each of these criteria were identified to "reflect various resource interests, through interpretation of the Glen Canyon project's formulation and legislation and adherence to the objectives of these environmental studies" (Flug et al. 2000, p.272). The seven criteria were assigned up to seven attributes each (a total of 29 attributes) that were weighted depending on their importance to the resource criteria. Each attribute in turn received a rating on performance against every alternative "hopefully by an objective professional who understands the influence of reservoir and flow release variables on the respective attribute" (Flug et al. 2000, p.271).

This simple summary establishes that the analysis was nested to the extent that the final ratings and ranking for each alternative were influenced at five points – the resource criteria, their respective ranking (and subsequent weights), the selected attributes and their weights, as well as the expert judgment on how well the alternatives score for each attribute. At the very least this process fails to demonstrate how community preferences are reflected in the outcomes. In addition, as the weightings at each level have the potential to impact on the final ratings and ranking of alternatives, it is clear that small errors or differences of opinion in weights may lead to changes in which option/s are preferred. This point is also illustrated by Flug et al. (2000) when a range of alternative rankings of criteria were tested resulting in changes to the preferred alternative.

The results of the MCA for Glen Canyon dam showed that of the nine flow alternatives, three consistently scored lower values, leaving five alternatives preferred from the baseline. The study further notes that the choice amongst these five alternatives remains one for the decision-makers, highlighting a final potential problem with this type of analysis. In introducing MCA, Flug et al. (2000, p.270) state that MCA "methods provide a framework to help water managers identify critical issues, attach relative priorities to those issues, select best compromise alternatives for further consideration, and enhance communication, hopefully to, gain general acceptance". While the analysis undoubtedly progresses some of these goals, it is unclear how the selection of five scenarios for "further examination" by a methodology as broadly targeted as MCA progresses decisions surrounding trade-offs for improving water resources management.

In considering an appropriate assessment framework for this research four reasons in addition to those noted above are proposed to justify not using MCA. The first relates to the requirement for access to experts and data in each field related to the criteria selected for evaluation. This was not considered feasible in this project. A second difficulty alluded to in section 2.3, is that of estimating the environmental outcomes from increasing water allocation to in-stream uses. There is a lack of concrete knowledge about the extent to which the human management of rivers effects on river health. Additionally, significant uncertainty exists about the extent that changes to river management will improve river health. As will be shown in section 2.3, at this point the default is to use the impact on hydrologic indicators from river management changes as a proxy for river health. Use of this default impedes the provision of a scale of river health impacts that participants can use for the ranking of options. Therefore, an ordinal ranking of options is the only alternative which restricts the decision process to one of thresholds where the decision-maker is left to judge whether a particular positive impact on hydrologic indicators is sufficient to outweigh a level of economic loss. The third and more important reason relates to the need to match the level of analysis with the aims of the study. In this case the aim is to develop and test an appropriate methodology for assessing the impact of using SCF in environmental flow management on both the regional economy and the health of the riverine environment of the Border Rivers Catchment. In a resource constrained environment one method of achieving this aim is to develop a framework that facilitates the preliminary ranking of alternatives against a baseline that may be undertaken in a desktop manner. In the event that the results indicate that further study is warranted, a more detailed approach can then be undertaken. This is likely to be considerably less resource intensive than undertaking a Multi-Criteria Analysis.

Finally, it is useful to consider MCA in the reference to the aim of seeking a Pareto improvement. While a potentially useful tool for analysing trade-offs between outcomes from different options, the final preferred option from an MCA is not necessarily an indication of a Pareto improvement. If we accept that the goal of the policy in question is to achieve a Pareto improvement through a move in a north-easterly direction from point C in Figure 2.6, or another point that represents the current status of the two outcomes, then the analytical method chosen has to provide information to at least show the direction of movement. The use of weights, aggregation of scores and ranking obscures an assessment of this direction of movement, thus hindering a judgment of whether adopting a particular policy would lead to a Pareto improvement. As clearly stated by Gillespie (2000, p. 1):

> *"Attempts to integrate the results of these assessments using the subjective and arbitrary procedures of MCA only serve to obscure the results of individual analyses and the tradeoffs involved between objectives".*

The final reason for not adopting MCA relates to the analytical limitations noted above and the lack of apparent adoption of MCA in policy decision-making in Australia. While there appear to be a number of benefits from the MCA framework there is little evidence that this framework is preferred by decision-maker, hence studies using this framework are unlikely to have results adopted for use or further study. Therefore, MCA was not considered an appropriate tool to test the study's hypotheses.

2.2.3. Threshold value analysis

An alternative to undertaking extended BCA that retains the neoclassical approach of BCA to valuing social welfare, but avoids the requirement to place monetary values on the environment is threshold value analysis (TVA). TVA is a two-step process where the first step is estimating the change in economic surplus from marketed goods and services expected from the change in resource allocation. This value is used in the second step as the opportunity cost of the resource use decision and is compared with the non-monetary valuation of environmental outcomes (Bennett 1999; Streeting and Hamilton 1991; Jayasuriya 2004). In other words, TVA provides a threshold which the environmental benefits should exceed for the policy change to be deemed acceptable. In relation to Figure 2.12, TVA can be used to identify the outcomes of each option in relation to each other and by identifying the trade-off, say between points c and x, allowing more informed decisions on options to be made. The use of minimum acceptable thresholds will further aid analysis by identifying options that are outside the efficient set similarly to point x. Use of TVA in resource allocation decisions places the final judgement of whether the (restricted) economic benefits outweigh the environmental costs into the hands of decision-makers such as policy-makers and/or politicians (Webster 1998; Bennett 1999).

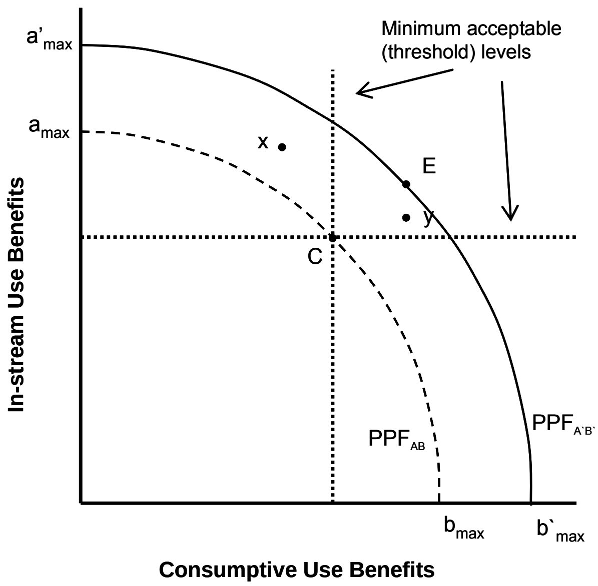


Figure 2.12: Using threshold value analysis in a PPF analysis to identify the trade-off amongst options.

The 1998 Snowy Water Inquiry provides a useful example of a process undertaken to examine trade-offs between users and to make recommendations on restoration of the Snowy and associated rivers. The Snowy inquiry necessarily adopted a wide ranging methodological approach as it was required to consider environmental, economic, social and heritage issues (Webster 1998). The inquiry Commissioner noted that when advised of the fact that flows had been decreased to one percent of the original flow below Lake Jindabyne, Australians were interested in returning some of these flows. However, when appraised of the opportunity cost of these returned flows to agriculture and the hydro electricity scheme, people found the trade-off decision challenging.

The economic analyses undertaken as part of the inquiry involved benefit-cost analysis of various options examining the trade-offs, particularly with irrigation and hydro electricity. The results of the inquiry were presented as "economic surpluses" for the benefits and costs. No attempt was made to put monetary values on environmental impacts. In addition to the methodological controversies associated with environmental valuation methods, focus group research by the inquiry suggested stakeholders would have difficulties identifying and separating trade-offs between the environment, social and equity values. Non-market values were considered by the inquiry but it appears that no formal analyses using tools such as choice modelling or contingent valuation were undertaken.

As part of the TVA analysis conducted for the Snow Inquiry, the change to economic surplus was used as an indicator of the value to society from the enhancement of environmental flows. Specifically, impacts on regional agricultural gross margins were used to determine changes in producer surplus, which provided an estimate of the opportunity cost of introducing environmental flows. A series of analyses were conducted comparing a standard case with various flow scenarios. Non-use values were not explicitly identified in this economic framework but are accounted for implicitly by the adoption of an acceptable level of riverine health, which provided a threshold between acceptable and non-acceptable outcomes.

The water reform process being undertaken in Queensland has also included examinations of the trade-offs between consumptive and environmental uses. In these examinations, the analyses of economic and environmental impacts have tended to be undertaken in what might be termed an isolated approach. The analyses were underpinned by a common catchment hydrologic model, which provided data to estimate the environmental outcome of various water management scenarios and water supply data for the financial analysis.

In the Fitzroy Basin catchment a number of economic analyses were undertaken to examine the impact of new flow rules which encompassed environmental flow outcomes. These included the impact on irrigation within the basin, potential impacts on commercial and recreational fishing, and the economic benefits that may be available through additional private development of infrastructure. The irrigation impact assessment from a range of environmental flow scenarios adopted a gross margin[8] approach where gross margin was used as a proxy for the net income derived from irrigation. Net economic impacts across the catchment were calculated by aggregating the gross margin across the subcatchments within the basin (DNR 2001). In the context of the PPF framework in Figure 2.12 the net impacts are used as estimates of consumptive use benefits. Gross margin is able to be used as part of the trade-off analysis because all other costs are held constant and therefore change in output flows through to changes in gross margin.

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[8] Gross margin is the difference between the gross income and the variable (direct) costs (Makeham and Malcolm 1981)

The analysis of environmental flow scenarios in the Fitzroy Basin comprised two steps that were related to the levels of infrastructure development since it is not legitimate to compare gross margins for different infrastructure (i.e. capital) development levels. In the first set of analyses, a scenario using the already existing infrastructure levels, water entitlements and costs was developed. Results from this scenario were compared against other scenarios where environmental flow rules were introduced. In the second set of analyses, a baseline scenario with additional infrastructure and other management rules was compared against environmental flow scenarios. In reporting the findings of the economic impact analyses no attempt was made to examine the trade-off between consumptive users and the environment. While this comparison was probably a component of the final decision process in the development of the Water Resource Plan and the associated development of the Resource Operations Plan, it is not clear how this comparison impacted upon subsequent decisions.

In addition to not examining the trade-off amongst the different scenarios there are a number of constraints inherent in the use of gross margins in this manner. The economic impact report (DNR 2001) discussed the limitations including, amongst others, the following issues:

- Failure to consider:
    - a number of financial aspects such as farm debt or the impact of changes to farm capital that may be required under different options. This may have considerable impacts on economic outcomes through the exclusion of costs, for example, interest (see also Douglas et al. (Douglas et al. 2004)); and,
    - the ability for capital change driving technological improvement, such as yield increases. These would be expected to occur at the farm and regional level over the medium and longer term, and may have considerable impacts on economic outcomes;
- The model did not account for benefits or costs to the urban and mining sectors either through their use of water, or the indirect impacts from irrigated agriculture; and,
- The aggregated nature of the model, where the basin was split into four subcatchments, and consequently did not account for a number of biophysical differences including soil types or ground water salinity variations.

Despite these issues, this approach was justified on the grounds that it was able to provide an indication of the potential magnitude of net irrigation impacts from environmental flow scenarios (DNR 2001).

In New South Wales (NSW) the process of introducing flows for environmental outcomes is being addressed through the development of Water Sharing Plans. These plans define water sharing rules that allocate water for environmental needs and define how water is to be shared amongst users (NSW Department of Water and Energy 2007a; Department of Infrastructure Planning and Natural Resources 2004). These environmental flow rules are developed within sets of guidelines and constraints that recognise the potential for economic-environmental trade-offs. For example Jayasuriya and Crean et al. (2001) note one constraint as the impact on water users being restricted to no greater than 10 percent of their average diversions. This implies that any flow rules require their impact to be examined prior to being implemented.

In evaluating the trade-off for the environmental flow rules in the Murrumbidgee Valley, Jayasuriya and Crean et al. (2001) and Jayasuriya (2004) utilised linear programming and optimisation to estimate the scale of impacts from proposed water management rules. This was achieved through estimation of the opportunity cost to the agricultural sector from the adoption of environmental flows. The analytical framework involved a combination of hydrologic and economic simulation modelling that incorporated both strategic and tactical on-farm decisions to be made in response to changed water supplies from the rules. In determining the impacts, regional gross margin for a base (current management rules) case was compared to scenarios incorporating environmental flow rules. The analysis did not identify either the economic value of the environmental benefits nor the expected environmental benefits using bio-physical measures. Rather a threshold value approach, where the impact costs identified provide an estimate of the level of benefits which would need to be attained from the environmental flow rules to make the alternative rules worthwhile on economic grounds, was used.

Eigenraam et al. (2003) used regional gross margin as an indicator of profitability of agriculture in the Murray River system area to estimate the impact on the irrigation sector of reductions in water availability. This analysis was undertaken over 22 regions in Victoria and New South Wales in a linear programming framework and sought to assess the change from a base case based on the flow and access rules to water before environmental flows were introduced.

While the approaches in the Fitzroy Basin and Murrumbidgee area estimate consumptive use benefits, and by extension the opportunity costs from adoption of scenarios, they do not on their own provide an indication of the in-stream use benefits that a decision-maker requires when undertaking a threshold value analysis. The benefits of TVA are that (when information on in-stream use benefits is included) it retains the link to social preferences and Pareto optimality. By adopting the approach of MCA and not placing monetary values on environmental benefits, TVA does not suffer from the limitations associated with tools such as choice modelling (see (Environment Protection Agency 2006). By not weighting respective economic and environmental outcomes the approach minimises the risk associated with MCA where a complex series of arbitrary weightings can cloud the ranking of options. While some may argue that decisions on preferred options in TVA remain weighted by a decision-maker such as politicians, the trade-offs are at least clear and the implicit weighting placed on outcomes by decision-maker/s is also clear.

2.2.4. Input-output analysis

The fourth methodology reviewed, input-output (IO) analysis, first proposed by Wassily Leontief in 1951, can be used to extend the economic impact assessment of private benefits identified in the previous section by estimating impacts on the regional economy. An IO model conceives the economy of a region as being divided up into a number of sectors allowing the analyst to trace expenditure flows and to estimate the flow-on impacts of policy changes. In simple terms, an IO analysis is based on an input-output table that captures the inter-industry transactions that occur in an economy for a particular year (United States Environmental Protection Agency 2000). The inter-industry transactions map the linkages between industries and form the basis for assessing flow-on impacts for changes in demand or production, primarily through the calculation of multipliers. IO multipliers are ratios based on the linkages between industries and provide estimates of impacts on the level of economic activity. For example gross output, value added, household income and employment are taken into account. A description of IO tables, their development and analysis can be found in Australian Bureau of Statistics (2001) and Australian Bureau of Statistics (McLennan 2008).

The primary use of IO in Australia is in impact studies to assess the expansion of a current industry, the introduction of a new industry in a region, or the economic significance of an existing industry (West 1999). A recent example of the use of IO analysis in Australia in a regional context is provided by CARE (2006). This study assessed the economic impact from the introduction of a marine park on the economy of the Eurobodalla region in New South Wales. The study estimated that the marine park would reduce the Eurobodella Gross Regional Product (GRP) by approximately $1.0 million and that this represented 0.1 percent of the Eurobodella GRP which was estimated to be $872 million. Further calculations assessed how the reduction in GRP would be offset by the activities involved in the implementation and on-going management of the marine park (CARE 2006). Similar calculations were also made for household income and employment.

The process of undertaking an IO analysis, while time consuming and data intensive, provides the analyst (and by extension the policy-maker) with considerable information on the structure of the economy in question. Estimation of the linkages and importance of a particular industry to an economy, such as in the Eurobodalla case, can be used in policy-making to ascertain whether impacts are expected to be positive or negative and on what scale. This information can be used in the policy development process where decisions on mitigating options can be made where impacts are expected to be negative.

IO analysis suffers from a number of shortcomings, both practical and theoretical. From a practical standpoint disaggregated IO tables do not exist for many areas and their development requires detailed information on industry expenditures, where they occur and the flow of expenditures in these receiving industries. This is a costly exercise.

The theoretical shortcomings relate to the use of IO analysis in estimating the regional, state or economy wide impacts from a given shock. They include:

- the typical static nature of IO models does not take into account the dynamic processes involved in the adjustment to an external change;
- the use of fixed coefficients means the analysis assumes that an industrial structure remains unchanged by an economic event and provides no scope for the substitution of inputs of production;
- the coefficients represent average relationships rather than marginal responses to stimuli and therefore result in responses that are constant;

- the lack of supply side constraints for inputs, labour etc, results in potential over-estimation of impacts as there is no rationing device when scarcity exists; and,
- IO analysis usually does not take into account compensating adjustments for government programs stemming from where the funds are sourced from.

(United States Environmental Protection Agency 2000; Bureau of Transport Economics 2006; Layman, 2002).

The use of IO analysis in this study could assist in identifying the regional economic impact of alternative management options by taking the private benefits analysis using gross margin approach a further step and estimating the impacts on the regional economy. However, while IO analysis is used to measure the economic impact of a policy change or activity the method does not estimate the magnitude of the benefits and costs involved. As such it provides no indication of whether a particular policy or activity should be undertaken from a social viewpoint (Marsden Jacob Associates 2006; Bureau of Transport Economics 2006). Bennett (2000) suggests that IO analysis is a useful planning tool to assist policy-makers understand the consequences of a policy or activity on an economy. However, similarly to Marsden Jacob Associates (2006), Bennett (2000) concludes that because the values generated in IO analysis do not relate to social welfare they are unsuitable for use in deciding whether a policy or activity will result in an improvement in social welfare. This inability to estimate whether an outcome is a Pareto improvement led to the decision that IO analysis was unsuitable for use in this study.

A further practical reason for not using the IO method relates to the mis-match between the static nature of IO analysis and the time series nature of the biophysical and environmental impacts undertaken in this study. An IO analysis is a static representation of the structure of an economy at a point in time, typically one year. As such, an impact modelled in IO analysis represents a short-term impact on the economy. The time series nature of the modelling in this study permits the examination of the impacts on irrigators through time and for a period of time. For example, the impacts on irrigators can be assessed and water supply can be estimated across a planning horizon of twenty years where the outcomes for each year are partially a function of the preceding year. This structure does not match with that of IO analysis, which because of its static nature cannot trace and examine impacts though time.

2.2.5. Computable general equilibrium models

An alternative and extension to IO analysis is found in computable general equilibrium models (CGEs). A CGE model is a mathematical model of a national economy or a region that can be utilised to undertake quantification of the costs and benefits of environmental policy. A key advantage of CGE models is that they allow for adjustments in all sectors and thus overcome some of the limitations of the IO technique. Similarly to IO models CGEs are impact analysis models that assess the impact of a policy (often termed a ‘shock’) on a national, state or regional economy.

A recent example of the use of a CGE in an agricultural setting is an assessment of the impacts of the 2002-03 drought on regions in Australia using the TERM[9] model (Horridge et al. 2003). Results were presented in terms of real household consumption, real investment, real GRP, aggregate employment and real wage rate at a regional level e.g. Eyre South Australia. Additional results for each region by industry e.g. rice, cotton, sheep etc. were also available.

Similarly to IO analysis, CGE models could be used to assist in assessing the impact on the consumptive use benefits axis in Figure 2.6, i.e. estimation of the movement from C to y. However, as with IO models, CGE models at the regional scale, a relatively new phenomenon, do not commonly exist and are highly specialised and expensive to develop. Furthermore as CGEs use IO tables in the analytical framework they suffer from a number of the shortcomings of IO models. Of particular importance are the points raised by Marsden Jacob Associates (2006) and Bennett (2000) regarding the unsuitability of IO analysis information in determining the social welfare impacts of a policy or activity.

2.2.6. Summary of the economic framework

The proposal to limit the analytical framework to assessing in-stream and consumptive uses is acknowledged as a simplified approach however it does permit the examination of the trade-offs of moving water from the production of one outcome to the other. A more complete analysis would consider other factors such as social impacts, administrative feasibility, fiscal effects etc. From a broad theoretical perspective, van der Lee and Gill (1999) argue that a trans-disciplinary approach encompassing social, cultural, economic and environmental information is required to make such decisions. However, they propose that there is no generally accepted method for making decisions on water allocation between competing users. Furthermore, while there have been a number of attempts to develop such a framework, some decisions have been arbitrary, hasty and often politically driven and are therefore sub-optimal. They contend that the practice of meeting the needs of particular ecosystems is flawed and it is unrealistic for any discipline in isolation to be able to provide sufficient information for decision-making.

While agreeing with the theoretical viewpoint of van der Lee and Gill (1999), it is also clear that in the final analysis this aim may not be achievable in practice. Therefore, simpler frameworks and tools to identify the scale and direction of trade-offs involved in policy alternatives need to be used to minimise the negatives highlighted by van der Lee and Gill (1999). Knights et al. (1995) suggest that environmental flow decision-making in the water reform process involves a trade-off between the level of environmental, economic and social risk and degree of damage to the environment that the community is willing to accept. Selection of the tools to assess these factors involves a trade-off between theoretical robustness and practical decision-making.

From one perspective MCA appears to be the method of choice because of its ability to analyse impacts of outcomes that are not all measured in dollar terms. It (along with all other assessment methods) has a number of weaknesses and is thus deemed

[9] "The Enormous Regional Model".

to be inappropriate for use in this analysis. Instead, threshold value analysis (TVA) with stronger links to assessing Pareto outcomes from a range of alternatives as shown in Figure 2.6 is selected for use in this study.

The use of TVA is justified on the grounds that it is underpinned by its links to benefit-cost analysis, and by extension, to the notion of social welfare and Pareto optimality. That is, what can be measured is measured. By excluding the explicit valuation of environmental impacts this analytical method may more acceptable to decision-makers and as shown by Webster (1998) and Streeting and Hamilton (1991), TVA is an approach that has been used in policy deliberations. These factors mean that of the analytical choices TVA is relatively simple but still robust and able to achieve the project aims.

While TVA provides a benefit through not requiring monetary values to be placed on the environment, it does also add a limitation to the judgement of the direction of movement from point C in Figure 2.13. Extended BCA, by valuing in-stream uses, would provide an estimate of the change from point C towards point z, i.e. an estimate that included movements on both the x and y axes (e – f and b - a). TVA however, provides no information on the movement along the in-stream use benefits axis and as such cannot provide an estimate of b – a, or indeed if the impact on the y axis is positive from point C. To enable the TVA approach to be used in this study an estimate of the environmental impacts needs to be attained for comparison with the economic impacts.

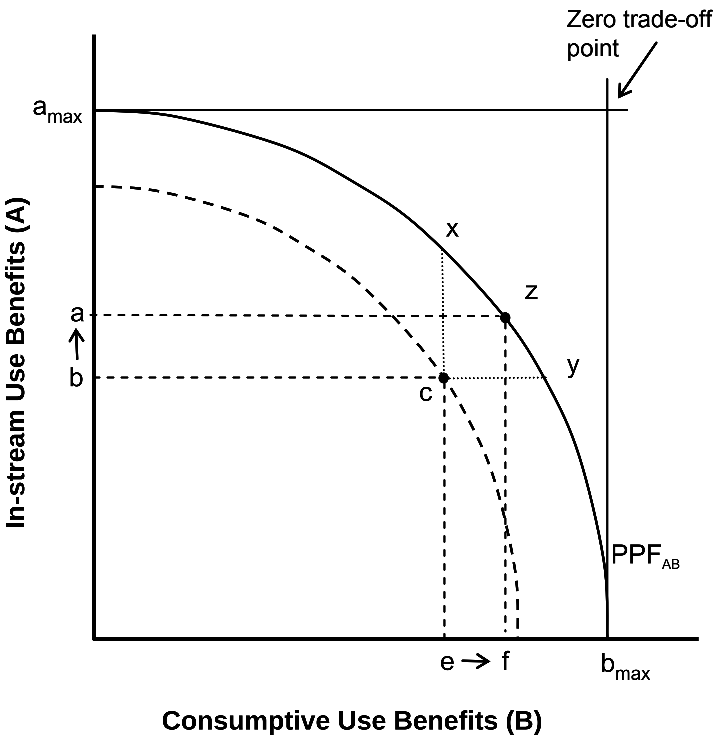


Figure 2.13: Incorporating a policy change into the analysis of trade-offs between in-stream and consumptive benefits of water use.

2.3. Estimating in-stream benefits

To enable a judgement on the Pareto outcome of a particular option the methodological framework must include an analysis of the impact on in-stream use benefits of changes in allocations, i.e. $b - a$ (y axis in Figure 2.13). This involves considering methodologies that could be used to estimate the gains or losses in environmental benefits from different flow management scenarios. At a broad level there are a number of methodological options.

In the first instance the values in extended benefit-cost analysis may be used. As already mentioned, tools such as contingent valuation and choice modelling (Environment Protection Agency 2006) can be used to estimate values of different options along the y-axis in Figure 2.13. Estimates of in-stream use benefits using these tools however, are driven by the preferences of those surveyed. In developing a trade-off model, the trade-off is driven by changes in the physical relationship. Therefore tools such as contingent valuation and choice modelling which value preferences for outcomes, not the outcomes themselves, are themselves limited.

The same can also be said of the other method that links to economic analysis, multi-criteria analysis. Similar to the proposal regarding benefit-cost analysis, the subset of a MCA pertaining to the environmental values could be used to determine environmental outcomes from policy or management options. However the use of weights for criteria is strongly linked to preferences and less linked to physical relationships, thereby also making this option unacceptable.

The alternative to the approaches taken in a MCA or extended BCA, and implied in the discussion on TVA, is to estimate the environmental outcomes from flow management scenarios using a physically based method. This method uses measures of the hydrologic characteristics of rivers that are related to riverine environmental health outcomes to gauge the impact of flow management scenarios and is described in following sections. While not providing a quantitative monetary measure in Figure 2.13 of b – a, this approach nonetheless (assuming a positive move from e to f), will provide sufficient information to infer if the movement from point c is in a north-easterly direction towards the frontier.

The first step in estimating environmental outcomes entails exploring issues surrounding the development and use of flow management rules for environmental outcomes. This is critical to enable estimation of outcomes and involves consideration of what river health actually means and how humans have impacted on river health by changing how rivers flow. The influence of human management is a focus as riverine ecosystems are affected by natural events such as drought and floods in the absence of human intervention (Davies et al. 2008). Technical review of the 'science' of environmental flows is not attempted as that is outside of the scope of the study. Rather the focus is on exploring how the impact of human river management on river health is assessed, and making use of this knowledge to examine the efficacy of the flow scenarios in achieving gains or losses in environmental outcomes. This will provide the basis for the non-monetary valuation of environmental outcomes, which forms the second component of the threshold value analysis (y axis in Figure 2.13) to determine the efficacy of the SCF information in decision-making. The next section will examine the definition of environmental flows, followed by a discussion of the broad methodological issues, before concluding with an overview of the methods that have been used in similar studies.

2.3.1. Environmental flow management – background and policy direction

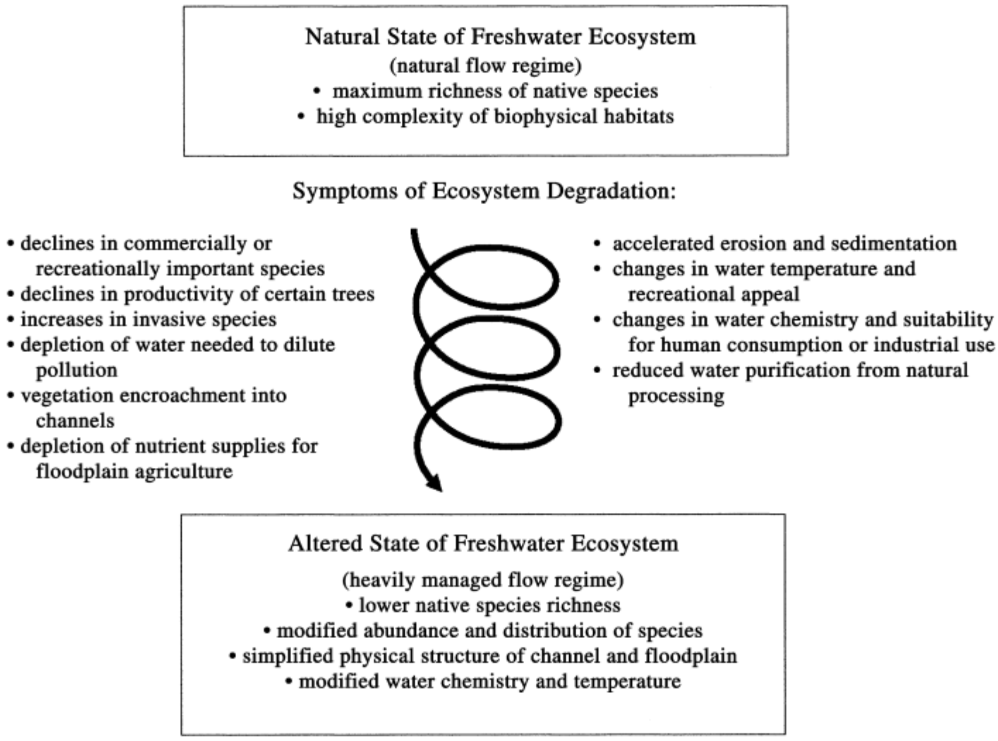
In Australia the broad policy that elucidates the direction for environmental flow management (EFM) is the *National Principles for the Provision of Water for Ecosystems* (see Appendix B). These principles clarify the notion of ecological sustainability: "to sustain and where necessary restore ecological processes and biodiversity of water dependent ecosystems" (ANZECC 1996, p. 5). The specific principles related to EFM are:

> *PRINCIPLE 6 Further allocation of water for any use should only be on the basis that natural ecological processes and biodiversity are sustained (i.e. ecological values are sustained).*
>
> *PRINCIPLE 9 All water uses should he managed in a manner which recognises ecological values.*
>
> *PRINCIPLE 11 Strategic and applied research to improve understanding of environmental water requirements is essential.*

(ANZECC 1996, p. iii).

This policy direction has been agreed to by the state jurisdictions that hold the water management responsibilities through the 1994 CoAG agreement (National Competition Council 2004).

The concept of EFM has emerged as a central component of new approaches to water management. The science of environmental flow assessment (EFA) and the extension of management has grown out of the recognition that hydrological alteration of river systems has resulted in environmental degradation and the subsequent need to determine how far a river can be altered from the natural whilst remaining ecologically viable (Tharme 2003). Richter et al. (2003) describes how altering the flow regime of a river has a number of knock-on effects to physical, chemical, and biological conditions and functions leading to degradation of the river ecosystem (Figure 2.14).



(Source: (Richter et al. 2003)

Figure 2.14 Developing rivers for extractive purposes results in degradation of the river ecosystem.

EFA is simply defined by Tharme (2003, p. 400) as "assessment of how much of the original flow regime of a river should continue to flow down it and onto its floodplains in order to maintain specified, valued features of the ecosystem". The EFA broadly leads to a management regime for the river EFM). The ultimate aim of EFM in this study is to at least maintain and where possible improve riverine health. This is congruent with Cooling et al. (2002 p. 75) who proposed that the objective of EFM is to "restore elements of water regime which will arrest ecological change and secure or restore the biological diversity of flood plain systems". Richter et al. (2003, p. 207) similarly states that "the ultimate challenge of ecologically sustainable water management is to design and implement a water management program that stores and diverts water for human purposes in a manner that does not cause affected ecosystems to degrade or simplify". Shiau et al. (2007) put it more simply

> *"A key challenge for environmental flow assessment is to determine how much of the original flow regime should continue to flow down a river and onto its flood- plains in order to maintain the valued features of an ecosystem"*

This then leads to the question: what level of restoration is required for the river to be considered healthy? The concept of river health is not easily defined and opinions differ on what is a healthy river and how best to approach the provision of environmental water flows (Department of Natural Resources 2001; Whittington 2000; Cottingham et al. 2005). Recent World Bank studies suggest that healthy rivers are those that are relatively undisturbed (Davis et al. 2004). Marshall et al. (2001, p. 4) refer to “good condition” or in a “near natural state”. Arthington (1998) refers to maintenance of functional integrity similarly to Davies et al. (2008, p. 5) who propose that a river is “healthy when its essential character is maintained over time, notwithstanding disturbances due to human activities or the vagaries of climate”. Allan and Lovett (1997, p.7) discuss developing “environmental flow strategies that embrace ecological sustainability and the needs of particular water dependant ecosystems”.

In Australia streamflow management plans for environmental requirements have been largely focused on maintaining ecological values through ensuring ecosystem viability underpinned by supplying sufficient water (Cottingham et al. 2005). Jones et al. (2003, p. 13) broaden the definition noting that “a healthy working river is one that is managed to provide a sustainable compromise, agreed to by the community, between the condition of the river and the level of human use”. This broadened definition recognises the trade-off involved between consumptive and environmental outcomes. In the final analysis, the specific definition is perhaps not that important because the stated environmental goals of any river management regime provide the strongest indication of how the water manager defines either explicitly or implicitly environmental health and environmental flows.

The policy direction from CoAG (and more recently affirmed in the NWI) regarding river health has now been embedded by government as a policy goal of both the Queensland and New South Wales governments in their respective water management legislation (Department of Natural Resources (Queensland) 2004; New South Wales Department of Land and Water Conservation 1997). The policy processes adopted by each state government in meeting the water reform agenda differ, but the aim appears consistent across jurisdictions.

In Queensland, the *Water Act 2000* mandates the provision of sufficient water for the environment to ensure river health is maintained (Department of Natural Resources (Queensland) 2004). This is being achieved through the development of Water Resource Plans, which identify the social, economic and environmental goals for each catchment. The environmental goals aim to protect the health of natural ecosystems for the achievement of ecological outcomes (Queensland Government 2002). However, it is important to note that the Act does not define what ‘river health’ means. In New South Wales, the *Water Management Act 2000* (New South Wales Government, 2002) includes recognition that the health of the rivers has to be protected. As such the Act identifies a process whereby long-term objectives for river flows and quality are set for each river, and rules are developed to ensure river health is maintained (New South Wales Department of Land and Water Conservation 1997). Both of these legislative instruments from Queensland and New South Wales reflect the principles identified in the *National Principles for the Provision of Water for Ecosystems* and mandate the minimum level of environmental benefit that society is willing to accept as depicted by the horizontal line bb in Figure 2.7.

2.3.2. Evaluating flow management regimes for environmental outcomes

Translating the policy goal to changes in actual river management is a difficult process primarily because research into methods of analysing environmental outcomes from different water policies and management options had been largely neglected until environmental problems became evident in the late 1980s and early 1990s. While this research is now progressing, significant gaps in knowledge and conflicting views on methodologies still remain (Arthington and Zalucki 1998; Stewardson et al. 2002; Land and Water Australia 2009; Rogers 2006). Arthington et al. (2006, p. 1312) note that a key challenge is the ability to convert "general hydrologic-ecological principles and knowledge into specific management rules for particular river basins". In practical terms Arthington (2006) describes this as being able to advise interested stakeholders and water managers how much a river's flow regime can be altered before aquatic ecosystems degrade. Despite the lack of appropriate scientific knowledge, there remains a need to initially develop management procedures in the short-term to facilitate achieving the longer-term goal of ecological sustainability. Given this need both Queensland and New South Wales governments have adopted procedures for implementing the environmental flow policy goals. While these procedures have a similar basis, it is useful to consider the issues that arise when developing flow rules to improve river management for environmental outcomes.

Despite the divergent views on specific definitions, the *National Principles for the Provision of Water for Ecosystems* (Appendix B) cover: unregulated river flow; water in wetlands and underground water; water released from storages (including volumetric allocations); inundation levels in wetlands and water in transit for other uses; including where this water has a defined flow pattern to meet environmental needs (ANZECC 1996). Although the principles provide a framework for guiding policy direction they nevertheless lack clarity about how the specific outcomes of environmental flow regimes might be evaluated so that judgement on the policy goals can be undertaken.

The process of developing river flow rules requires an understanding of the current status of riverine ecosystems and the ability to analyse environmental outcomes of policy and management changes. Gippel (2001) proposed that this is a multi-disciplinary task including studies of stream ecology, hydrology, geomorphology, water quality, flow distribution and control, and hydraulic habitat. These broad theme areas form the basis either wholly or in part of the major techniques used in assessing the environmental health of river systems. Indeed the remaining knowledge gaps research over the last couple of decades have produced a substantial increase in knowledge. Tharme (2003) undertook a global review on EFA and unearthed 207 methodologies in 44 countries.

In the global context Tharme (2003) groups and describes the majority of EFMs into four broad groups (of six) types - hydrological, hydraulic rating, habitat simulation, and holistic methodologies (congruent with King et al. (2003). In reviewing the need for, and progress of, flow restoration and protection in Australian rivers Arthington and Pusey (2003) briefly examine issues surrounding environmental flow methods. While citing a considerable number of references that detail and contrast "environmental flow methods" Arthington and Pusey (2003) also identify and briefly explain a number of broader methodologies comprising the:

- holistic approach;
- building block methodology;

- expert panel method;
- scientific panel method;
- flow restoration methodology;
- benchmarking methodology; and,
- DRIFT (downstream response to imposed flow transformation).

Without attempting to describe these methods here, it is noteworthy that Arthington and Pusey (2003) stated they are all underpinned by the “natural flows paradigm” outlined in Poff and Allan (1997) and the basic principles of river corridor restoration outlined in Ward (2001). In summary, the aim of these methods is to “maintain or partially restore important characteristics of the natural flow regime required to maintain or restore the biophysical components and ecological processes of in-stream and groundwater systems, floodplains and downstream receiving waters” (Arthington et al. 2003, p. 381).

The rationale for the “natural flows paradigm” can be ascribed to the relationship between flows and ecological factors that reflect the health of the riverine environment. The flow regime is characterised by the quantity, frequency, timing and duration of flow events, rates of change and predictability/variability (Arthington and Pusey (2003). Richter et al. (2003, p. 207) refers to the flow regime of a river as the “master variable” because it influences the components of the river ecosystem e.g. fish populations and nutrient cycling. Stewardson et al. (2003) supports this noting that the decrease in natural flow variations of regulated rivers has raised concerns that pre-regulation ecological communities will not be maintained. Arthington and Pusey (2003, p. 389) assert that “rivers and their floodplains need their natural flow regime in all of its spatial and temporal variability to maintain their natural ecological integrity and long-term evolutionary potential”. This does not mean that improving flow regimes will meet all riverine environmental health objectives. It is expected that restoration work and other actions to improve outcomes will be undertaken by other programs. Bunn and Arthington (2002) reviewed a number of studies supporting the recognition of the important link between the flow regime of rivers and the ecology of rivers and floodplains. They contend that flow is strongly related to physical habitat, which is significantly related to biotic composition in identifying four guiding principles regarding the influence of flow regimes on aquatic biodiversity. They conclude in support of Poff and Allan (1997) that the natural flow regime is highly influential on the biodiversity of streams, rivers and their floodplain wetlands. Gippell (2001) likens the natural flow regime to the method of last resort and therefore the preferred indicator of environmental needs due to the limited understanding of the relationship between flow variability and biological processes. The issue of limited knowledge is supported by Marshall et al. (2001) who contend that the relationship between flow regime and biological processes is complex and understanding these relationships is constrained by a lack of research.

The adoption of a methodology underpinned by the “natural flows paradigm” is not straightforward, nor without weaknesses and criticisms. Richter et al. (1997) reported that a common criticism of most models and methods was a tendency to be simplistic and reductionist in relation to complex ecosystem processes. Gippel (2001, p. 4) puts the criticism very simply “hydrological processes alone do not sustain aquatic life”. Furthermore, there is a linearity problem where to the extent that ecological health is related to flow, providing a proportion of natural flow (however defined), *ceteris paribus* does not necessarily result in a proportional ecological response (Gippel 2001). Arthington et al. (2006) supports the focus on the 'natural flow paradigm highlighting that

> *"There is now general agreement among scientists and many managers that to protect freshwater biodiversity and maintain the essential goods and services provided by rivers, we need to mimic components of natural flow variability, taking into consideration the magnitude, frequency, timing, duration, rate of change and predictability of flow events (e.g., floods and droughts), and the sequencing of such conditions"*

Despite the knowledge issues and limitations raised above, there is a need to adopt methodologies to inform decisions on the allocation of water to the environment in whatever form the allocation may occur as these decisions arise in the reform process. In citing other work, Richter et al. (1997) pointed out that while there are weaknesses in focusing exclusively on flows and that this was unlikely to succeed, ecological health cannot be maintained without hydrological integrity. Therefore, river management regimes need to account for the natural flow paradigm.

A major difficulty for governments has been how to deal with the lack of understanding of the complex processes affecting riverine health and how to develop EFM regimes given current knowledge. This is particularly difficult given the disconnect between ecological science and water management decision making (Richter et al. 2006). To over come this disconnect Richter et al. (2003) propose a six step process to developing environmental flow (management) regimes within which the EFA knowledge sits:

Step 1: Estimating ecosystem flow requirements
Step 2: Determining human influences on the flow regime
Step 3: Identifying incompatibilities between human and ecosystem needs
Step 4: Collaboratively searching for solutions
Step 5: Conducting water management experiments
Step 6: Designing and implementing an adaptive water management plan

Richter et al. (2006) also describes a broadly similar process for developing EFM recommendations comprising (1) an orientation meeting; (2) a literature review and summary of existing knowledge about flow-dependent biota and ecological processes of concern; (3) a workshop to develop ecological objectives and initial flow recommendations, and identify key information gaps; (4) implementation of the flow recommendations on a trial basis to test hypotheses and reduce uncertainties; and (5) monitoring system response and conducting further research as warranted. These processes recognise the adaptive management loop needed to achieve the longer-term goal of ecological sustainability but also incorporate methods to initially develop management procedures in the short-term. The use of SCF information sits within this adaptive management framework and should its use prove positive further refinement could be undertaken within this framework.

Key challenges in the water reform process have included ascertaining what condition the rivers are currently in, and how develop management rules and to gauge the expected outcome from changing how rivers are managed. One method involves undertaking studies into the health of riverine ecosystems and then observing the outcomes of management changes through monitoring of ecological indicators. At issue with this approach is the length of time and resources required to make sufficiently robust findings. This also assumes that there is agreement on the methodologies that will provide a robust judgement of ecological health. The other method is to draw on the relationship between hydrological processes and riverine ecosystem health. There are also issues with this approach, particularly the imperfect link between hydrologic processes and ecological health (as noted above).

A pragmatic approach that appears to have been adopted by a number of jurisdictions is to initially use the relationship between ecological health and hydrologic processes to ascertain the "baseline" level of health and make the initial judgements on expected outcomes of changes to water management regimes. This is then augmented by a commitment to longer term research and monitoring of ecological indicators. This approach was explicitly recognised in the 2002 Independent Report of the Expert Reference Panel on Environmental Flows and Water Quality Requirements for the River Murray System. This report stated:

> *In the short to medium term, the benefits of river management actions are likely to be assessed by performance against hydrological outcomes and indicators. However, we emphasise that hydrological outcomes are only an interim performance indicator and that ecological outcomes and indicators must be used to measure the ultimate effectiveness of river management.*
>
> *…Ecological outcomes of improved river management should be assessed using ecological indicators such as those developed in the Sustainable Rivers Audit, the National River Health Program and the National Land and Water Resources Audit.* (Jones et al. 2003, p. 7).

Given the desktop nature of this part of the study, consideration needs to be given to the hydrologic indicators as a guide to environment benefits, how they are used and where they may be appropriate. This can be a complicated exercise. Gipel (2001) also comments that the number of parameters that can be used to describe hydrologic variability is extremely large. Stewardson and Cottingham (2002, p. 4) concur noting "application of the natural flow paradigm is problematic, particularly when selecting measures of hydrological variability to preserve in the regulated regime. Nevertheless, there are a number of studies proposing different "methods" for assessing river management strategies under the natural flow paradigm, each using ranges of hydrologic indicators.

Jones et al. (2003) use the project objectives set by the Ministerial Council to develop a set of desired ecological outcomes for the Murray River system. In assessing management actions they derived five eco-hydrological attributes for the system and subsequently identified threats and environmental flow requirements (EFRs) for these attributes. For each of these attributes (flow volume, flow distribution, flow variability, connectivity and water quality), threats and EFRs, a suite of hydrological indicators was used to assess the short-term benefits of river management options (see Appendix C Table C.1.).

In using these indicators to assess river management options Jones et al. (2003) adopt the decision rule shown in Table 2.1. This process is explained as a risk based assessment framework centred on answering the question: "If we do x, what is the likelihood or probability of having a healthy working River Murray System?".

Table 2.1: Decision rule to assess the benefit of river management options

Key system level hydrological attributes (percent of natural)	Probability of having a healthy working river
≥ two thirds	High
≥ half	Moderate
< half	Low

Source: Jones et al. (2003).

While there are differences in the detail, the approach which makes use of changes to hydrologic indicators in assessing river management options is not uncommon. Nor is it without criticism. Arthington et al. (2006, p. 1312) criticise the simple "rules of thumb" approach proposing that

> *"Such simplistic guides have no documented empirical basis and the temptation to adopt them represents a grave risk to the future integrity and biodiversity of the world's riverine ecosystems"*

The rationale by Arthington et al. (2006) is that such an approach runs the risk of river managers simply assuming that if the river has two-thirds of the median annual flow remaining then the ecosystem health is satisfactory. Recognising the difficulty in developing EFM regimes in particular due to the lack of appropriate data Arthington et al. (2006) develop a short term approach whereby classes of rivers are identified based on key attributes of flow variability and then EFM regimes are developed based on these classes. Along side this approach a commitment to longer term research to capture the appropriate data is required.

Richter et al. (1997) proposed the "range of variability approach" (RVA) as a method for setting streamflow-based river ecosystem management targets. Based on the "natural flows paradigm" the RVA involves characterising the pre-development streamflow record (historical record or modelled) using thirty-two hydrological parameters which form the basis of flow management targets. Examined from another perspective, this method uses hydrological parameters to assess river management options in order to attain sustainable aquatic ecosystems.

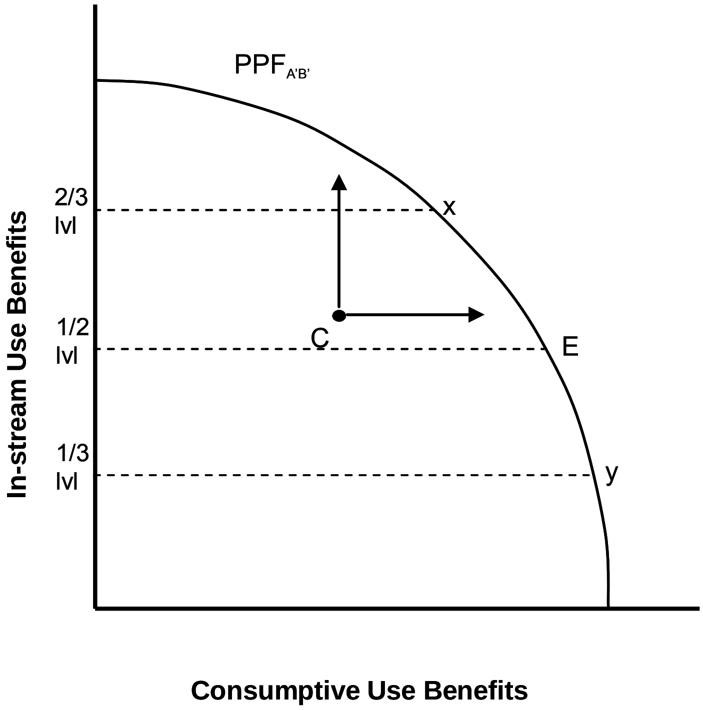
Stewardson and Cottingham (2002) propose a "flow events method" integrating a range of methods to assess the ecological impact of flow regulation and for developing environmental flow rules. In undertaking the analysis of the Broken River the appropriate indicators were identified based on the ecological outcome sought.

The staged approach, using preliminary assessments of river management regimes based on hydrologic indicators and a longer-term monitoring program in concert with further research into ecological linkages and impacts, is essentially the approach being adopted by jurisdictions in a range of water allocation planning exercises. While there are differences in approaches, each jurisdiction has adopted processes designed to allow a judgement to be made on how current water management arrangements are affecting riverine health. This allows them to make progress in developing flow regimes to either maintain or improve riverine health. These processes include the specification of a number of flow objectives that flow management regimes need to achieve and a number of indicators which are used to assess whether these objectives are likely to be met. The processes adopted by each state are based on the natural flows paradigm.

In Queensland a number of studies on methods of assessing river management regimes have been undertaken across various catchments. Each of these studies has considered impacts using a range of indicators, the components of which vary by location (amongst other factors). Lists and explanations of indicators for the Burnett catchment can be found in Brizga (2001), the Condamine catchment (Department of Natural Resources 2000), the Fitzroy catchment (Department of Natural Resources 1998) and the Border Rivers catchment (Department of Natural Resources and Department of Land and Water Conservation 2000). The catchment water resource plans (WRPs) then state the intended ecological outcomes for a catchment including environmental flow objectives and associated performance indicators (Queensland Government 2002).

In NSW the natural flows paradigm underpins the new water management regimes as outlined in a catchment water sharing plan (WSP) that must "protect the water source and its dependent ecosystems and establish environmental water rules" (NSW Department of Water and Energy 2007a, p. 5). In establishing these rules the environmental rules are designed to replicate natural flow patterns as espoused by the natural flows paradigm (NSW Department of Natural Resources 2009). WSPs also identify indicators and a monitoring regime used to determine whether the plan is meeting its objectives.

Acknowledging the criticism by Arthington et al. (2006) the decision rule used by Jones et al. (2003), or the target values for indicators used by Queensland and New South Wales, are advantageous in this research as they can be translated into production possibility frontier space where the percentages of natural key hydrological attributes can identify thresholds for risks to environmental health. For example, the three levels of hydrological attributes from Table 2.1 can be depicted as three trigger levels in trade-off space as shown in Figure 2.15 similarly to the proposals of minimum acceptable levels or thresholds displayed in Figure 2.7. That is the lines 1/3lvl, 1/2lvl and 2/3lvl are analogous to the line bb in Figure 2.7 and could represent minimum levels of in-stream benefits that society will accept. Once one level is selected further allocation of water to consumptive use would drive in-stream benefits below the minimum level and therefore would not be permitted. These thresholds may play a number of roles in analysing outcomes from policy proposals.



(Source: Jones et al. 2003)

Figure 2.15: Translating the decision rule into production possibility frontier space.

In the first instance the thresholds may be used to hypothesise points of change in the shape of the frontier itself. In this case point D in Figure 2.15 may be the point where the in-stream benefit response relationship changes, as shown in the complementary and supplementary graphs of Figure 2.8. In the second instance the threshold may be a level identified in water resource or water sharing plans below which a policy option is ruled out as it would be viewed as having an outcome resulting in an unacceptable level of environmental risk. In Figure 2.16 the line EE represents the minimum level of environmental outcome that will be accepted by decision-makers.

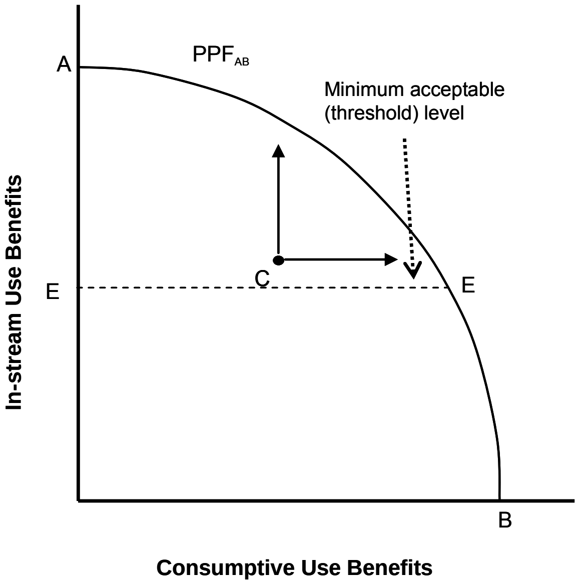


Figure 2.16: Using a set level of in-stream use outcome as a threshold between acceptable and unacceptable outcomes where below EE is unacceptable.

2.3.3. The environmental analytical framework to test the research hypothesis

In a study that is examining whether alternative flow management rules will be acceptable from an economic and environmental perspective, the methods adopted to gauge success or failure need to be physically driven, able to be used in the scenario analysis, and sufficiently robust to provide confidence in the results. While there are acknowledged shortcomings in relying on the natural flow paradigm, it does have a number of advantages. It is a reasonable indicator of the environmental impact of flow regimes, it is able to be run in a modelling framework, and it can therefore be used in scenario analysis. It has the added advantage of being able to be updated as new knowledge on more appropriate indicators come to light. Furthermore it is congruent with the decision-making process adopted by both Queensland and New South Wales in ongoing water reform processes. Thus, the methodology adopted in this research to estimate the impact of various river management regimes on riverine ecosystem health uses changes to hydrologic indicators of river flows which are compared to the values of hydrologic indicators under the natural flow regime. This is based on the assumption that if hydrologic indicators are moved closer to the values of the natural flow regime then riverine ecosystem health will improve.

The actual modelling framework will be detailed in the methodology section.

2.4. Seasonal climate forecasting

2.4.1. The interaction between SCF information and the production possibility frontier

The use of additional information to improve outcomes and produce a Pareto improvement occurs through providing more certainty in the knowledge of the supply of water as a factor of production. The outcomes from distributing the factor of production between the two competing uses is assessed using the PPF and has two aspects, short and long run.

In the short run the changing nature of water supply on an annual basis means that the location of the frontier in a PPF is constantly changing (Parkin (1990). In wetter years more consumptive use e.g. production of irrigated crops, will be attained (*ceteris paribus*) while in dryer years when streamflows are less, production will decrease. When making decisions on the annual distribution of water between the two uses, water managers estimate the available supply in the upcoming period (by combining knowledge of current supply and an estimate of future supply[10]) and make a distribution decision. This results in an outcome that is either on or within the PPF such as points C or D in Figure 2.17.

The highly variable nature of streamflows (Figure 1.2) translates to high variability in irrigation water supplies, particularly that component related directly to streamflows during the crop growing season. This uncertainty means that achieving a Pareto efficient level of production such as point C on the frontier may be unlikely. By definition, point D represents an inefficient allocation of water between the two outcomes and a sub-optimal Pareto outcome (Dudley 1997; Johansson 1991; Boadway and Bruce 1984). If SCF information is considered when the PPF is viewed as being short run, then the information is being used to estimate the water supply in the upcoming period. If sufficiently accurate, the outcome will be a movement of the frontier outwards and a movement in the starting point C or D outwards as well. This results in more production of both environmental and consumptive outcomes.

In this sense the PPF represents the case where the level of supply of the factor of production, water, is known in each year and the allocation decision is subsequently made. In a planning sense this is the correct view to take when evaluating the impact of using SCF information as the outcomes over time are much more important than the outcome for an individual year. Both economic and environmental outcomes should be assessed over the long-term to provide more certainty that the outcomes are robust and not the result of chance.

The use of SCF information relative to the long term view is that the information is used to make decisions that primarily impact on the upcoming period, matching the allocation decision to actual supply. This produces a more efficient outcome that is assessed using the PPF in the longer term. Therefore assuming a starting point of C (Figure 2.17) the use of SCF information will have been successful if the outcome achieved is moved from C towards the new frontier $PPF_{AB'}$. Should this movement stay within the area Cxy then we can conclude that a Pareto improvement could be achieved by using the SCF in managing water for environmental outcomes. That is, using the same amount of water and current systems, either or both, regional output and environmental flows will increase.

Seasonal climate forecasting provides information on the impact of climate variability on a range of variables which can assist in decision-making. To date the use of SCF in Australia has largely centred on the ability to forecast rainfall in the upcoming season for use in agricultural production. This use has increased as it became understood that the El Nino Southern Oscillation (ENSO) influences the climatic patterns over much of the world, including eastern Australia (Stone et al. 1992; Latif

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[10] See section 3 for an explanation of current water management decision-making.

et al. 1994). A useful summary of the development of SCF in Australia can be found in Stone and De Hoedt (2000). They note that the current capabilities in SCF have been the result of research dating back to 1929.

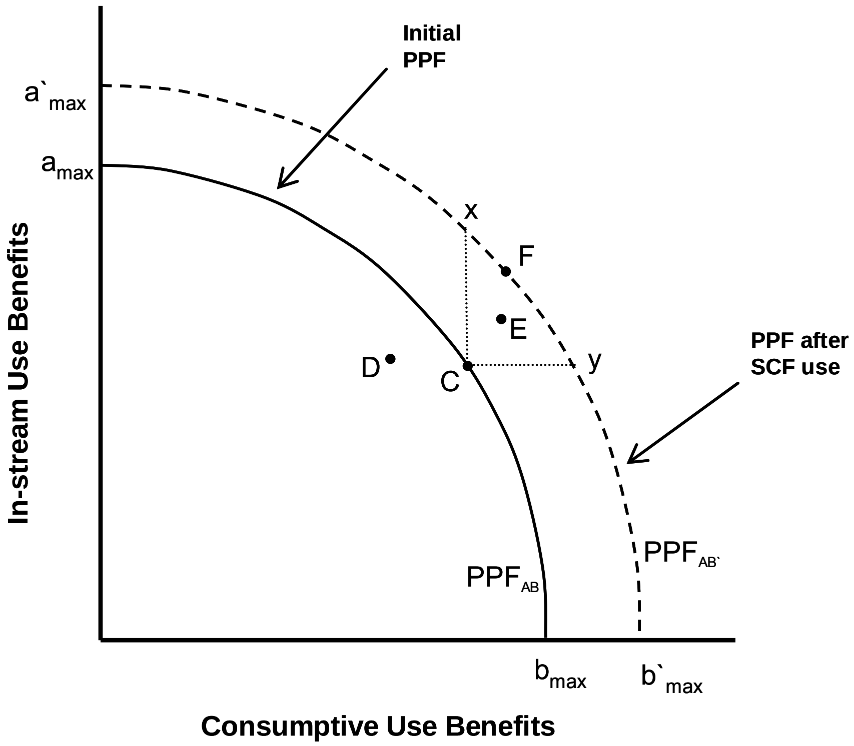

Figure 2.17: The impact of additional information in decision-making on the PPF.

Without delving into the scientific underpinnings of ENSO, a useful first step is to understand the general impact the ENSO phenomenon has on Australia. ENSO is related to the warming and cooling of sea surface temperatures in the eastern equatorial Pacific around the coast of Peru, and the consequential impact on rainfall and streamflow in Australia. An El Niño event generally produces lower than average rainfall and streamflow in parts of Australia. The inverse of an El Niño event, known as a La Niña, produces higher than average rainfall and streamflow in parts of Australia (Abawi et al. 2000).

The Southern Oscillation Index (SOI), which incorporates the sea-level-pressure difference between Papeete (Tahiti) and Darwin (Stone et al. 1992) measures the status of the ENSO phenomenon. The correlation between the SOI and rainfall can be used to forecast rainfall. There is a positive correlation between seasonal rainfall and the SOI in most of eastern Australia (McBride et al. 1983). Stone et al. (1992) used principal component and cluster analysis to group SOI values into five clusters or phases based on the magnitude of the SOI and on the direction of change in the Index. Consistently negative and rapidly falling SOI phases correspond with below median rainfall, while consistently positive and rapidly rising phases correspond with above median rainfall in eastern Australia.

2.4.2. Use of SCF information in agriculture

Given an ability to make seasonal forecasts of rainfall one challenge was to use this information to assist decision-making in industries where climate variability has a major influence. In the first instance this has occurred through the use of SCF information in agricultural decision-making (Stone and De Hoedt 2000) where research has been widespread, particularly in the United States (see for instance: (Mjelde et al. 1988; Mjelde et al. 1999; Mjelde 2002; Mjelde et al. 1997; Mjelde et al. 2000; Hill et al. 1999; Hill et al. 2000; Mjelde et al. 1998) and Australia.

In Australia Abawi et al. (1995) estimate that using a forecast of rainfall based on the SOI, wheat farmers can gain higher returns by adopting a strategy where farmers harvest the grain early and use mechanical drying in years forecast to be wet. Hammer et al. (1996) examine the use of SCF in tactical cropping decisions for nitrogen fertiliser and crop cultivar. Their results indicate that farmers can attain higher profits and/or reduce risk. Carberry et al. (2000) examine the use of SCF information in choice of crop rotations finding that farmers gain higher gross margins, and lower soil loss, but an increase in risk of financial loss. A range of applications are covered in Hammer et al. (2000) and Muchow and Bellamy (1991).

Despite these positive findings, it is cautionary to note that they have a range of uncertainty surrounding them; findings are often location and issue specific as forecast ability or skill[11] tends to vary spatially and temporally throughout the year. This notion is consistent with Mjelde et al. (1997) who in studying the use of Southern Oscillation information in corn and sorghum farming found Southern Oscillation forecasts to be useful to corn but not sorghum producers.

2.4.3. Use of seasonal climate forecast information in water resource management

In addition to information on probable future rainfall, SCF provides advance information on potential hydrologic regimes that may be used for environmental flow management. The rationale for this stems from two bodies of research. The first is the research into the link between the ENSO phenomenon and streamflow characteristics. Secondly, that into the potential uses of the findings from this research in water resources management.

In the last decade a number of studies have found strong relationships between streamflow variability and the ENSO phenomenon (Chiew et al. 1994; Dracup and Kahya, 1994; Kuhnel et al. 1990; Hamlet et al. 1999; Simpson et al. 1993). Subsequently the information has in some cases been developed into forecast systems for streamflow based on the SOI that aim to enable better informed water management decisions by providing advance information on some hydrologic features of rivers over a specified period (Abawi et al. 2001; Piechota et al. 1998).

For instance, Abawi et al. (2001) showed that the SOI, with a lead time of 3 to 6 months, is a useful predictor of streamflow. This was particularly the case when

[11] According to Katz and Murphy (1997, p. 31) skill refers to " the accuracy of the forecasts of interest relative to the accuracy of forecasts produced by a naïve forecasting system such as climatology or persistence".

comparing opposite phases of the SOI, including rapid fall vs. rapid rise, or consistently positive vs. consistently negative phases respectively. Figure 2.18[12] provides an example of the ability to forecast streamflows for the spring-summer period in the Border Rivers catchment of eastern Australia. It also provides some evidence for the potential use of SCF in water resource management. The distribution of streamflow volumes from October to February, using the consistently negative and rapidly falling SOI phases in September, was calculated by pooling the cumulative streamflows for all years in which the September SOI was classified as either consistently negative or rapidly falling, and then calculating the probability distribution of the respective events.

Figure 2.18 shows a consistent shift in the probability distributions indicating that when the SOI phase is either consistently positive (CP) or in rapid rise (RR), significantly more streamflow is expected during the spring-summer period. Conversely when the SOI phase is either consistently negative (CN) or in rapid fall (RF), the cumulative streamflow volumes between October and February are considerably lower. To put these results in some context, when the September SOI phase is CN or RF, the cumulative flow for October to February at Goondiwindi at 50 percent probability is approximately 209 000 ML as opposed to 356 000 ML when the SOI phase is CP or RR. Without this response we would expect to have a cumulative median (50 percent probability) flow of 289 000 ML. According to Abawi et al. (2001) this difference of 147,000 ML represents 49 percent of average annual irrigation diversions in the Border Rivers catchment for the 1994/95 to 2003/04 water years (see Table 3.2).

While studies have shown strong relationships between streamflow variability and the ENSO phenomenon, it has also been established that the strength of this relationship varies spatially and temporally. Chiew and McMahon (Chiew et al. 1994) found differences in signal strengths for ENSO in eastern Australia, while Piechota et al. (1998) found differences in forecast signal intra-annually and between ENSO events. Nevertheless, the conclusion by Piechota et al. (1998, p.3043) that "the ENSO streamflow links are real and significant, and not using the extra information provided by the ENSO indicators would be short sighted" remains valid and warrants further examination. This is congruent with Kiem et al. (2001, p. 715) who in assessing the ENSO-runoff link using a number of ENSO classification methods concluded that "significant subjectivity exists in the adoption of ENSO classification schemes". This implies that there may be value in assessing a range of forecast methods to determine which are more suited across space and time.

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[12] The data for this figure was sourced using the IQQM hydrologic model (see methodology) to model the natural streamflow at Goondiwindi.

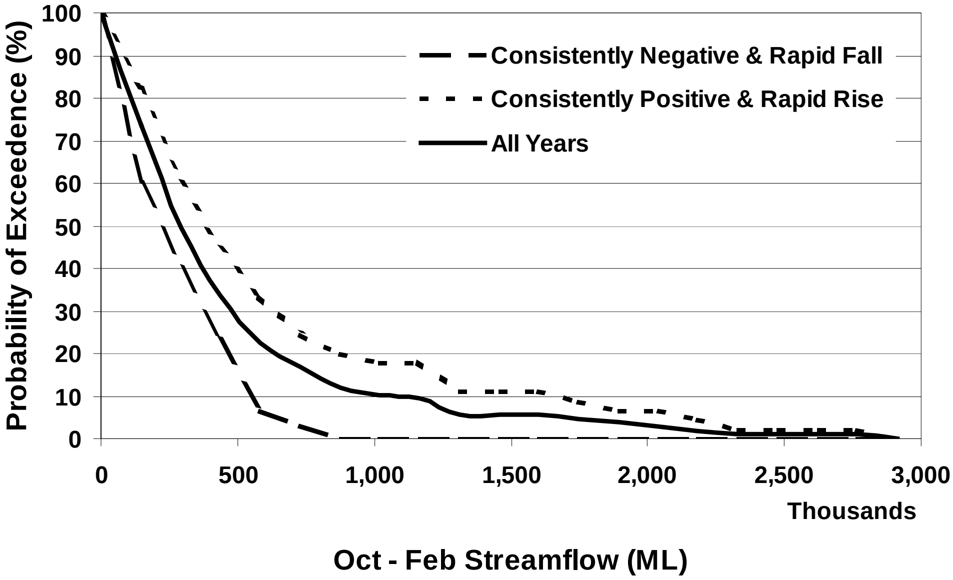


Figure 2.18: Forecast of Streamflow for October to February (1890 – 1996) based on the September SOI phases.

In this study three seasonal climate forecast methods all based on the ENSO phenomenon are tested in the trade-off analysis framework. At the most basic level is the partitioning of years into El Niño and La Niña episodes (Allan et al. 1996b) where generally less streamflow is expected in El Niño years and more in La Niña years. The second type of forecast is based on the Southern Oscillation Index (SOI) which is the most common measure of the ENSO on Australian rainfall. In simple terms a negative value of the SOI suggests lower than average rainfall (and streamflow) over most of eastern Australia while a positive value suggests higher than average rainfall (and streamflow) (Abawi et al. 2001). The third forecast type, SOI phases developed by Stone et al. (1992) uses groupings of SOI values into clusters or phases based on the magnitude of the SOI and on the direction of change of the Index. The consistently negative and rapidly falling SOI phases correspond with below median rainfall (and streamflow) while the consistently positive and rapidly rising phases correspond with above median rainfall (and streamflow) in eastern-Australia (Abawi et al. 2001; Ritchie et al. 2004). The SOI values and SOI phases are also tested using a number of different values, in the case of SOI values and a number of different lead time in the case of the SOI phases. Therefore, while this study is not intended to be a comprehensive assessment of the use of SCF information, it will nonetheless be an important examination of ENSO based seasonal climate forecasting tools.

Notwithstanding these issues, the potential to use SCF in environmental flow management stems from the relationship between the ENSO phenomenon (as measured by the SOI) and streamflow that has enabled streamflow to be forecast. To some degree this potential has been shown by the limited studies that have been undertaken on incorporating SCF information into water resources management. An early attempt to use climate predictions in water management was reported by Changnon (1986) where climate prediction information was the basis for making a decision on the management of two reservoirs in southern Illinois. Ex-post analysis showed the decision to be correct as the predicted above normal rainfall occurred. Although this study had positive results it remains a single event case study that in the absence of other research cannot be extrapolated to justify broader uptake of SCF information in decision-making.

Scientific studies with more robust findings have been undertaken by Chiew et al. (1999) and Hamlet et al. (2002). Chiew et al. (1999) investigated the potential use of SCF and streamflow persistence information in reservoir management for setting urban water restriction rules in Victoria, concluding that the use of SCF information has risks but that net benefits could be attained.

Hamlet et al. (2002) used scenario analysis to examine in both economic and environmental terms the ability for long-lead streamflow forecasts to be used in improving the operating performance of a reservoir in the US Columbia River Basin. The trade-off analysis undertaken is loosely akin to the analytical framework of this study, using the production possibility frontier (Figure 2.6). In Hamlet et al. (2002) net revenues represent consumptive uses (x axis), while the reliability of achieving other system objectives including minimum flows for fish is representative of environmental use benefits (y axis). However, it is noted that this objective set includes a number of irrigation objectives which are consumptive use objectives. By simulating changes to decision rules based on forecast information in a scenario analysis, they showed that annual net hydroelectricity revenue could be increased by approximately US$40 million per year (compared to a status quo) without significant impacts on other system objectives. The trade-offs were further highlighted by increasing net revenues for subsequent scenarios from hydroelectricity to around US$153 million per year for minor impact on other system objectives.

The current study extends the approach of Hamlet et al. (2002) from a situation where the consumptive use benefits accrue to a single entity generating hydropower to a situation where use benefits are less directly linked to reservoir operations and instead accrue to a number of entities (i.e. irrigators whose benefits are also related to the biophysical cropping system). Additionally, incorporating consumptive use objectives (irrigation) with in-stream objectives obscures understanding of the trade-offs as allocations to different users change. This aspect will be overcome by adhering strictly to a consumptive use – in-stream use comparison.

Despite positive findings, the challenge to make use of the information provided by SCF remains one of the key issues to be addressed. As pointed out by Piechota et al. (1998) the influence of ENSO on streamflow is now well documented. Furthermore, the potential for the use of SCF information is also well recognised. For example, when examining the policy implications of climate forecasts for water resources in the US Pacific northwest, Callahan et al. (1999, p. 269) note "climate forecasts have the potential to improve water resource management in this system supporting management decisions that decrease its vulnerability to droughts, floods and other crises related to climate variability". This conclusion was based on a study demonstrating that climate forecasts are under-utilised by managers. Interviews were conducted with thirty-one forecasters and water managers from twenty-eight organisations with wide ranging roles and responsibilities, including hydropower, environmental management, flood control and irrigation management. The authors concluded that while potential forecast utility was high, the information was rarely used for operational decision-making (Callahan et al. 1999). Forecast information was used as background information. While the potential is recognised, it has not translated to high rates of adoption of SCF information in decision-making (Callahan et al. 1999; Hartmann et al. 2002). In Australia, Long and McMahon (1996) also found limited direct use of forecast information in water management decision-making, but some use as qualitative background information.

An alternative approach to assessing the use of climate information in water resource management was undertaken by Ray (2004). Rather than focussing on the actual use of climate information in operational decision-making Ray (2004) broadened the definition of 'use' into three categories. Type 1 use occurs when the information is consulted but not utilised any further. Type 2 use refers to situations where the information is considered in the process of making decisions and Type 3 use transpires when the information is incorporated into operational models for decision-making. In studying the interaction of climate information and reservoir management Ray (2004) sought to understand the context of management decisions being made and the level of flexibility reservoir managers had in making decisions. In undertaking this Ray (2004) studied the use of climate information in decision-making and ascertain how to improve the use of such information. The findings by Ray (2004) are broadly consistent with those reported above with most use found to be Type 1: consultation, with little Type 2 consideration. Types 2 and 3 were lacking primarily because climate information products were not available at that location or not compatible with operational decision-making procedures.

The conclusion from these findings is that a considerable challenge remains in assessing the usefulness of SCF information in water management decision-making. One step in this challenge is to continue to examine the usefulness of SCF information in case study situations. In undertaking these studies it is important to incorporate the issues raised in the above review. In particular, the need to examine a range of ENSO classification or forecast methods, the impacts of spatial and temporal variability on outcomes, and the subsequent conclusions on forecast efficacy, are critical.

2.5. Conclusion

The problem being considered in this research is one related to meeting the challenges and constraints identified by Schofield et al. (2003). Relating the decision problem to the production possibility frontier is a practical method of visualising the ultimate goal identified by ARMCANZ to achieve the "highest and best value of the limited resource for community benefit whilst ensuring that use of the resource is ecologically sustainable" ARMCANZ (1995, p. i). This visualisation highlights the trade-offs that may be outcomes from the use of SCF information, but also highlights that conceptually at least win-win situations are possible.

The visualisation also highlights the multidisciplinary nature of the problem being studied. Not only is this a function of the potential economic-environmental trade-offs inherent in the ARMCANZ goal but also related to what may be deemed practical decision-making. As will be shown in the discussion, in addition to ascertaining methods to measure the economic and environmental outcomes from the use of SCF information, a range of related issues need to be considered prior to the information being adopted in decision-making. The key point being that focusing solely (or too strongly) on one discipline (or outcome) alone runs the risk of treating the other discipline (or outcome) in a cursory manner. This would undermine any findings of the research that may indicate there is merit in using the SCF information in environmental flow management decision-making.

Given the problem being studied, and the decision framework, the review canvassed analytical approaches concluding that threshold value analysis in conjunction with an estimation of the environmental outcomes represents a theoretically robust assessment framework. It is also pragmatic in that these tools are used by decision-makers responsible for water management decisions. This implies that results are more likely to be considered for adoption or further research. The following chapter provides an overview of the study catchment and provides information on the approach to water management. The fourth chapter builds on the findings of the literature review and outlines a methodology that is robust in theoretical underpinnings, yet is to an extent pragmatic in providing a sufficient level of complexity essential for decision-making.

3. The study area – Border Rivers catchment

3.1. Location and population statistics

This study was carried out in an area of the Border Rivers catchment (BRC) in the northern part of the Murray-Darling Basin (Figure 3.1). The BRC stretches from the western side of the Great Dividing Range to Mungindi and has a total area of 49,500 km^2 (Department of Natural Resources and Department of Land and Water Conservation 2000) draining the northern NSW New England Tablelands and southern QLD. The catchment population was estimated to be approximately 59,000 (NSW Department of Water and Energy 2007b) which, typically of rural areas in Australia, has decreased by some 5 percent over the period 1991 to 2006. The importance of agriculture is indicated by employment statistics, with the proportion of employed persons in the agriculture forestry and fishing sector at 28.8 percent. This is compared to the next largest sector of retail trade at 13 percent (Frontier Economics 2007).

Joint management is a feature of the BRC as it straddles the Queensland – New South Wales border with around 51 percent or 25 580 km^2 of the catchment in NSW (Hope and Bennett 2003). The specific study area was confined to the broad acre irrigation area on the NSW side of the catchment between Goondiwindi and Mungindi, to eliminate jurisdictional variables. Data is also available for this area, as will be further discussed.

Figure 3.1: Location of the Border Rivers catchment, Australia.

3.2. *Production Characteristics and Water Use*

The BRC gross regional product (GRP) has historically been heavily reliant on agriculture, particularly beef cattle, irrigated cotton, cereal crops and dryland cereal crops. Queensland Treasury (2008) estimated that in the Darling Downs and South West regions[13] agriculture, forestry and fisheries respectively were responsible for 15.4 and 32.5 percent of the 2005-2006 GRP. Broadacre irrigated cotton is grown mainly on the plains between Goondiwindi and Mungindi, and across the whole region the main cereal crops are wheat, sorghum, barley and oats. In more recent times irrigated small crops such as grapes, stone fruit, vegetables and apples grown in the upper reaches of the catchment in the Granite Belt area around Stanthorpe have become more important.

In 1996/97 the value of agricultural production for the entire BRC was $824 million with irrigated agriculture accounting for 32 percent ($271 million). By 2001 value of agricultural production had grown to $1.045 billion. While the value of irrigated agriculture in 2001 was not ascertained irrigated cotton accounted for 83 percent of the 147,736 hectares of irrigated agriculture in the Border Rivers Catchment underlining its importance (Frontier Economics 2007). From an irrigation perspective, irrigated cotton is the most important single crop estimated to account for approximately $168 million (62 percent) (Hope and Bennett 2003). The value of intensive small crops has been estimated at $70 to $90 million (Department of Natural Resources and Department of Land and Water Conservation 2000). In 2001 the total value of agricultural production in the BRC was $1.045 billion, comprising $723 million of crop production, of which cotton accounted for $364 million or approximately 50 percent (Table3.1). Cotton is the primary irrigated crop accounting for 83 percent of the 53,900 hectares of the area developed for irrigation in 1997/98. The production of irrigated cotton in this catchment has grown rapidly since the late 1980s from approximately 150,000 bales in 1989/90 to in excess of 460,000 bales in 1999/2000 (Beeston 2000). These statistics are now somewhat dated as they are not often captured at the catchment level, but cotton is reported to remain the major crop (Ashton and Oliver 2008).

Table3.1: Value of agricultural production in the Border Rivers catchment 2001 ($,000).

Production type	$'000	$'000
Crops		723,026
Cereals	202,260	
Cotton production	364,470	
Livestock products		73,870
Livestock slaughterings		248,202
Fruit		43,430
Total		**1,045,098**

(Reported in Frontier Economics (2007), Source: ABS 7117.0.30.001 (2003)).

[13] The Darling Downs and South West are defined as the Australian Standard Geographical Classification 2006 Statistical Divisions.

As the catchment straddles two states, Queensland and New South Wales, access and use of the water resource is managed by an agreement developed by the two state governments (DNRM 2002; Queensland Department of Natural Resources and Water & New South Wales Department of Water and Energy 2009). The major users of water in the catchment include irrigated agriculture, stock watering and town water supply. Irrigated agriculture is by far the biggest water user in the catchment responsible for approximately 98 percent of diversions as shown in Table 3.2. The table summarises data from the annual water audit monitoring reports from each state jurisdiction to the Murray-Darling Basin Commission and illustrates considerable variation in diversions, but strong similarities in proportions. The accuracy of data is variable and each report contains estimates of the accuracy of diversion estimates which can be between plus or minus 18 percent of total diversions.

Table 3.2: Diversions in the Border Rivers catchment through time.

Year	Irrigation diversion (GL)	Other diversion (GL)	Total (GL)	Irrigation diversions as a percent proportion of total water
1994/95	110	3	113	97.3 percent
1996/97	292	5	297	98.3 percent
1998/99	293	3	296	99.0 percent
1999/00	345	4	349	98.9 percent
2000/01	517	3	520	99.4 percent
2001/02	344	5	349	98.6 percent
2002/03	200	4	204	98.0 percent
2003/04	312	4	316	98.7 percent
2004/05	296	6	302	98.0 percent
2005/06	263	3	266	98.8 percent
2006/07	200	5	205	97.5 percent

(Sources: (MDBC 2001; MDBC 2006b; MDBC 1997; MDBC 1998; MDBC 2000; MDBC 2002; MDBC 2003; MDBC 2005; MDBC 2008; MDBC 2007; MDBC 2006a)

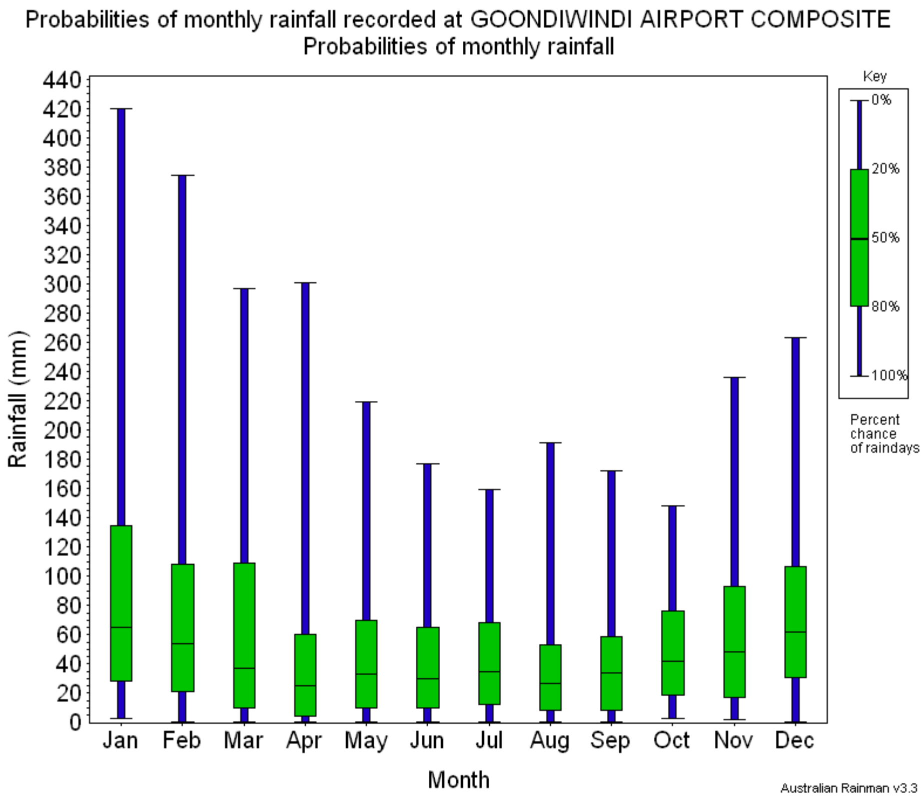
3.3. Water availability and the river system

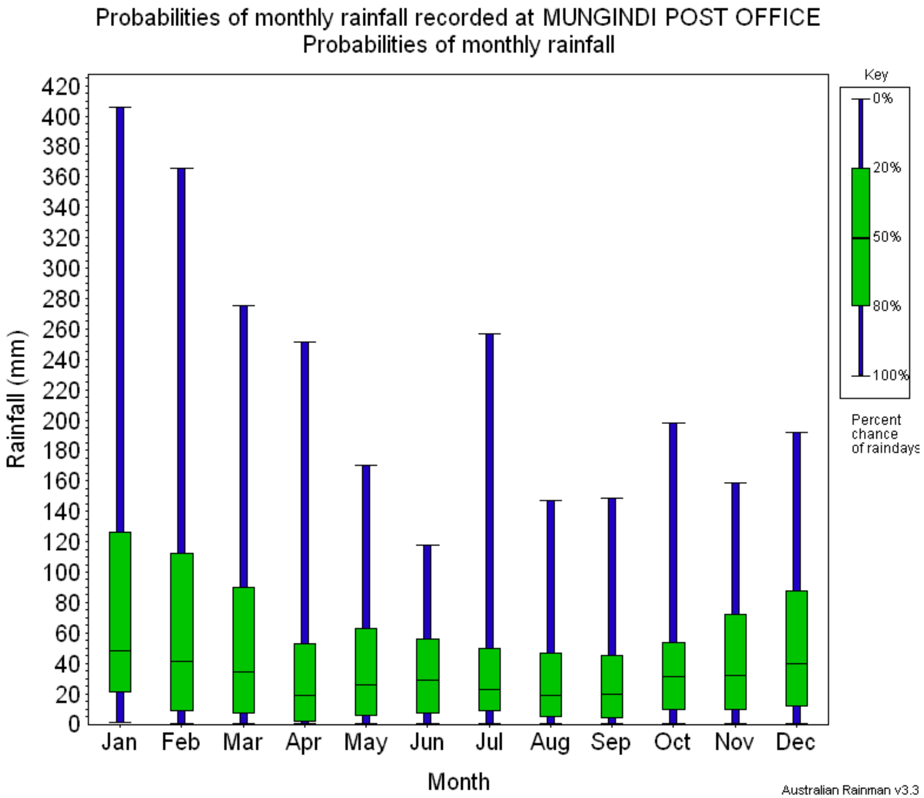
As previously noted, rainfall in Australia is highly variable and the BRC similarly shows a highly variable rainfall pattern. Figure 3.2 highlights this for two rainfall stations: Goondiwindi, located in the central area of the catchment; and, Mungindi in the western end of the catchment. As median rainfall decreases in a westerly direction with Stanthorpe in the east having a median rainfall of 753 millimetres, Goondiwindi 609 millimetres and Mungindi 514 millimetres (Source: Australian Rainman). This pattern is confirmed in which compares the rainfall statistics for Stanthorpe in the eastern part of the catchment, Goondiwindi in the centre and Mungindi in the west. Despite the slight trend of summer dominance shown in Figure 3.2, the importance of irrigation water supply is emphasised by the variability of rainfall and the relationship between rainfall and evaporation shown in Figure 3.3 (Abawi et al. 2001).

Table 3.3 Comparison of rainfall across the Border Rivers Catchment

Monthly rainfall recorded at STANTHORPE POST OFFICE													
Statistical summary													
	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Year**
Mean	97	87	66	43	47	47	50	43	52	69	74	94	770
Median	87	71	51	31	37	37	41	35	45	61	67	81	753
Standard deviation	60	65	53	41	40	40	37	42	42	44	42	59	205
Highest on record	259	305	311	242	232	246	205	365	236	238	255	323	1835
Lowest on record	5	0	0	0	0	0	0	0	0	6	0	2	368
Mean raindays	10	9	9	7	7	7	8	7	7	8	8	10	97
No. of years	127	127	127	127	126	126	126	126	126	126	126	126	126
Monthly rainfall recorded at GOONDIWINDI AIRPORT COMPOSITE													
Statistical summary													
	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Year**
Mean	81	71	58	37	44	40	42	34	38	49	59	71	623
Median	65	54	37	25	33	29	34	26	34	42	48	62	609
Standard deviation	64	59	57	44	42	35	34	31	34	34	47	47	170
Highest on record	420	374	297	301	219	177	159	191	172	148	236	263	1035
Lowest on record	3	0	0	0	0	0	0	0	0	3	2	0	265
Mean raindays	7	6	5	4	5	5	6	5	5	6	6	7	67
No. of years	121	121	121	121	120	120	120	120	120	120	120	120	120
Monthly rainfall recorded at MUNGINDI POST OFFICE													
Statistical summary													
	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Year**
Mean	72	64	51	32	36	34	34	26	27	38	42	51	507
Median	48	41	34	19	26	29	23	18	20	31	32	40	514
Standard deviation	65	70	52	39	34	28	36	26	25	34	36	43	162
Highest on record	406	366	275	251	170	118	257	147	149	198	159	192	1074
Lowest on record	1	0	0	0	0	0	0	0	0	0	0	0	171
Mean raindays	6	5	4	3	4	4	5	4	4	5	5	5	54
No. of years	113	113	113	113	112	112	112	112	112	112	112	112	112

(Source: Australian Rainman).





(Source: Australian Rainman).

Figure 3.2: Rainfall probabilities for Goondiwindi and Mungindi.

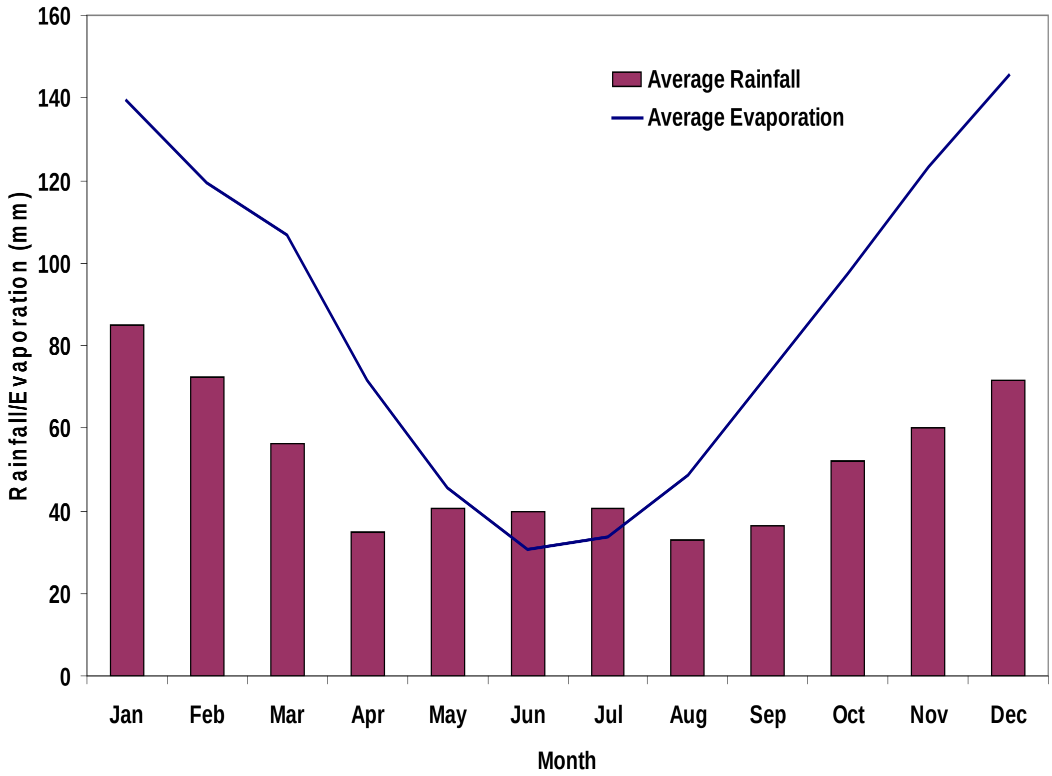


Figure 3.3: Relationship between mean monthly evaporation and rainfall in the Border Rivers catchment.

The outcome of the rainfall–evaporation relationship is a crop water supply deficit over all months of the year (on average), with the exception of June and July. As noted in chapter 1, the water supply deficit is not static and is expected to worsen as the impacts of climate change become more apparent. While this is an issue for all cropping, it is particularly important for summer crops such as cotton. To allow consistent cropping, irrigation is the key method of filling this deficit, which necessitates storage of large volumes of water to match supply to crop water requirements.

The river system in the BRC comprises three subsystems: the Macintyre Brook; Severn-Macintyre; and, Dumaresq-Macintyre-Barwon as shown in Figure 3.4. The primary rivers are the Macintyre River, Macintyre Brook, Dumaresq River, Servern River and the Weir River.

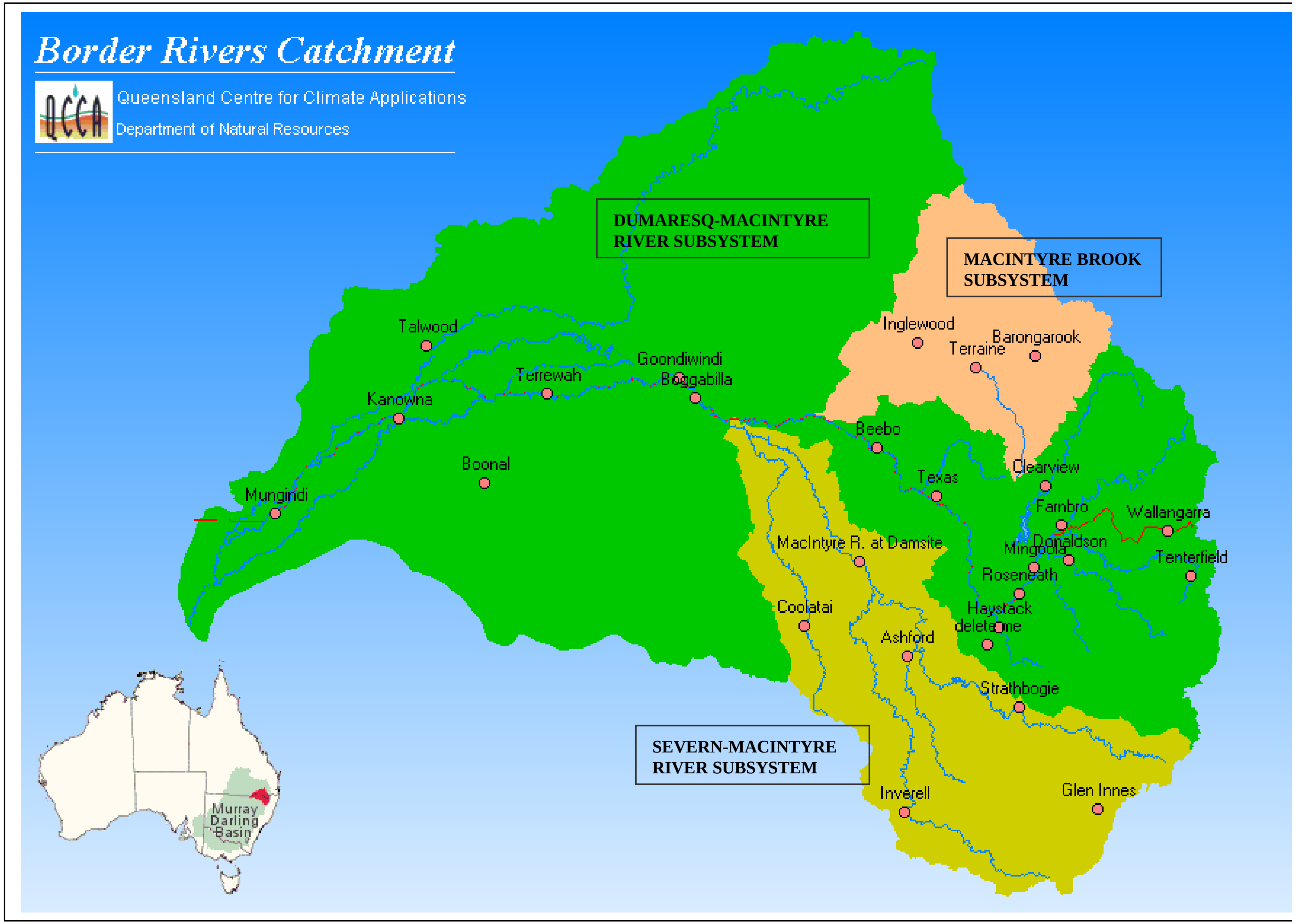


Figure 3.4: Border Rivers catchment.

Water for consumptive use is sourced from state owned dams and weirs, privately owned on-farm dams, and within system flows not controlled by state owned infrastructure. A considerable volume of water is stored and managed in the three state owned dams with a storage capacity of 647.6 gigalitres made up of:

• Coolmunda dam	75.2 gigalitres
• Glenlyon dam	256 gigalitres
• Pindari.dam	312 gigalitres
• Bogabilla weir	6.1 gigalitres

(Abawi et al. 2001).

In addition to water stored in major dams the privately owned on-farm dams have a storage volume of 287.9 gigalitres. These on-farm storages range in size from 6,000 to 44,900 megalitres (Abawi et al. 2001) and form an integral part of the infrastructure system developed to overcome the crop water supply deficit (Department of Natural Resources and Mines (Queensland) 2004).

The BRC with the importance of agricultural production, its climatic driven crop water deficit and subsequent water storage capacity, is an appropriate location to test the research hypothesis. The other key factor that needs to be considered is whether the water allocation and management arrangements are conducive to being manipulated as required for this study.

3.4. Water management and allocation

Water management in the BRC is to some extent complicated by the fact that two states have jurisdiction over parts of the catchment. Due to this, in 1946 an agreement between these two states established a management arrangement for sharing and managing the water resource. This agreement has been revisited and an intergovernmental agreement has been developed to ensure that the catchment water resources are managed sustainably for “environmental, social, cultural and economic values”. In broad terms the intergovernmental agreement sets out provisions for the sharing of water between the two states, common environmental flow rules, water allocation and access, interstate trading, and coordinated monitoring and reporting (NSW Department of Water and Energy 2007b). The water resource planning processes currently underway in both Queensland and New South Wales are consistent with the Inter-Governmental Agreement (IGA) and aim to achieve the intended outcomes of the water reform agenda of ARMCANZ and the National Water Initiative.

In Queensland the reform process is being achieved by developing a Water Resource Plan (WRP) under the *Water Act 2000*. The WRP for the Border Rivers catchment has been finalised and is available at http//:www.nrw.qld.gov.au. The social, economic and environmental objectives and outcomes specified in a Water Resource Plan for an area are achieved through a Resource Operations Plan (ROP). The BRC ROP, released in draft form in January 2007, addresses a number of issues including detailing the operating rules for infrastructure operators so management of dams and weirs complies with the Water Resource Plan’s objectives for water users and the environment (Department of Natural Resources and Mines 2009; McLennan 2008).

A detailed explanation of the water management rules for the Queensland component of the BRC can be found in the Water Resource Plan for the Border Rivers Catchment (DNRM 2002; DNRM 2003).

NSW is similarly undertaking a water management planning exercise under the *Water Management Act 2000* to establish a management regime to share water in the NSW section of the BRC in a manner that will achieve environmental, economic and social outcomes. Details of the water management rules for the NSW component of the BRC can be found in the Draft Water Sharing Plan NSW Border Rivers Regulated River Water Source (NSW Department of Water and Energy 2007b; NSW Department of Water and Energy 2007c; NSW Department of Water and Energy 2007a).

Within the overarching water management arrangements an explanation of the water management regimes is useful background knowledge as it may impact on the interpretation of study results. This section discusses the range of water sources available to an irrigator, making some comment on their relative importance in the context of this study. The following section briefly outlines the changes that have been made to the management arrangements. More detailed information can be found from the NSW Department of Water and Energy (NSW Department of Water and Energy 2007b; NSW Department of Water and Energy 2007c; NSW Department of Water and Energy 2007a).

3.4.1. The irrigation water supply equation

A water licensing system controls access to water from the government owned water supply schemes in the BRC. The relevant state government agency licences usage of the water resource, Department of Natural Resources and Water in Queensland and Department of Natural Resources in New South Wales. From the perspective of an individual irrigator, the water licensing and operational rules affect farming decisions such as how much area to plant to irrigated crops. The water supply (WS) equation facing an irrigator can be summarised as:

WS = On-allocation + Off-allocation[14]+ Overland flow + On-farm storage + Rainfall + Stored soil moisture.

Of the water supply variables, on and off-allocation and, to a certain degree overland flow, are controlled by the state water licensing system. The licensing system in both states can be broadly divided into two levels: high security; and, general security. These security levels align with the ranking of allocation for a given supply. High security users mainly include town, industrial, stock and domestic water users, and some high value mainly horticultural crops are allocated 100 percent of licensed volume in all but extreme drought years. General security licences are allocated a water supply for the year according to a range of management rules in each state and have less surety of supply than high security users. Broad acre irrigated agriculture, such as irrigated cotton, is typically a general security water user (Abawi et al. 2001).

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[14] Off-allocation is now referred to in New South Wales as “supplementary water” and in Queensland as “unsupplemented water”.

General security water licences, hereafter referred to as licences, are specified in terms of a nominal allocation that represents the maximum volume of water (in megalitres) a licence holder may have access to from the state owned[15] dams in any given water year, which in the BRC runs from October 1 to September 30. This water supply volume, termed "on-allocation" water, is known to the irrigator at the start of the irrigation season because the state water authority announces the percentage of nominal allocation that irrigators will have access to in that year. For example an irrigator with a 1,000 ML licence may be advised that they can access 50 percent or 500ML of their nominal licence volume. For the irrigator this amount in all but extreme drought years represents the minimum amount of water supply they can access during the water year. If inflows occur water authorities review storage levels and increase allocations according to the levels of inflows received.

Despite significant development of water storage infrastructure (both public and private) in the BRC during the last two decades, water remained a limiting factor in a considerable number of years for irrigated agriculture. Rather than using only the volume of licensed water already in storage for irrigation during the growing season, a large proportion of the water used for irrigation in this region is sourced from high streamflow events during the growing season. Access to this additional water supply, termed "off-allocation", may be granted to water-licence holders when dams spill or high flows enter the river system. State authorities announce this access only when all other user needs (including environmental) have been met and any such access is independent of an irrigator's annual licensed volume. However, the amount of "off-allocation" able to be accessed by each irrigator is capped for the water year. From an irrigators perspective while this water represents a significant proportion of the manageable water supply, the amount for the upcoming period is unknown because access is directly related to flow events at a point in time.

In the BRC, off-allocation water accounts for a considerable amount of water diverted for irrigation. In comparison to the total of 643,000 megalitres of storage in Coolmunda, Glenlyon and Pindari dams, on-farm storage in NSW and Queensland now accounts for 155,000 and 300,000 megalitres of storage respectively. This development is primarily due to the importance of off-allocation water in the total water supply for irrigators (NSW Department of Water and Energy 2007b). For example, an examination of the measured relative use from each source, on- and off-allocation, for the water years 1991 to 1999 for the New South Wales part of the BRC shows that over this period off-allocation provided the majority of water used. On average, off-allocation accounted for 55 percent of water diverted with the remaining 45 percent obtained from on-allocation (Figure 3.5).

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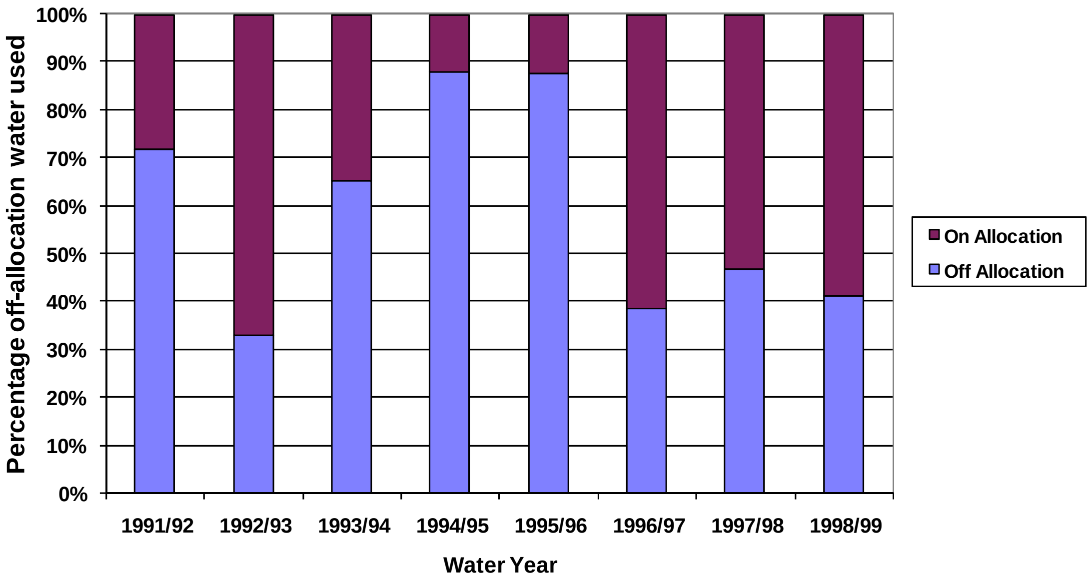
[15] Note 'state owned' is used loosely here as some corporatisation of water management has occurred.



(Source: Abawi et al. 2001)

Figure 3.5: Amount of on and off allocation diversions for NSW irrigators in the Border Rivers based on recorded data 1991-1999.

Overland flow is similar to off-allocation water in that it is related to high rainfall events however it refers to water, including floodwater, that flows over land otherwise than in a watercourse or lake. Similarly to off-allocation and rainfall, its volume for the upcoming season is unknown in advance. At the beginning of the water year in the BRC the known variables are restricted to on-allocation, on-farm storage and stored soil moisture.

3.4.2. Study and current water management arrangements

As part of the ongoing water reform arrangements, changes have been made to how water is operationally managed in the BRC. As such there is some difference between the current management arrangements for water in the BRC and that used in this study. The management arrangements used in this study were those in force in the later 1990s. While the management changes would have some impact on the use of forecasting in off-allocation management, they have no impact on assessing whether under a set of rules the forecasting technique is sufficiently accurate to achieve the set objectives.

In the 2001/02 water year the overarching water management regime in the BRC changed from what is termed "annual accounting" to "continuous accounting" (MDBC 2003). While the licensing arrangements remained the same, as explained in the previous section, this change effects how water is allocated amongst licence holders in any given water year. In an annual accounting system the accounting and allocation or sharing of water resources occurs on an annual basis. Announcements on the availability of water to entitlement holders are made at the beginning of the water year and updated periodically to account for changes to water supply levels.

Water availability is determined by summing the volume of water in storage at a point in time, water use since the start of the water year, estimated recession inflows, supplementary downstream tributary inflows and expected minimum inflows. Deducted from this total are high security requirements, reserves, operational losses, estimated evaporation losses and transmission losses. The total volume is divided by the total entitlement (i.e. sum of all licensed volumes) of the BRC and multiplied by 100 to calculate the proportion of the licence volume that licence holders may access in that water year. This is the announced-allocation or on-allocation and may be increased during the water year as inflows to the storages occur. However as pointed out by Jayasuriya (2004) the use of historical minimum inflows means that there is approximately a 1 in 100 chance that the announced allocation may be reduced during the year if the minimum historical inflow does not occur. Should this happen the initial estimate of water availability will be too large and availability may be reduced.

In addition to the announced-allocation, individual licence holders may have access to a proportion of their allocation from the previous year in cases where the complete entitlement for that year was not used. This is termed carry-over, and its use is subject to a number of rules that either discount the amount of carry-over water that may be used, or reset the carry-over volume to zero.

Continuous accounting is an extension of annual accounting where carry-over has a maximum limit set but there are no discount or reset rules and water is apportioned amongst entitlement holders periodically throughout the water year or as inflows occur. Carry-over of unused allocation is managed whereby in essence licence holders have an account where water not used in one year can be carried into the next year, but as dam inflows occur and more water is credited to their account it can only be credited up to a set proportion (105 percent) of the licensed entitlement. Once this limit is reached, any further inflows spill into the accounts of other licence holders that have used more of their allocation (Podger 2006; MDBC 2003; NSW Department of Water and Energy 2007b). Implementation of the Queensland resource operations plan and the NSW water sharing plan both involve a transition to a continuous accounting water management system (NSW Department of Water and Energy 2007b; Department of Natural Resources and Water 2007).

The change from annual to continuous accounting is accompanied by a change in the IQQM hydrologic model used for the BRC to account for the variations in operational procedures. The version of IQQM used in this study however models the annual accounting situation and any explanation of the IQQM and its functions contained herein refers to this version of the IQQM. The reason for this is primarily related to the ease of altering the IQQM to incorporate rules for testing the hypothesis. These rule changes are outlined in chapter 4. An additional reason relates to the sensitive nature of the on-going water reform process. Nonetheless seasonal climate forecasting is, in this study, about providing information on future supply, which is unknown. Wherever water captured within the cropping season is required to grow the crop in that season, forecast information is potentially applicable. Therefore, changes to water management procedures in different version of IQQM do not alter the potential usefulness of SCF information but may mean that the SCF information would be used in a different manner.

The implications for this study from the use of the annual-accounting version of IQQM are judged to be relatively minor because the key water supply variable used in this study, off-allocation[16], is largely unaffected by the change to continuous accounting. Also, while the strength of seasonal climate forecasting signal varies spatially, the change away from annual-accounting regimes is not universal and as such results may be applicable at other locations, subject to the usefulness of SCF information in those locations.

3.5. The study site

While the study is within the BRC, the specific location is the irrigation area on the NSW side of the catchment between Goondiwindi and Mungindi (see Figure 3.6). It was decided to restrict the study to this location primarily because a case study was needed to allow the IQQM hydrologic model to be customised in a manner that enabled the hypotheses to be tested. This aspect will be explained in detail in chapter 4.

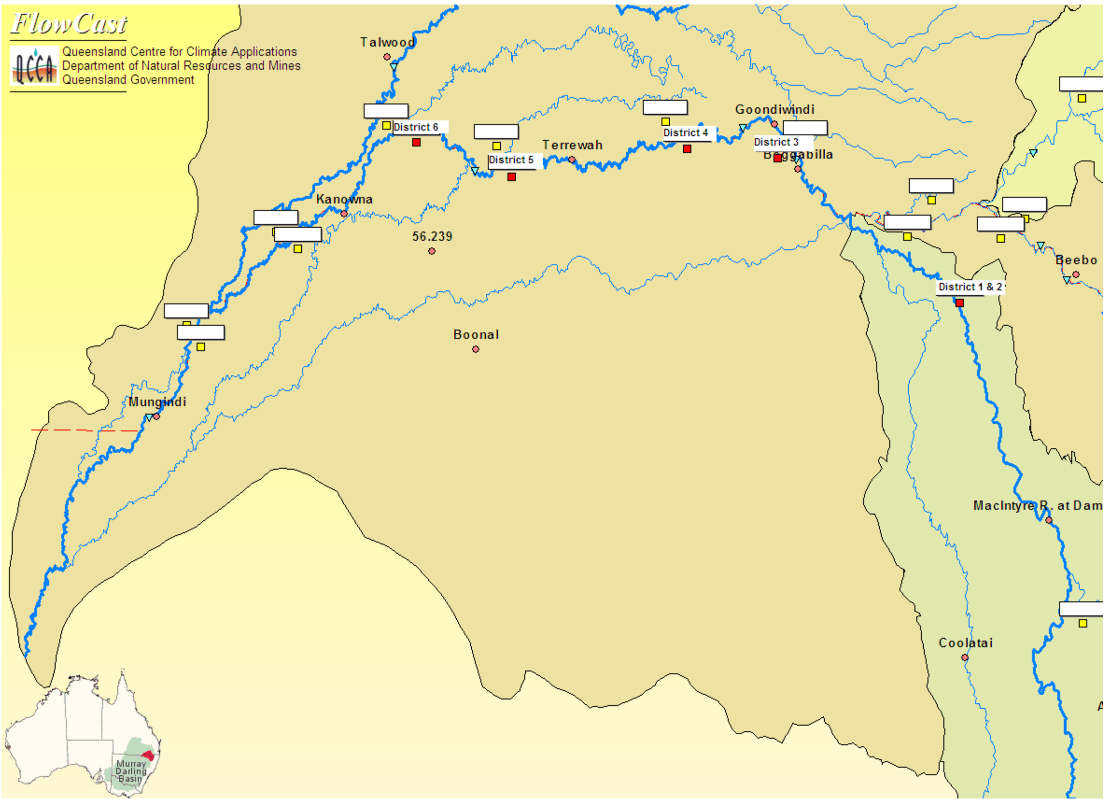


Figure 3.6: The specific study site within the Border Rivers catchment.

Restricting the study to a proportion of the overall catchment simplifies the modelling and interpretation of results. This minimises the changes that need to be made to the hydrologic model. Interpretation of results is also less complicated as the site chosen sits within one particular section of the regulated river system minimising the impact that other unregulated areas may have on the model and therefore the results. In terms of the size of the irrigation area covered by the study site it accounts for

[16] For justification see the methodology section.

approximately 91 percent of the licensed irrigation volume for NSW[17] and 70 percent of the total licensed volume. In the study site irrigated cotton accounts for approximately 96 percent of the total irrigated crop. The conclusion from these figures is that while the study site is not the whole catchment, it still accounts for a significant proportion of the licensed volume of irrigation water and irrigation area.

3.6. Conclusion

The problem being studied in this research is the way water has, and is being, managed and used is creating conflict between environmental and economic outcomes. It is proposed that seasonal climate forecasting information has the potential to forecast the variability of flows in coming periods and enhance the management of rivers to meet both consumptive and in-stream requirements.

In the BRC the agricultural sector is an important sector in the economy and as such represents a location where the conflict between consumptive and in-stream uses exists. While a considerable proportion of the value of agricultural production is reliant on irrigated crops the climatic conditions in the catchment indicate that without stored water for irrigation, the water deficit highlighted by Figure 3.3 would significantly impact on crop production. This underlies the worth of stored water for irrigation and has led to a series of public and private water storages being built and a set of management arrangements adopted to supply water for irrigation. These management arrangements strongly influence how water is used and also provide the opportunity for the potential of SCF information to be tested.

This conflict between uses and the ability to forecast the variability of flows in coming periods (as will be shown in chapter 4), combined with the existence of a hydrological model able to test water management scenarios over long time periods, makes the BRC a useful location to test the study hypothesis. While no two catchments in the Murray-Darling Basin and north of the BRC in the Fitzroy are the same, there tends to be broad similarities across a range of factors such as reliance on irrigated cropping, water scarcity, climate variability and to some extent storage and management arrangements. Therefore, lessons from this study in the BRC would be expected to be useful should consideration be given to using SCF information in these areas.

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[17] These statistics correspond to the irrigation and area parameters in the version of the BRC IQQM used in the study.

4. Methodology

4.1. Introduction

A key task in achieving the second study aim to evaluate the efficacy of seasonal climate forecasting on environmental flow management in the Border Rivers catchment is the development of an appropriate methodology. Chapters one and two began reviewing the rationale for, and goals of, undertaking water reform. Through this, one of the measures of success – managing the water resource to achieve the highest economic returns while maintaining ecological sustainability was identified.

It was determined that the overarching analytical framework for assessing the impact of seasonal climate forecasting on economic and environmental outcomes (in order to make inferences about the efficacy with which ARMCANZ goals) may be progressed was through a trade-off analysis using the production possibilities frontier (PPF) and the Pareto principle as shown in Figure 2.6. The selection of preferred options within this framework is by threshold value analysis (TVA).

In the context of Figure 2.6, assuming the initial policy outcome position is point C and a Pareto improvement is achieved, then a threshold decision needs to be made between preferences for outcomes x, E or y, where C is the status quo. If the decision is between an outcome within the frontier and one on the frontier, then the outcome on the frontier will be preferred. This is because, firstly, it is more efficient, and secondly because it is expected that this outcome will be on a higher indifference curve similar to point E on indifference curve μ_1 in Figure 2.11.

Given an assessment framework to determine whether the use of SCF information makes society "better off" the next task is to provide the inputs to this framework. That is:

- data on the consumptive and in-stream outcomes from a range of scenarios where water is increasingly allocated from producing one outcome to producing the other, and a range of mixed allocations in between to develop the basic PPF;

- the point on or within the PPF that represents the outcome from the current water management regime; and

- the outcomes from scenarios where the water management regime is altered based on SCF information.

This is achieved through the use of a modelling framework which interacts or links with the assessment framework as shown in Figure 4.1.

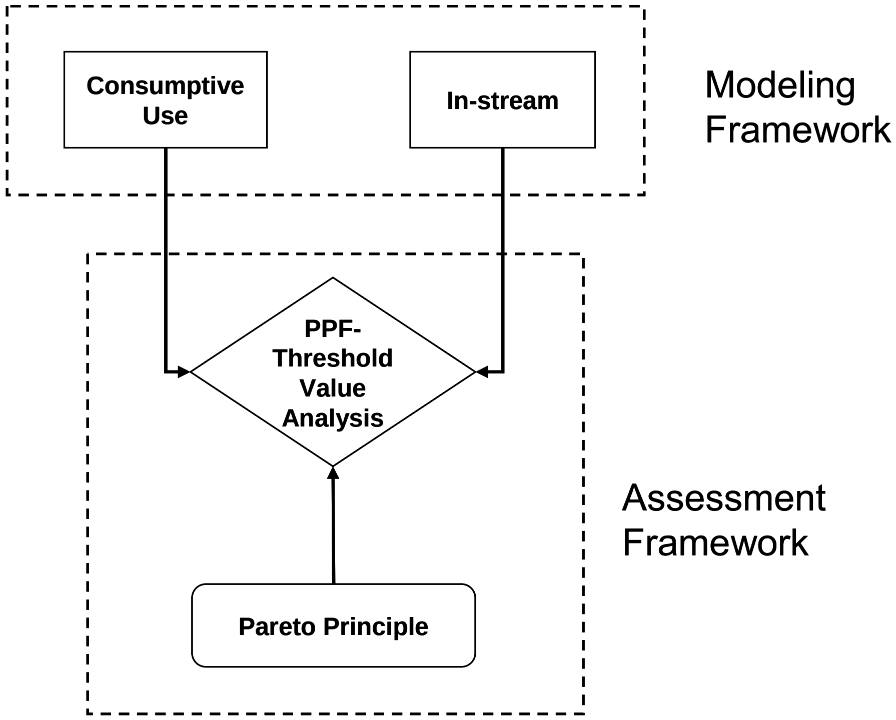

Figure 4.1: Mapping out the analytical frameworks used to test the study hypothesis.

The focus of the remainder of this chapter is on the methodology used in the modelling framework to produce the requisite data and how the assessment framework is implemented. The next section of this chapter specifies and describes how the individual model components link to provide the data used in the assessment framework. The sections following provide a more detailed explanation of each of the model components, their use in this research, technical specifications and operating requirements. The final sections of the chapter describe the implementation of the modelling framework and the scenarios examined in the analysis.

4.2. Model framework

The modelling framework adopts a systems approach, combining a number of different model components covering river flow management, economic and environmental analysis, and finally SCF information. The meta-model, henceforth referred to as the eco-environmental threshold meta-model (EETMM), is a set of sequential models with data from one model being used as input to the next. Figure 4.2 provides a simple schema of the main EETMM components required to estimate the economic and environmental outcomes from alternative water management scenarios for the PPF assessment framework.

These models were selected for a range of reasons. Firstly, these are being used in water policy- making. In the case of the hydrologic model IQQM, it currently underpins water planning decisions in much of Queensland and NSW. The environmental model is based on the framework that also underpins water reform and planning decisions in Queensland and New South Wales. Second, as will be shown, the economic analysis has been used in water planning and other resource decision-making within Queensland and NSW. Finally, the models are either already available as is the case with the IQQM and environmental model, or are able to be constructed as part of the research project within a reasonable timeframe.

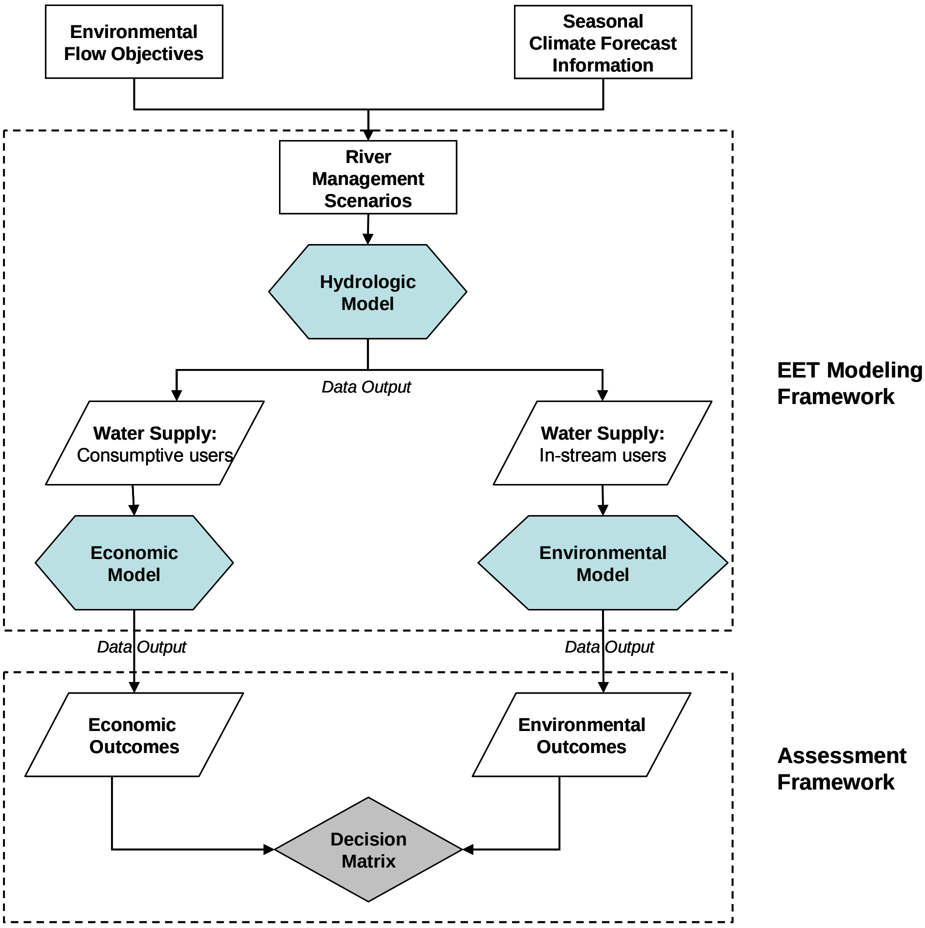

Figure 4.2: Major interdependent or linked modelling components of the EET meta-model.

4.3. EET meta-model components

The EETMM comprises five components. The three shaded components represent the models and include the hydrologic, economic and environmental impact models. The two remaining components include the scenarios tested in the research (prior to the hydrologic model) and the decision matrix (cross hatched). In the decision matrix environmental and economic outcomes are compared to assess the efficacy of the SCF information consistent with the trade-off analysis discussed in the preceding chapter (see Figure 2.6). This section, and that following, supply the detail on how the modelling framework provides this information. The final sections of the chapter present detail on the assessment framework and the river management scenarios tested.

4.3.1. Hydrologic modelling

IQQM

The Integrated Quantity Quality Model (IQQM) used as the hydrological model in this study was developed by the NSW Department of Land and Water Conservation and the Queensland Department of Natural Resources and Water (NSW DLWC 1998). A hydrologic model such as IQQM tries to simulate the operation of a river to examine the impacts of water resource management policies on the various users, both consumptive and in-stream. The IQQM is extensively used in the water reform and water planning processes in Queensland and NSW. In Queensland, the IQQM has been widely used, particularly in a number of key catchments including the Condamine Balonne, Burnett, Fitzroy and Border Rivers. Its use in NSW catchments has been broad and includes the Barwon/Darling, Namoi, Gwydir, Border Rivers, Clarence, Peel, Hunter, Murrumbidgee, Lachlan and Macquarie catchments.

The IQQM is a daily time-step model that can be applied to regulated and unregulated streams. It is designed to be capable of addressing issues relating to water quantity, quality, resource assessment and ordering, inter-catchment water transfer and environmental flows (Centre for Natural Resources, NSW Department of Land and Water Conservation 1999) although full capability in water quality modelling has not been achieved. The strength in the model comes from its ability to simulate the major river basin processes including:

- flow routing in rivers including branches, loops and tributaries,
- reservoir operation,
- irrigation,
- urban water supply and other consumptive uses,
- wetland and environmental flow requirements, and,
- water use accounting systems.

The IQQM is a generic hydrologic model and requires an input file to specify the physical, climatic and hydraulic data for the catchment being studied which is used in running simulations (Figure 4.3).

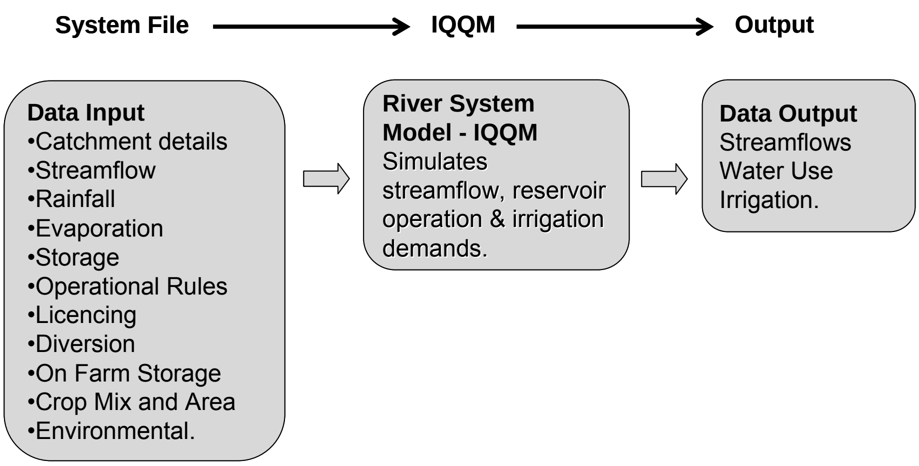

Figure 4.3: IQQM configuration.

In the model, river systems form a simplified representation of major river basin processes and demonstrate the physical structure of the catchment. Figure 4.4 illustrates the translation of a catchment with various processes on the left hand side: inflows; reservoir; town water demand; irrigation area; and, river gauging, into the schematic on the right hand side. The hydrologic model simulates river flows and operations using mathematical operations and equations in a water budgeting approach. The budgeting approach is implemented through accounting for flows moving downstream by tracking inflows from tributaries and outflows in the form of town and irrigation demand as well as natural losses such as seepage. Points within the river system where these inflows, outflows and other processes occur are referred to as nodes which are connected together by links.

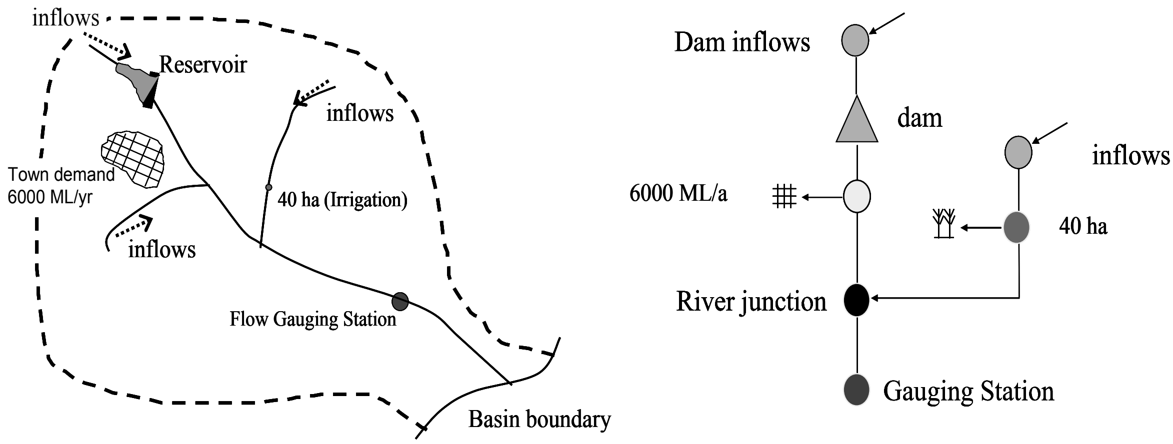

Figure 4.4: Transforming the physical landscape into an IQQM process diagram.

In the development of a catchment system file there are 13 major node types to represent, for example: various types of storages; tributary inflows; irrigation demand; off-allocation; town water supplies; and, wetlands. The location of a node such as the irrigation node in Figure 4.4 tells the IQQM that at that point there may be outflows from irrigation demand. So in any given year there is a crop type and area associated with that location and this drives a certain amount of demand and therefore outflow from the river. While the location of a node type tells the IQQM what processes occur at a point in the river, the input file (termed system file) identifies the parameter values for each node at each location in the system. An example of parameter values for an irrigation node (referred to herein as irrigation districts), are shown in Table 4.1. These parameter values are used by the model to calculate the volumes associated with processes that occur at each node type. For example, at an irrigation district assessment of available water resources, crop area decisions, water orders, crop water demand and operation of on-farm storages, are some of the processes modelled (NSW DLWC 1998).

The system file is then used by the IQQM during the simulation period to set the parameters governing crop planting and associated water use. The simulation is carried out over a period of years, in this study for the period 1894 to 1994. That is the simulation estimates the river flows, areas planted and water used as if the level of irrigation development identified in the system file existed in each of the years. In one sense this allows studies of how the system reacts both over time and under different rainfall levels. On the other hand as the system file is largely fixed for each simulation this approach also minimises the ability to have the levels of irrigation parameters change over time. This change would occur in a dynamic system where irrigators respond to trends in a range of factors including commodity prices by changing their setups including cropping mixes in the shorter term and over the longer term capital configuration.

Table 4.1: Example of parameter values for an irrigation district

Parameter	Value
On-farm storage volume	6,000 megalitres
Licensed volume	14,581 megalitres
Pump capacity	546 megalitres/day
Crop area	5,504 hectares
Crop types and proportions	
Cotton	4,623 hectares
Cereals	881 hectares

Of the 13 major node types in IQQM, three are critical to this study because they provide input data for the economic and environmental models. The first is an irrigation node or district. In addition to the information provided above it is an amalgamation of a number of adjoining farms representing areas that share similar characteristics such as crop mix. These may be from 295 hectares to 10,616 hectares (NSW DLWC 1998).

The second key node type is the off-allocation node used by the IQQM to determine daily access to off-allocation water for irrigation districts. The output from this node is a time series of volume of off-allocation water used by each irrigation district. The third key node type is a river gauge node used to calculate and output simulated flows at selected locations. The output from this node is a time series of flow volumes at a number of locations along the river system. This data is used in determining the impact of a flow regime on riverine environmental health.

<u>Selection of water supply parameter to study</u>

One of the key goals of this study is to evaluate the efficacy of seasonal climate forecasting on environmental flow management. It follows therefore that SCF information must be used in making water management decisions and these decisions need to be evaluated to ascertain if environmental flow outcomes are improved. In this study the method chosen to do this is to use SCF information to manage access to water for irrigation by attempting to match permitted access to supply. That is, the modelling allows access to more water for irrigation when river flows are high and less when river flows are low.

In developing options for how changes are made to water access it is useful to consider the components that make up the water supply facing an irrigator. The water supply components an irrigator considers in making irrigation and crop planted area decisions at the start of the cropping season are shown in Figure 4.5. Components linked via the full line are a function of previous years' rainfall, streamflow and water use, while those linked with a dashed line are generally determined by the upcoming season.

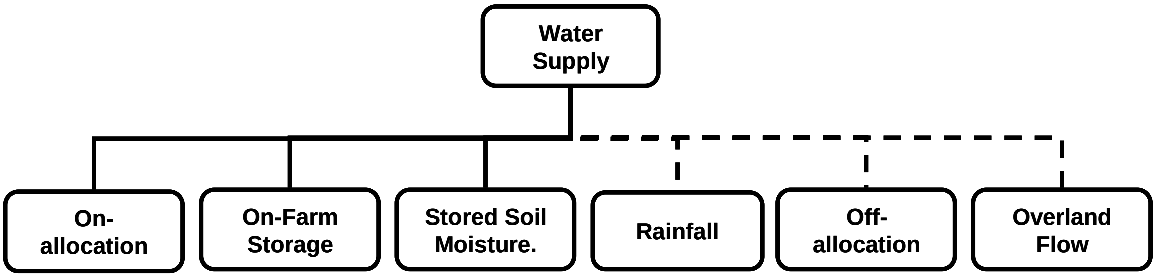

Figure 4.5: Water parameters facing irrigators when making planted area decisions.

Of these sources, some may be able to be managed based on SCF information whilst others cannot. Those that were determined not to be amenable to having access managed based on SCF information include: on-allocation; on-farm storage; stored soil moisture; and, rainfall. Overland flow has aspects that indicate it may be able to have access managed using SCF based rules, however there are a number of difficulties that prevent its use (see section 3.4.1). The final parameter, off-allocation water was identified as being able to be managed using SCF-based rules.

On-allocation water is a guaranteed amount known to the irrigator at the start of the irrigation season because the state water authority announces the percentage of nominal allocation that irrigators will have access to in that year. For example, an irrigator with a 1,000 ML licence may be advised that they can access 50 percent or 500ML of their nominal licence volume. The volume of on-allocation water available to a licence holder at the start of the irrigation year is calculated by dividing the total volume of water available by the total entitlement (i.e. sum of all licensed volumes) of the BRC and multiplied by 100. This in all but extreme drought years represents the minimum amount of water supply they can access during the water year and may be increased during the water year if there are sufficient inflows to dams. In determining the total volume of water available the key parameter is the volume of water held in the state-owned dams at a given point in time. This volume is a function of previous rainfall, streamflow and water usage. If irrigation plantings and water use in the previous year were high, then *ceteris paribus,* the volume of water in state owned dams at the start of the year would be expected to be low. Due to the lagged nature of this variable, it was not considered amenable to having its access managed based on SCF.

On-farm storage refers to water held by irrigators in their own dams built on their land. They are used to store water sourced from on-allocation, rainfall, off-allocation and overland flow. Similarly to on-allocation, if net water use in the previous year was high then, *ceteris paribus*, the volume of water in on-farm storage at the start of the year would be expected to be low. The combination of the lagged nature of supply into these dams, and the ownership status, rules out on-farm storage access management. Stored soil moisture is the volume of water held in storage within the soil. At the start of the irrigation season its level is based on previous rainfall and irrigation events, as well as water demand by previous crops.

Rainfall is used as supply in two primary ways. First, the volume of rain that falls directly on the crop area is used by the crop and stored in the soil. Second, rainfall may lead to runoff which can be captured before it flows into a river, creek or lake. This type of water supply is termed overland flow (DNRM 2002) and may be considered to be partly manageable as is evidenced by rules in some jurisdictions governing the proportion of rainfall able to be captured and used on farm. In NSW landholders are restricted to capturing and using ten percent of the average yearly rainfall runoff for their property. In Queensland the capture of rainfall is not specifically regulated, however water resource plans do specify rules for the management of overland flow, which is closely related to rainfall (DNRM 2003).

Once a regulatory framework for managing access to rainfall itself or via overland flow is in place, the ability to forecast rainfall (McBride et al. 1983), opens up the possibility of using forecast information to manage this access. This could be through the use of a policy where the proportion of rainfall able to be captured and used in any given year is dependent on the amount of rainfall expected as identified by a seasonal climate forecast. Despite this potential, in the first instance there is a paucity of reported statistics on the importance of the volume of overland flow actually captured and used in the BRC. In addition, the ability for IQQM to model overland flow to an acceptable level of accuracy is limited. Therefore there is limited ability to test this type of option at this stage.

The final component of water supply, off-allocation, may be granted to water-licence holders when dams spill or high flows enter the river system. In recent history the volume of off-allocation water used is a considerable percentage of total (Figure 3.5) and accounted for approximately 55 percent of water diverted in the NSW part of the catchment for the water years 1991 to 1999 (Abawi et al. 2001). While this water represents a significant proportion of the irrigation water supply, its usefulness suffers because it is water supply obtained within the irrigation season and as such the volume able to be taken in any given year is unknown at the start of the season when the summer irrigated cotton crop is planted. The inability to know the full water availability for the upcoming cropping year increases the uncertainty of what area to plant for irrigated crops. If the crop area is too small and a high off-allocation volume is received then a proportion of the off-allocation water available may not be able to be taken as the on-farm storage may be too full due to insufficient crop demand for irrigation water. This situation represents an opportunity forgone. Even if there is sufficient on-farm storage allowing additional off-allocation water to be held over until the next year losses to evaporation and seepage during the storage period will reduce the volume of water available for irrigated cropping the following cropping period. On the other hand planting too large an area in expectation of a high volume of off-allocation water availability introduces the risk of not being able to meet crop water demand due to insufficient water supplies. However, the correlation of streamflow to off-allocation water supply, combined with the ability to forecast streamflows, identify this variable as one that may be able to be managed based on seasonal climate forecasting.

The ability to forecast streamflow volumes in upcoming periods (see Abawi et al. (2001), Piechota et al. (1998), Chiew et al. (2000), Hamlet et al. (1999) and Simpson et al. (1993), as well as Figure 2.18), combined with the capacity to test options in line with the assessment framework, make the use of the off-allocation parameter the best variable for this study. Therefore, this study examines what impact changes to the rules governing access to off-allocation based on seasonal climate forecast information may have on both environmental and productive outcomes from water use.

IQQM modifications for this study

The use of off-allocation water as the variable to change is not straightforward because the IQQM used in the water reform planning processes is not configured to run scenarios where off-allocation access for irrigators is changed between years of a simulation. The version of IQQM used in the water planning process (hereafter referred to as the "IQQM planning model") treats potential off-allocation as set for each irrigation district for each simulation. The intention in this research to alter access to off-allocation (the independent variable) between water years based on seasonal climate forecasting necessitated changes to the manner in which the IQQM calculates and apportions off-allocation water to irrigators[18]. While this approach takes the study somewhat outside of the current water management system it is necessary because as noted by Hamlet et al. (2002) the rules for the current water management systems were developed at times when SCF information was not available hence no consideration was given to incorporating this information. Whether or not this study yields positive results the approach of operating outside of current management arrangements is likely to be an avenue for improving water management outcomes over time.

The primary amendment in the IQQM (hereafter referred to as the "IQQM research model") involved changes to the off-allocation water access decision rules in the Border Rivers IQQM system file to permit testing of the hypotheses. This entailed changing the rules surrounding the constant off-allocation cap to allow a cap that is variable for each year of the simulation. The volume of the cap is to be informed by SCF information. The decision process incorporated into the IQQM shown in Figure 4.6 is based on a decision tree approach (Hameed and O'Neill 2005). In contrast to the IQQM planning model, for each year of the simulation the research model selects an off-allocation limit for that year that is based on an input file identifying the year type. When a high flow event that triggers off-allocation occur, the amount of off-allocation water available to take is then limited by this limit or cap.

The year type input file in conjunction with the decision process shown in Figure 4.6 are the mechanisms for testing the hypothesis. The input file is a time series of year types based on a particular seasonal forecast or a fixed baseline for that year. There are three year types, corresponding to dry (1), medium (2) and wet years (3). Each of these year types has an off-allocation limit that the model uses to restrict the supply of off-allocation water to NSW irrigators for that particular water year. Further details on the implementation of the modifications to the IQQM are contained in a later section that outlines the range of scenarios tested.

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[18] The modifications to the IQQM system file for the BRC were made by the Surface Water Assessment section within the Department of Natural Resources and Water and the revised system file is referred to as the IQQM research model.

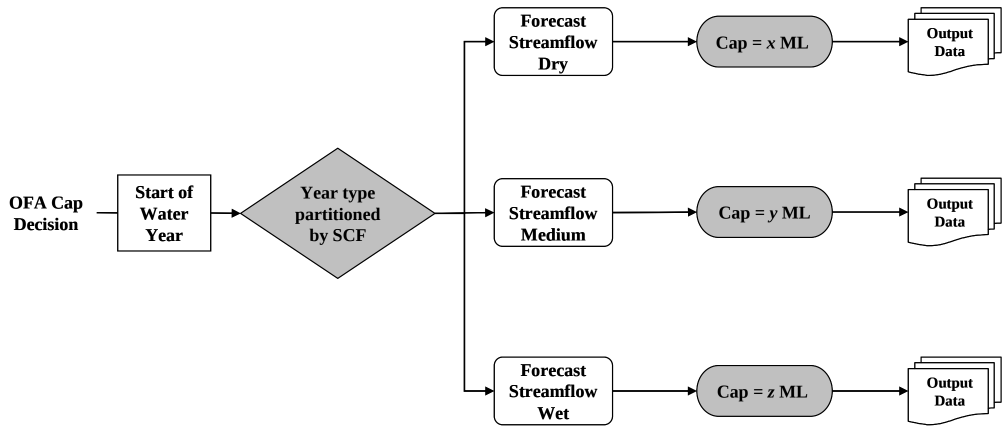


Figure 4.6: Implementing the decision rule for a variable off-allocation cap in IQQM.

The modification to off-allocation calculations require a number of changes to parameters within the IQQM that has necessitated a number of changes to the irrigator off-allocation cap settings for the BRC system file used in this study. The irrigation district parameters – licence, on-farm storage and plant area settings used in this study are listed in Table 4.2.

Table 4.2: Licensed volumes, on-farm storage volumes and crop parameters for the NSW irrigation districts in the IQQM research model used in this study

Irrigation district [19]	**Licence. vol**	**On-farm storage volume**	**Max irrigable area**	**Crop types**	
	(ML)	**(ML)**	**(ha)**	**Cotton**	**Cereal**
District 1	14,581	6,000	2,372	1,992	380
District 2	29,721	13,500	3,602	3,602	0
District 3	30,843	10,700	4,184	4,184	0
District 4	81,926	44,610	10,616	10,616	0
District 5	23,905	18,300	4,327	4,327	0
District 6	31,172	29,000	5,861	5,685	176
NSW total	212,148	122,110	30,962	30,406	556

Using IQQM output

Outputs from the IQQM are then inputs to the other model components – the economic and environmental models (Figure 4.2). There are four IQQM outputs used in these other model components: three in the financial and economic modeling, and one in the environmental modeling. The three output variables from the IQQM for the financial and economic modeling relate to the operation of irrigation enterprises - crop area, crop water delivery, and crop water depletion.

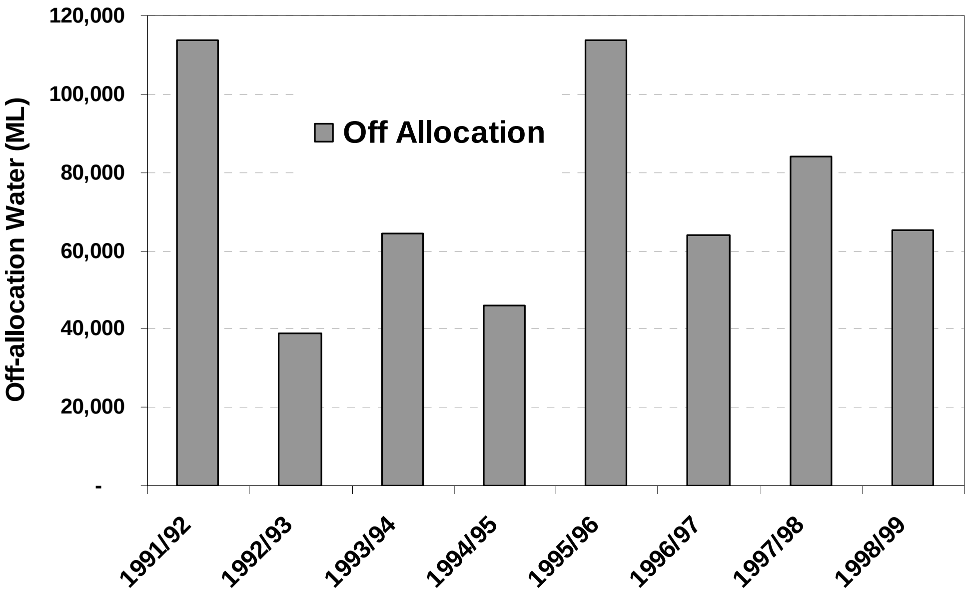
[19] Recall that an irrigation district is an amalgamation of a number of adjoining farms with similar characteristics. For the purposes of this study each irrigation node is effectively treated as a single farm.

Crop area decision-making process and IQQM

As it forms the foundation of the financial and economic modeling it is important to consider how irrigators make planting decisions and how these decisions are made within IQQM. In the BRC cotton is planted from mid-September (Abawi et al. 2001) at which point irrigators consider, amongst other things, commodity prices, cash flows, attitude to risk and the amount of water they expect to have to grow a crop. At planting time, of the six water sources open to an irrigator, in reality the volumes of water available from only three are known with certainty. These are represented by the cells linked with full lines to the water supply in Figure 4.5.

On the first of October each year the catchment dam manager notifies irrigators of their announced allocation[20] volume which is in addition to any water already held in the on-farm storage and stored within the soil profile. The remaining variables are not known with certainty at planting time as they rely on rainfall and runoff events during the growing season (broken line in Figure 4.5). However, in the case of off-allocation, the irrigator is aware of the maximum amount that may be available in the event of a wet season occurring. This maximum, known as the off-allocation cap, is set out in the management rules of the system. In making the area planting decision, irrigators factor in the possibility of intra-seasonal increases in announced allocations after dam inflows as well as potential rainfall, off-allocation and overland flow. The 'expectation' of rainfall, off-allocation and overland flow that may be received intra-seasonally is likely to be based on a number of factors including a reflection of the current season, past experience and potentially a seasonal rainfall forecast. Such decisions entail accepting a certain amount of risk, an example of which is indicated in Figure 4.7 by the variability of off-allocation volume used in the 1990s. This is only an indication in some cases that off-allocation water may have been available, but irrigators may not have been able to access it if the on-farm storages were full at the time. In the final analysis the assumption is that irrigators make plant area decisions based on the known water supply, an estimate of intra-seasonal water supply (which is likely to be influenced by their attitude to risk) and other factors such as commodity prices.

[20] In this case announced allocation includes carryover from the previous year, see section 3.4.2.



(Source: Abawi et al. 2001)

Figure 4.7: Historical off-allocation use in NSW section of BRC.

Plant area decisions in IQQM are based on certain aspects of the decision process outlined above. It is not possible for parameters such as commodity prices and financial status to be taken into account within the IQQM planting decision process and these are assumed to be held constant. Rather, this process is structured around an assessment of expected available resources or expected water supply, and a number of user defined parameters for each irrigation district:

- licensed volume (ML);
- maximum potential irrigable area;
- irrigation development factor;
- crop types and factors; and,
- pan evaporation and expected rainfall.

Calculation of the expected supply is based on a relationship between historical start, or announced, allocations and maximum final allocations for water years where the expected supply in the plant area calculation is the expected final allocation. A detailed explanation of this calculation and the user defined parameters can be found in NSW DLWC (1998). However it is worth noting that the decision is obviously based on what crop types are grown in the irrigation district in question (see Table 4.2). Expected rainfall is user defined and is estimated by examining the rainfall record, calculating a probability distribution from the record and selecting a level of probability of receiving rainfall. For instance, the amount chosen may represent the amount that is received during the cropping season in 75 percent of years. The final planting area decision based on this information in the IQQM planning model is calibrated against the historical plantings for the model calibration period. In seeking to match the model planting decisions to that of the calibration period small changes are made to the relationship between announced allocation and maximum final allocation (NSW DLWC 1998).

The implication from the method used to determine area of crop planted relates to the inability to take account of the non-water and crop related factors such as commodity prices which make the method used a simplification of the actual decision-making process. In actuality it is likely that commodity prices will have an impact on the level of risk that irrigators are likely to take regarding their expectations of intra-seasonal increases in allocation and off-allocation access. However, any shortcomings are overcome through the approach of comparing results of scenario analyses against each other holding all other parameters, including commodity prices, constant to allow comparison of like with like.

Crop water delivery

In calculating returns to irrigation one important variable is the amount of water applied to the crop. This is both a strong determinant of yield and a potentially substantial variable cost to account for. Crop water demand and water applied are calculated by the IQQM based on area, crop type, evaporation, crop watering efficiency and soil moisture accounting (DNR and DLWC 1999).

In developing the modeling components a complication arose due to the output of IQQM being an aggregate of the water applied to all irrigated crops being grown at any particular time. While the model calculates water requirements for each crop separately, the output is not separated by crop types. This means for example, that an output of the water applied in November may include water applied to a summer sorghum crop in addition to that of cotton. In order to calculate the economic outcomes of scenarios, results of each crop grown must be calculated which necessitates that both crop yields for each crop, and an estimate of water applied to each crop, must be made.

The means to overcome this problem is via the calculation of the crop water requirements made by IQQM which drives the amount of water supplied to the crop. Effectively a monthly water requirement is estimated for each crop, cotton, summer cereal and winter cereal, on a per hectare basis. The requirements are summed for each month and the proportion of each crop requirement is calculated. For example if the total crop requirement is 3 ML for December 1950 and the cotton crop requirement for that month is calculated to be 2 ML then the proportion water requirement for cotton is 67 percent. This proportion is then applied to the aggregate water applied December 1950 to calculate how much water was applied to the cotton crop in that month. In practice to address this problem the IQQM was run with each irrigation district growing a small amount of one crop only per season to calculate the monthly requirement for each crop throughout the water year. This resulted in the calculation of a share of the monthly water applied to each crop. The monthly water applied was subsequently apportioned to each crop according to the share of requirement. For example, if the requirement for 1 hectare of cotton in November in a particular year is 1.0 megalitre and for summer cereal 0.5 megalitres, then the resulting share of water applied is the ratio 0.67:0.33. This ratio implies that in periods of water shortage water will still be applied in this ratio ignoring the probability that in such cases the least profitable crop is unlikely to receive water in favor of the most profitable crop. In the final analysis, the calculation outlined has little bearing on the study undertaken because as shown in Table 4.2 cotton accounts for the majority of crop planted in the study area.

Crop water depletion

An important criteria for triggering irrigation, including in IQQM (NSW DLWC 1998), crop water depletion (or soil moisture depletion) can also be used as an indication of the health of the crop. Soil moisture depletion is part of the water balance process and in simple terms an irrigation event is triggered in IQQM when the soil moisture depletion reaches a certain level. However this variable can also be used in the economic model to partially mitigate an anomaly that arises when using IQQM to calculate the volume of water used by a crop.

IQQM is a hydrologic model and is not developed to model crop growth nor calculate the yield of a crop. Therefore, while the water balance process determines irrigation events this is important from a water modeling perspective only. In some years there is sufficient water to plant cotton but water for irrigation runs out part way through the season and subsequently after a streamflow event more water may become available for irrigation. In this case when crop water demand is unable to be met the crop is subject to water stress which can restrict crop growth and yield. Gibb et al (2009) show that photosynthesis in cotton rapidly decreases as available soil moisture content falls below 20 percent. Furthermore, the yield response to stress levels is dependent on the level of stress and the stage of crop growth. According to Gibb et (2009) moderate stress that could be caused by an increased irrigation deficit or severe stress potentially caused by providing less than three irrigations would reduce yield in the former and lead to a low yield in the latter.

In years where irrigation water runs out there is no difficulty and a yield for that particular year is calculated as explained in section 4.3.4 based on the volume of water applied to the crop. However, the IQQM can overestimate water applied when a streamflow occurs after a period where the crop water demand has not been able to be met and further water becomes available. This can lead to overestimating yield if the volume of water applied in these years is used to calculate yield without accounting for the yield loss due to crop stress.

In this study soil moisture depletion in IQQM has been tracked and in periods of water shortage where crop water demands are unable to be met it increases, (i.e. the soil moisture content falls to low levels), and crop water stress occurs. In years where irrigation water subsequently becomes available the potential to overestimate yield is addressed by nominating a level of soil moisture depletion where crop wilting point is reached and yield loss occurs. The yield reduction factors used in this study relate to the level of stress and the stage of crop growth and are identified in Table 4.3. In effect this means that if there is insufficient water available to meet crop water demands in December for example and the soil moisture level drops below the critical threshold (180 mm), the yield will be decreased by 100 percent for that year.

Table 4.3: Yield reduction factors due to crop death

Month	Yield reduction
October	0 percent
November	0 percent
December	100 percent
January	70 percent
February	40 percent
March	0 percent

4.3.2. Environmental model - riverine health indicators

The second major component of the EETMM identified in Figure 4.2 is the environmental model. This model is used to estimate the impacts of the various scenarios on riverine health and assess whether the environmental outcome from a scenario is a movement up or down the Y axis as one component of the trade-off analysis between consumptive and in-stream use benefits. Specifically, assuming a starting or baseline condition at point C in Figure 4.8 the role of the environmental model is to estimate the change in in-stream benefits, Δa, to provide one coordinate for determining the location of the scenario result E.

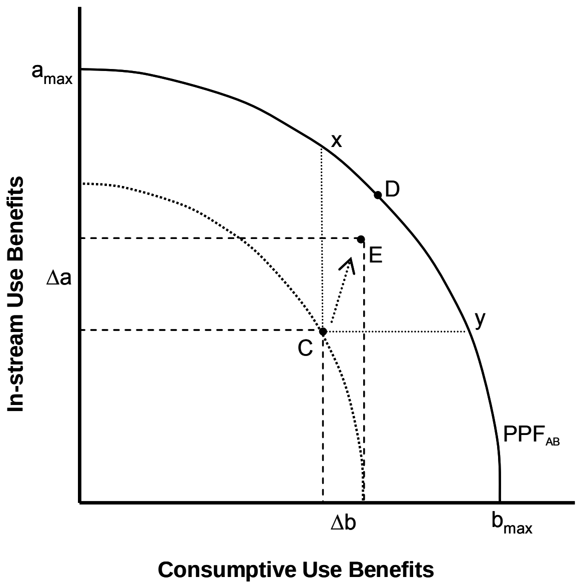


Figure 4.8: Assessing the impact of a change scenario on in-stream use benefits.

In the absence of robust and acceptable techniques (to decision-makers) for estimating monetised values for impacts of changes in water management regimes on in-stream benefits, the method selected is based on the physical relationship between water management changes and in-stream health. This approach fits with the scenario analysis approach adopted as part of this study that restricts the assessment of the in-stream impacts of different water management regimes to a modelling approach. The physical relationship for the environmental model is denoted by natural flows and the hydrologic indicator approach canvassed in the literature review. The relationship between hydrological processes and ecological factors is used to assess or make judgements on the health of the riverine environment (Bunn et al. 2002). This is implemented through the use of hydrologic indicators which are used to measure the characteristics of the modified flow regime to that occurring in the natural or pre-development state. To a large degree under the natural flows paradigm the closer the hydrologic indicators to the natural state, the healthier the river is assumed to be as there is less change to flow characteristics.

The approach used to calculate these indicators in both Queensland and NSW requires the hydrologic model IQQM to examine the variation between long-term flow statistics (performance indicators identified above) of consumptive scenarios with the natural flow regime. This comparison requires the development of an IQQM scenario for the catchment for the pre-consumptive use or 'natural' flow situation. In simulating natural flows irrigation use data, land use changes data, storage information, and flow control parameters are not included in the model (Abawi et al. 2001; Department of Natural Resources and Department of Land and Water Conservation 2000). This scenario is used to calculate these hydrological indicators that are compared with those from other production scenarios. The size of the variation between the scenario being considered and the natural scenario is used to assess whether the scenario will have a detrimental effect on riverine health.

The appropriateness of different hydrological indicators is an area of ongoing research and there is some variation in the indicators that are believed to be best used to assess impacts of changing flow regime on riverine environmental health. Whittington (2000) recommended that a comprehensive set of hydrological indicators be developed and those most ecologically relevant to each basin be chosen for assessment purposes. Appendix C Table C.2 provides a list of the key flow statistics chosen by Technical Advisory Panels[21] (TAPs) for the Condamine Balonne, Fitzroy, Logan and Barron WRPs. Perusal of this table highlights a level of similarity in the flow statistics appropriate across all of the WRPs (Whittington 2000). There is also a level of congruency with the hydrologic indicators catalogued in Jones et al. (2003) (more detail on the statistics can be found in (Whittington 2000; Brizga 2001; Department of Natural Resources 2000; Jones et al. 2003).

Hydrological indicators chosen for assessment purposes should be associated with the hydrological attributes which are related to the environmental outcomes being sought, i.e. different ecosystems may have different flow requirements, and that this may vary across the landscape (Whittington 2000; Brizga 2001; Jones et al. 2003). In broad terms, the outcomes being sought for riverine health in the BRC are identified in the respective water legislation of the Queensland and NSW governments. In Queensland, the *Water Act 2000* mandates the provision of sufficient water for the environment to ensure river health is maintained (Department of Natural Resources (Queensland) 2004), while in New South Wales, the *Water Management Act 2000* (New South Wales Government, 2002) identifies that health of the rivers has to be

[21] Technical Advisory Panels (TAP) were used to provide advice to the Queensland Department of Natural Resources and Water on the environmental requirements of aquatic and riparian habitats and species in that particular basin (Whittington 2000).

protected. More specifically, the Border Rivers Water Resource Plan in Queensland states that one of the objectives of the plan is to "to achieve ecological outcomes consistent with maintaining a healthy riverine environment, floodplains and wetlands" (DNRM 2003, p.6). The corresponding Water Sharing Plan in New South Wales has a similar objective to "implement environmental flow rules that protect, maintain and enhance the environmental, cultural and heritage values" (NSW Department of Water and Energy 2007b, p.3). Given these intended outcomes, both jurisdictions have also adopted methods based on the natural flows paradigm to assess whether these outcomes are being achieved by the planning arrangements being implemented.

Implementation of these objectives is assessed through the use of environmental objectives and associated performance indicators. Section 10 of the Queensland Water Resource (Border Rivers) Plan (WRP) (DNRM 2003) lists performance indicators and acceptable parameter values for the environmental flow objectives:

(a) end of system flow;
(b) low flow;
(c) summer flow;
(d) beneficial flooding flow; and,
(e) 1 in 2 year flood
(See Appendix C Table C.3)

These indicators, which according to the overview of the draft WRP for the Border Rivers (DNRM 2002), are related to the ecologically important characteristics of the flow regime and are assessed at a number of locations across the catchment. Objective (a) must be maintained to at least 61 percent of the pre-development (natural) pattern, while objectives (b) to (e) should be maintained in the range of 66 to 133 percent of the pre-development (natural) pattern. The study by Jones et al. (2003) introduced in chapter 2 of this thesis is the basis for the selection of the percentage range (DNRM 2002).

In NSW the draft Water Sharing Plan (WSP) for the Border Rivers Regulated River Source similarly lists a number of performance indicators to determine the performance against objectives (NSW Department of Water and Energy 2007c). The following are those relating to environmental flows:

(b) change in low flow regime,
  - number of days per water year where flow is below the natural 95^{th} and 80^{th} percentiles.
  - average and maximum number of days per water year of continuous periods of flow which is below the natural 95^{th} and 80^{th} percentiles.

(c) change in moderate to high flow regime,
  - number of days per water year where flow is above natural 30^{th}, 15^{th} and 5^{th} percentiles.
  - average and maximum number days per water year of continuous periods of flow which is above natural 30^{th}, 15^{th} and 5^{th} percentiles.

These indicators are both measured at end of system and other key sampling sites in the water source (NSW Department of Water and Energy 2007c).

In addition to these performance indicators NSW has an agreement with Queensland to maintain the end of system flow at a minimum of 61 percent of pre-development (natural) pattern (NSW Department of Water and Energy 2007b). The end of system flow is calculated as the mean annual flow at Mungindi at the terminal point of the catchment.

Even with these issues and differences, the selection of the flow statistics used in this study was constrained in a number of ways. First, in implementing this method for the study an issue arose regarding the version of the IQQM model and the associated software package "Post Processor" used to estimate the flow statistics (see below). That version of IQQM and Post Processor was not configured to calculate all of the indicators identified above.

Second, at the time of writing this thesis there are significant gaps in knowledge on the relationship between riverine health and streamflow and on the most appropriate indicators to use to measure health. These scientific knowledge gaps result in a level of uncertainty for the results of the scenario analysis. This uncertainty means that a change in hydrologic indicators which indicate an improvement in riverine health would be achieved is not guaranteed.

Finally, in addition to or because of, these knowledge gaps there was a lack of commonality regarding selection of indicators between the planning processes of the two states. These differences include disparity between the earlier studies, the draft, and the final plans made indicator selection difficult. At a broader level the hydrologic indicators chosen by each state have commonality in that they both target change in the low flow regime and aspects of the moderate to high flow regimes. The exception is the agreement on end of system flow.

In the assessment undertaken for the "Information Paper - Border Rivers Flow Management Planning, Stage 1, July 2000" (Department of Natural Resources and Department of Land and Water Conservation 2000) a number of the hydrologic indicators which are closely related to indicators in the WRP and draft WSP were used. In particular:

- median and mean annual flow; and,
- proportion of percentile flow duration: 2 percent, 10 percent, 50 percent and 80 percent.

These indicators are all ratios of modelled water flows and volumes under "developed" and "natural" conditions[22] and have been adopted in this study for use in assessing the environmental impacts of river management regimes and estimating Δa in Figure 4.8. The indicators are measured at the location of Irrigation District 6 which is the final decision point in the catchment for this study. This represents the end-of-study point analogous to the end-of-valley location which in the case of mean annual flow (mean AF) has a close relationship to the agreement by NSW and Queensland to maintain an end of system flow level.

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[22] Flows modelled for developed conditions include water extracted for irrigation, town water supply and other uses. Flows modelled for 'natural' conditions have infrastructure and water extraction data removed.

Mean annual flow ratio (mean AFR) and median annual flow ratio (MAFR) both provide an indication of the volume of water being diverted upstream of a particular point in the catchment, i.e. a measure of the impact of water resource development. Mean AF is calculated by summing the total volume for each year of a scenario and dividing by the number of years in the analysis represents the average volume of water in a year that flows past a point. It is reported as the ratio of the Mean AF of the consumptive scenario to the natural scenario.

MAFR is the median volume of water in a year that flows past a point. It is calculated by first ranking the annual flow volumes from lowest to highest and determining the middle or median value. It can be thought of as the annual flow volume that is equalled or exceeded in 50 percent of water years in the simulation period (Brizga 2001; Department of Natural Resources and Department of Land and Water Conservation 2000).

The proportion of the flow duration percentile (PFlowDur) is the ratio of the flow duration percentiles under "natural" and "developed" conditions and reflects the change from the natural case. Flow duration percentiles calculate the percentage of time that a particular streamflow is exceeded (see Figure 4.9). In this case the 80 percentile flow duration (80 PFlowDur) may be used to represent low flows, the 30 and 50 PFlowDur medium flows and the 10 PFlowDur represents high flows. In the analysis to be reported in the results section one scenario has daily flows ranging from 0 to 109 344 ML/day for the simulation period 1 January 1890 to 31 December 1997. The 80 percentile flow duration for this simulation is 88 ML/day while the equivalent for the natural flow is 95 ML/day. Therefore the 80 PFlowDur is the ratio of these or 88 percent. In interpreting PFlowDur a value close to one indicates that irrigation development has had little impact on the proportion of time a particular flow percentile is reached, however a value close to zero indicates a highly impacted situation. In this study the impact of a SCF flow scenario can be assessed by determining if the PFlowDur indicator for the scenario has increased or decreased relative to a baseline or without forecast scenario. If the PFlowDur indicator has increased closer to one the scenario may be assessed as improving the hydrologic indicator and therefore riverine environmental health. Alternatively if the PFlowDur indicator is decreased from the baseline other things being equal it is expected that riverine environmental health may decrease.

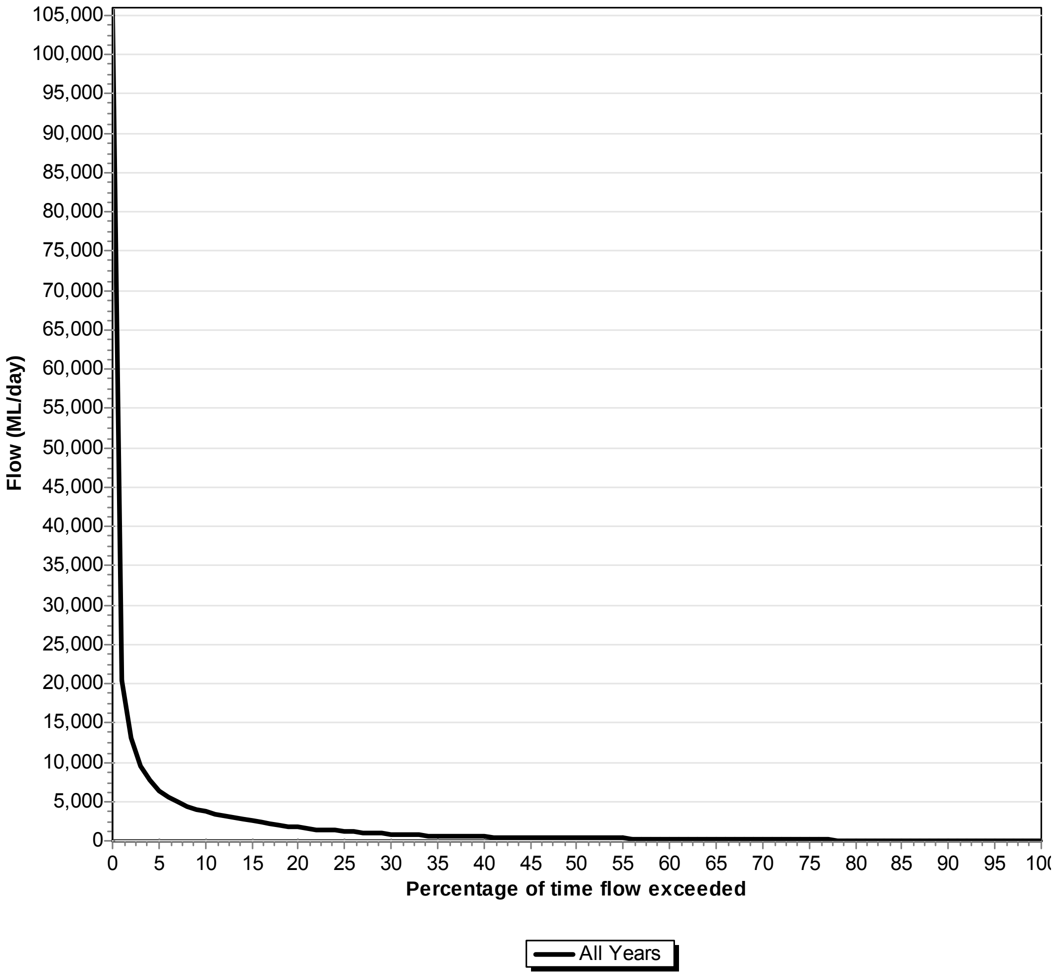


Figure 4.9 Graphical representation of the proportion of flow duration percentile

In the decision-making process utilised in this study two levels of interpretation are used to determine whether a particular scenario is acceptable. At the first level of analysis the process suggested by Jones et al. (2003) is adopted. That is if the key hydrological attributes are above two-thirds of natural then there is a high probability that the river is in a healthy working state. If the key hydrological attributes are above half of natural there is a moderate probability that the river is in a healthy working state. Where the key hydrological attributes are less than half of natural there is a low probability that the river is in a healthy working state (see Table 4.4).

Table 4.4: Decision rule used to assess the benefit of river management options

Key system level hydrological attributes (percent of natural)	**Probability of having a healthy working river**
≥ two thirds	High
≥ half	Moderate
< half	Low

(Source: Jones et al. 2003)

The second level of analysis assesses the change in the hydrological performance indicators from the baseline scenario. In absolute terms it might be argued that a decrease from the baseline values could indicate that the riverine health would be degraded if the particular flow management regime was adopted. Both of these analyses will allow a comparison to be made on the impact of a particular flow regime on the production possibility frontier. In the case of the decision rule by Jones et al. (2003) the flow statistics calculated will identify where the level of in-stream health is as identified in Figure 2.15. If the scenario outcome is a change in the hydrological indicator such that it drops below one of these thresholds $^1/_3$, ½, or $^2/_3$, (from the baseline level), then that scenario will not be judged as achieving a Pareto improvement. For example, assuming a starting point of C in Figure 2.15 if the hydrological indicator decreases but not below the ½ threshold then no change in riverine environmental health is assumed to have occurred. However if the hydrological indicator decreases below the ½ level then the scenario cannot be judged as having a Pareto improvement even if the consumptive use benefit change is positive. The second and more precise decision rule will rely on estimating Δa as shown in Figure 4.8 to ascertain the Pareto outcome.

Calculation of the performance indicators is undertaken using the "Post Processor" software program developed by the Queensland Department of Natural Resources and Water (Bennett et al. 2002). Figure 4.10 schematically shows the steps in assessing the environmental assessment of flow management scenarios.

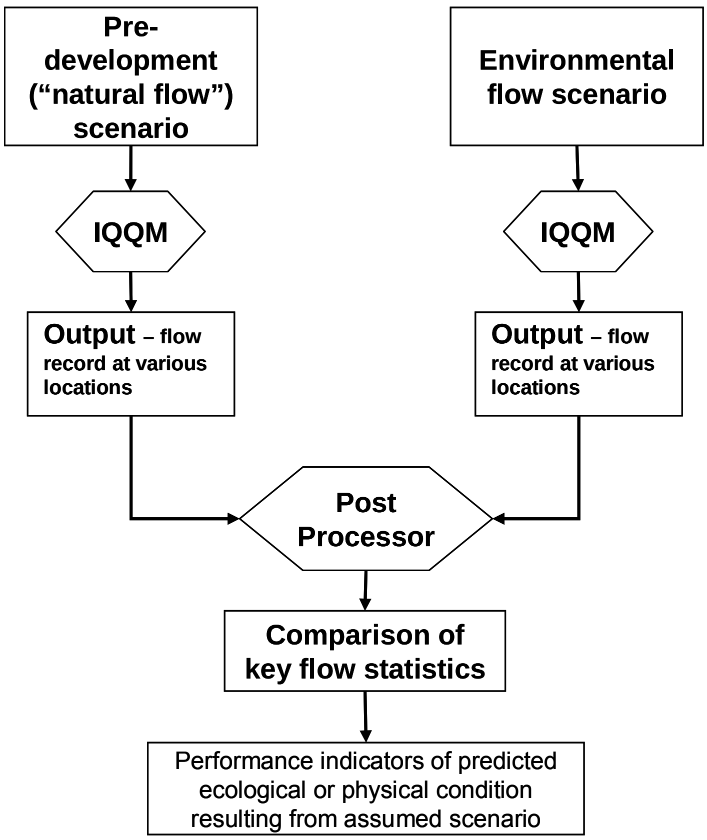


Figure 4.10: Use of Post Processor in calculating environmental performance indicators.

As identified above, output from the Post Processor program is used to judge whether a particular scenario represents an improvement or decline in riverine environmental health and subsequently the implications for the Pareto outcome.

4.3.3. Economic model

The analysis framework depicted in Figure 4.2 highlights economic analysis as one of the three model components of the EETMM to estimate economic or consumptive outcomes. This estimate represents the x axis coordinates of Figure 4.11 and is used in the threshold value analysis to assess the trade-off between economic and environmental benefits and costs. To explain the method used in this study for estimating economic outcomes the theoretical background and specific analysis details will first be introduced with an explanation of the operation of the economic model following.

Similarly to Jayasuriya, Crean and Jones (2001), DNR (2001), Eigenraam (2003), Webster (1998) and Murray-Darling Basin Commission (2004b), the analysis of the impacts on irrigated farming in the subcatchment is undertaken using a comparison of regional gross margin (RGM). RGM is used as proxy for the income derived from irrigation indicate changes in producer surplus as an estimate of the opportunity cost of introducing environmental flows. For example as discussed in chapter 2, as part of the Fitzroy Basin WRP the impact of new flow rules on irrigated agriculture were analysed by comparing aggregated gross margin across the subcatchments within the basin (DNR 2001). The estimates of RGM, a proxy for producer surplus allowing a comparison of producer surplus changes between scenarios, represent the opportunity costs of agricultural production which can be compared to the estimate of environmental change from each scenario. Therefore, in this study, the change in RGM between scenarios is used to represent the movement along the consumptive axis (Δb) of Figure 4.11 as the second component of the trade-off analysis between consumptive and in-stream use benefits.

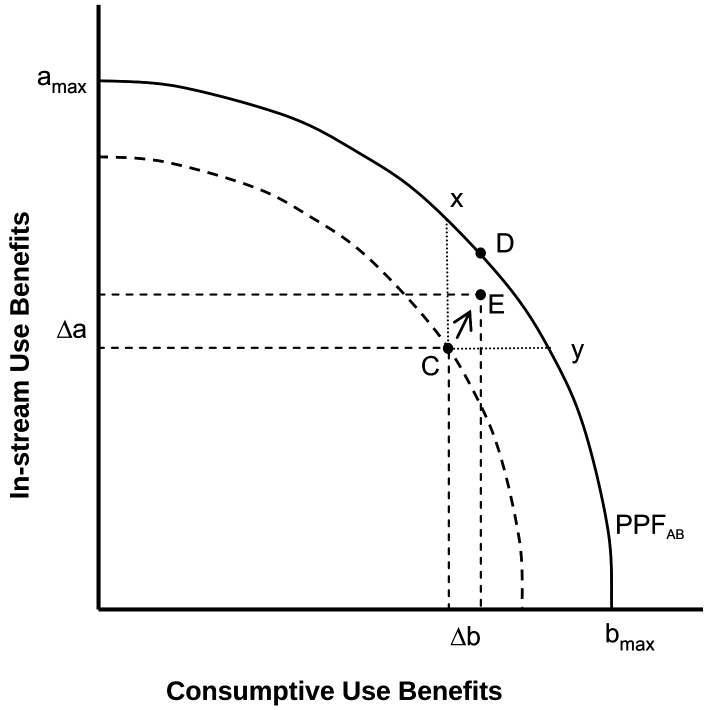


Figure 4.11: Assessing the impact of a change scenario on consumptive use benefits.

For a particular scenario, RGM is obtained by summing the total gross margin for the irrigation districts which can be described as:

$$RGM = \sum^{n=6} TGM \qquad \text{where:}$$

TGM = total gross margin for each district summed across n = 6 irrigation districts.

For each irrigation district:

$$TGM = \sum^{i=1 to 3} Ai.(Pi.Yi.) - (Ci) \ \text{where:}$$

Ai = area of each crop i in the district[23]

Pi = price of the crop i in the district

Yi = yield of each crop i in the district

Ci = variable costs for each crop i in the district.

This definition is consistent with that of Eigenraam et al. (2003) who described regional gross margin as:

> *"...gross agricultural income less the variable costs incurred in production aggregated across the relevant region."*

[23] There are three potential irrigated crop types in the model – cotton, summer cereal (sorghum) and winter cereal (wheat).

The six districts are described in Table 4.2 and covers the area shown in Figure 3.6 but is restricted to the NSW side of the border. Therefore regional gross margin in this study is defined as above and is restricted to this particular area.

This allows for comparison of the existing state of knowledge if forecasting was entirely accurate and forecast with current tools. Assessment of the matrix of RGM outcomes is carried out in a number of ways to build an understanding of how different flow scenarios impact on irrigators. The most straightforward is the simple mean of the 100-year simulation results of RGM. This is augmented by analysis of the variability of the annual results over the simulation period. Additional examination of the changes in the mean RGM for 20-year periods over the simulation period are used to highlight the response in a period more akin to the long-term planning horizons of farms.

Economic model output

The economic model output is used to assess whether irrigation water users are worse or better off as a proxy for identifying whether each scenario results in economic benefits or costs. In this case assuming scenario 1 is the baseline scenario and scenario 2 and 3 are different management options, then scenario 2 where mean regional gross margin is lower than the baseline scenario, cannot be a Pareto improvement because producers are made worse off. Scenario 3, however, with an increase in regional gross margin, is either an actual or potential Pareto improvement. In terms of the production possibility frontier, scenario 2 indicates a movement to the left of the line Cx in Figure 4.11, i.e. Δb is negative, and is not acceptable. In scenario 3, Δb is positive, indicating a movement to the right of Cx in Figure 4.11 and may be acceptable depending on the environmental impacts.

4.3.4. Economic model framework

The economic model, built in Microsoft Excel, has a number of primary input sources for water supply, crop yield and financial information as shown in Figure 4.12.

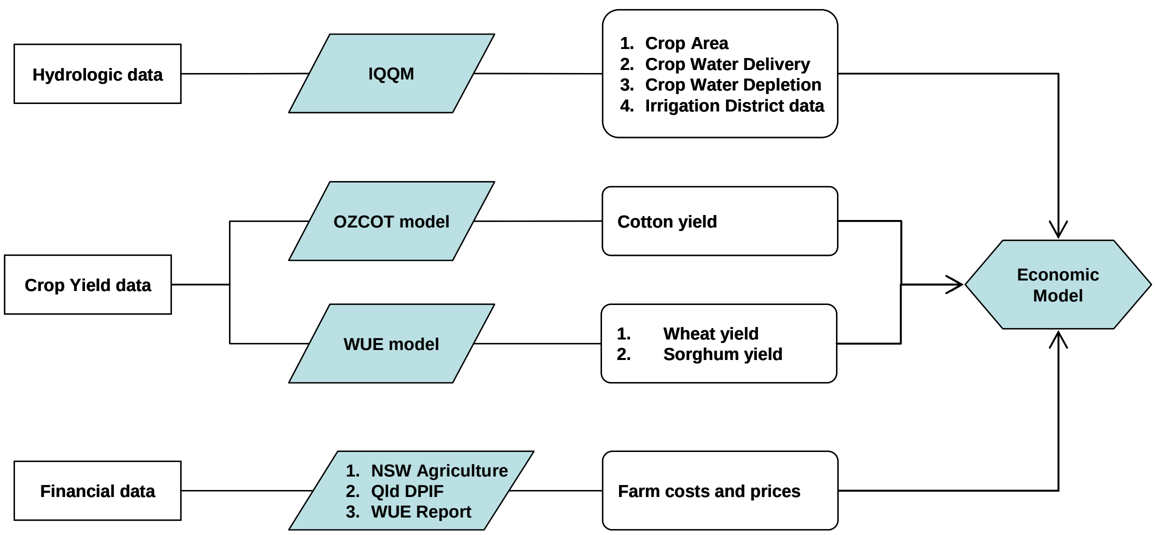


Figure 4.12: Input parameters and data sources for the economic modelling.

The three inputs - crop area, crop water delivery and crop water depletion - have previously been explained. However, calculating gross margins requires estimates to be made of yields for the crops grown. In the irrigation districts three irrigated crops are possible – cotton, sorghum and wheat. The following sections explain how yield estimates are made for these crops.

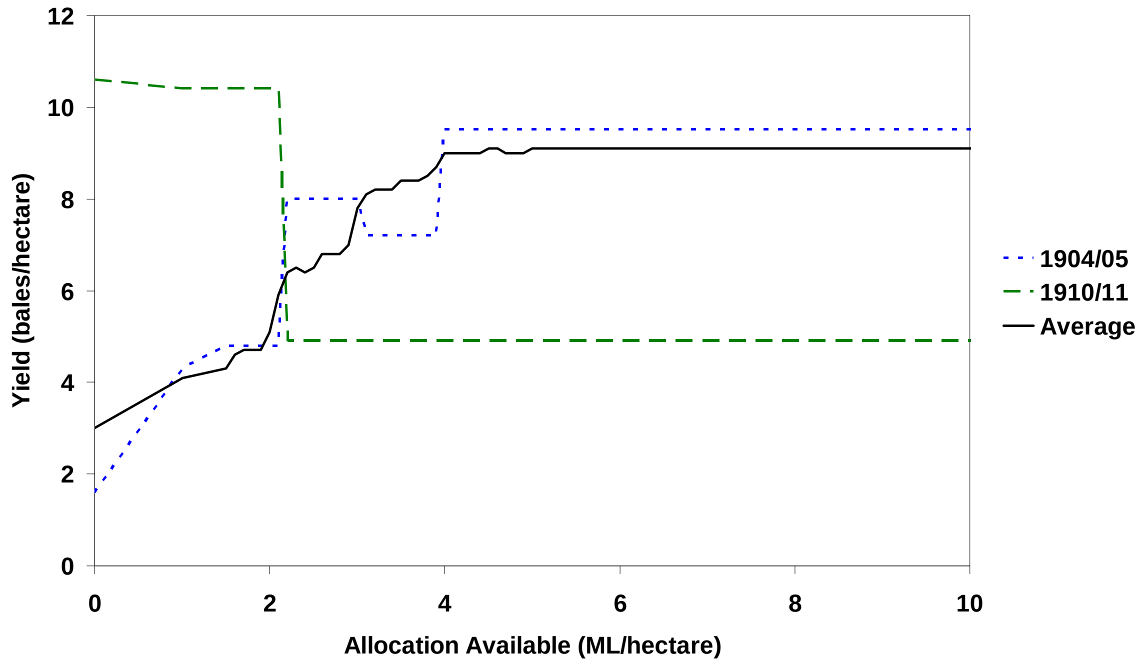
Cotton yield

Irrigated cotton yields for each year across a range of water supply levels were obtained from the agronomic model OZCOT developed by the Commonwealth Scientific and Industrial Research Organisation, Division of Plant Industry at Narrabri (Hearn 1994). For this research the OZCOT model was configured to Goondiwindi in the Border Rivers Catchment using Goondiwindi rainfall records for the period 1894 to 1994. Additional specifications used are reported in Appendix D The output of the OZCOT model gives a yield prediction in response to available water allocation from a zero (rain grown crop) to 12 ML per hectare.

In OZCOT the irrigation demand by a cotton crop in a region is driven by the daily plant available water content (PAWC) which triggers an irrigation when the PAWC drops to a predetermined level. The irrigation trigger is subject to a water supply constraint, and if insufficient water is available for an irrigation, the crop continues to grow under rain fed or dryland conditions. This procedure is explained in Ritchie et al. (2004).

The output of the OZCOT model can be displayed as a quasi-production function in Figure 4.13 or configured into a matrix of yields for a range of water allocation levels from 0 to 12 megalitres per hectare for each year of the simulation as shown in Table 4.5. In Figure 4.13 the average yield attained for each allocation level of the 100-year simulation from 1894 to 1994 assumes the shape of a classical production function showing an increasing level of output as the level of input increases up to a maximum point where diminishing marginal productivity occurs to such an extent that output decreases (Doll and Orazem 1984). This relationship would hold for irrigated cotton as the crop will respond to increases in water availability until a limit is reached when the yield will decrease as a result of water logging. This classical relationship is not exhibited for all of the OZCOT output as the model will not continue irrigation above plant requirements, therefore the yield output displayed in Figure 4.13 does not peak and then decline with increased water availability (Ritchie et al. 2004).

The implications from this are two fold. Firstly, there will not be an overestimation of yield based on water supply. Simple water budgeting methods where the yield is estimated based on how many bales of cotton are produced per megalitre of water applied can result in such an overestimation. On the other hand, there is also a risk of overestimating costs for irrigation where the IQQM model might calculate an application of 10 ML per hectare in a particular year but the OZCOT model calculates that only 8 ML per hectare need be applied to maximise yield. In this case the additional 2 ML per hectare may not have been applied by an irrigator in a real world situation. While the impact on yield is nil, there will be additional costs for applying the additional 2 ML per hectare.



(Source: Ritchie et al. 2004).

Figure 4.13: Quasi-production function, yield versus water allocation.

The second point is highlighted by two example irrigated cotton seasons: 1904/05 and 1910/11 chosen to highlight the yield implications of using a model such as OZCOT instead of an average yield over a simulation period or a regression of yield against key yield determining parameters such as water supply. In 1904/05 the yield increases with increasing available water allocation to a maximum of 8 bales per hectare at the allocation level of 3ML per hectare. After this point, the yield dips before increasing again to 9.5 bales per hectare. In the 1910/11 season after the 2.1 ML per hectare level of allocation is exceeded the yield drops dramatically from 10.4 to 4.9 bales per hectare as water applied increases. This response is explained by a water logging effect causing a yield decline where rainfall occurred shortly after irrigation. Small water logging events can have a temporary effect on yield when the crop yield potential at first declines after a water logging event but increases as the crop begins growing again after the soil dries out and further irrigation is applied. While this effect is somewhat unexpected in that a production function typically assumes a relationship where yield increases with increasing input levels up to a maximum after which over use of the input leads to yield declines it can be minimised by using an averaged quasi-production function calculated from the OZCOT output. However as noted by Ritchie et al. (2004) the use of individual year yields reflects the actual yield variability that occurs in cropping.

The yield matrix can be used in a number of ways in the economic model to determine yields. The first method involves using this matrix as a lookup table for determining cotton crop yield for each year of the economic and financial model based on the water supply per hectare calculated in the IQQM simulation. A second method that was examined was to calculate the mean yield per hectare over the simulation for each allocation level. This approach would smooth the yield curve to a shape closer to that of the classical production function. The differences in these curves can be seen in Figure 4.13 and the implications of these yield calculation methods for the economic model can be considerable. For example at the allocation level of six megalitres per hectare the difference between the yield simulated for 1910/11 at 4.9 bales per hectare and the mean at 9.1 bales per hectare is significant. At a price of $400 per bale[24] this equates to a change in revenue of $1,680 per hectare. In the economic model the complete matrix of results in the format of Table 4.5 is a yield lookup table to ensure that the interactions between irrigation, rainfall and yield are taken into account. The use of the complete matrix instead of a smooth yield curve based on average yields means that the yields carried through to the regional gross margin calculations reflect the variability inherent in the cropping environment. This is particularly important for cotton as it represents the dominant crop grown in the study area.

The additional specifications for the OZCOT model used in this research are reported in Appendix D .

Table 4.5: Cotton yield data (bales/hectare) output from the OZCOT model as water available for irrigating the cotton crop increases (also see Appendix D **)**

	Water available for irrigation (ML/ha)												
Year	**0**	**1**	**1.5**	**1.6**	**.**	**.**	**.**	**.**	**11.6**	**11.7**	**11.8**	**11.9**	**12.0**
1895	3.4	3.5	3.4	3.4	.	.	.	.	5.2	5.2	5.2	5.2	5.2
1896	3.8	3.9	3.6	3.6	.	.	.	.	9.8	9.8	9.8	9.8	9.8
.	.	.	.	.	.	.	.	.	.	.	.	.	.
.	.	.	.	.	.	.	.	.	.	.	.	.	.
.	.	.	.	.	.	.	.	.	.	.	.	.	.
1993	3.9	3.8	3.6	3.6	.	.	.	.	10.6	10.6	10.6	10.6	10.6
1994	0.1	0.1	0.1	0.1	.	.	.	.	9.3	9.3	9.3	9.3	9.3
Mean	3	4.1	4.3	4.6					9.1	9.1	9.1	9.1	9.1

Wheat and sorghum yield (WUE model)

Apart from irrigated cotton, the IQQM crop model "plants" either a winter or summer cereal crop in the area being studied. In this study these crops are assumed to be irrigated wheat and sorghum for winter and summer crops respectively. The yield for these crops are calculated using a water use efficiency (WUE) concept (GRDC 1998) where yield is a function of the total water supply:

> Water supply (mm) = Stored water (mm) + In crop rainfall (mm) – Water loss factor (mm).

Where the water supply is known, the yield is calculated as:

> Yield Potential (kg/ha) = Water supply (mm) x WUE (kg/ha/mm)

where WUE is determined by the ratio of historical crop yields (kg/ha) and water supply (mm).

[24] The price per bale of cotton used in the economic model was $400 per bale. This was based on the average of the 2005 and 2006 prices reported in the Australian Cotton Comparative Analysis 2006 (Cotton Catchment Communities CRC & Cotton Research and Development Corporation 2007)

The WUE calculation was devised to assist farmers and cropping advisors in benchmarking and comparing management options. While being comparatively simplistic, it was chosen because it is able to be calculated quickly while remaining accurate when compared to other methods (Freebairn et al. 1997). This simple calculation is justified by the relatively small ratio of irrigated cotton to cereals (approximately 3 percent). A full explanation of the model is located in Appendix E .

Financial data

The final data required for the economic model are the costs and prices for the regional gross margin calculations. Details of the cost and price data are listed in Appendix F . The sources of this data were NSW Agriculture (2003b), NSW Agriculture (2003c) and NSW Agriculture (2003a).

Running the production model

The economic model is built and run in a step wise fashion with eleven steps as shown in Figure 4.14 which in essence calculate the irrigation district gross margins for each of the six districts and sum them to give regional gross margin for each scenario. Within each scenario, the key steps once a scenario is setup are:

Step 1: Select the scenario to run.

Step 2: The first irrigation district is selected.

Steps 3 and 4: The IQQM output files are opened and the area planted for each year of the simulation period (1894 to 1994) is read into the model for each of the three crop types: irrigated cotton; irrigated sorghum; and, irrigated wheat. In addition, the volume of water applied to each of the crops for each cropping year of the simulation is calculated using the Crop Water Delivery IQQM output.

Step 5: The yields for each of the irrigated crops is calculated. For cotton this is achieved using a lookup table as referred to in Table 4.5. The irrigated sorghum and wheat yields are calculated using the WUE method.

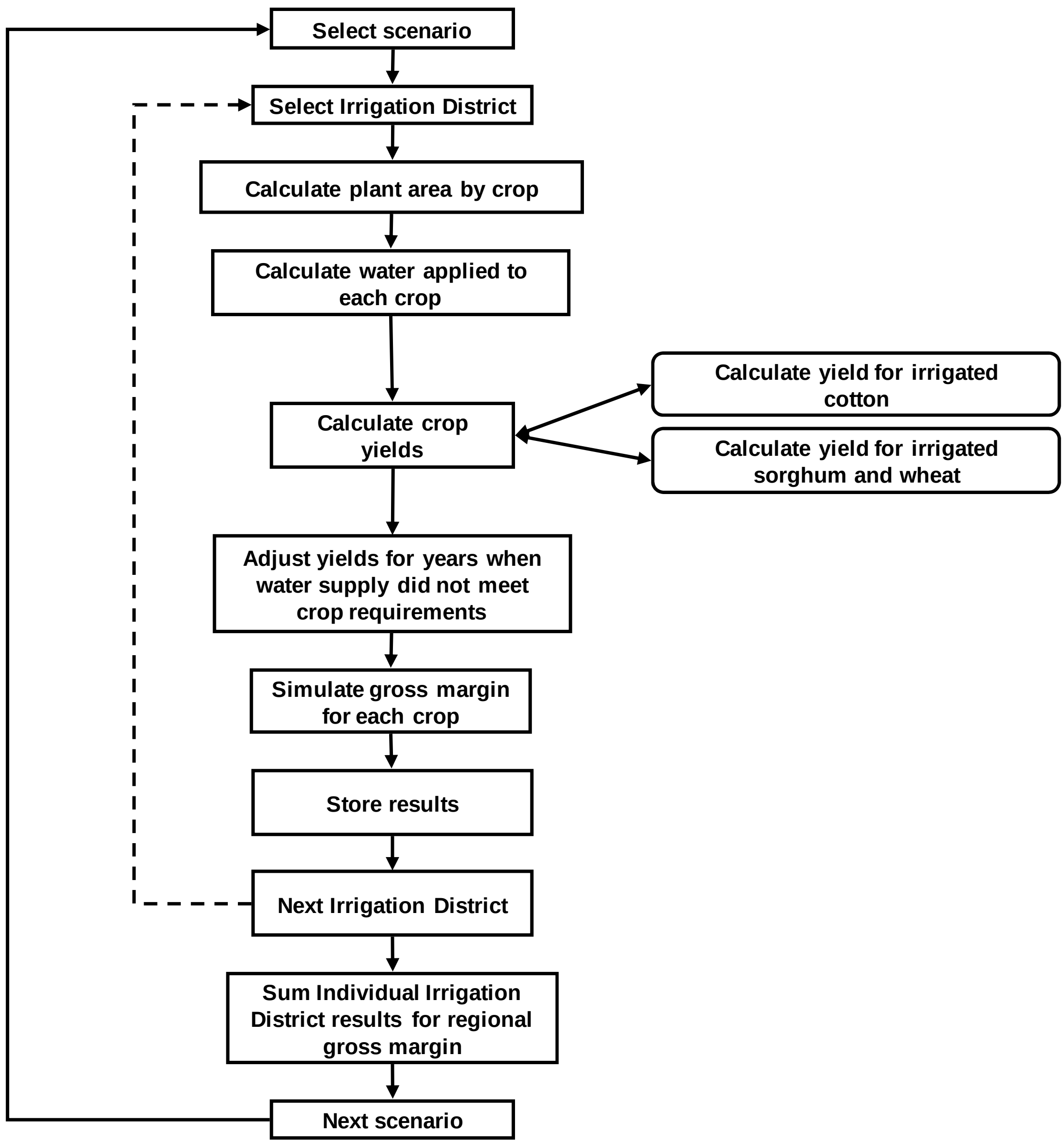


Figure 4.14: Structure of the production model.

Step 6: In this step the cotton yields are adjusted for those years when water supply was insufficient to meet crop requirements during the season. This is achieved by examining the IQQM Crop Water Depletion data output to identify the maximum level of depletion in each month of the cropping season. Where the soil moisture level fell below a threshold level of 180mm in a particular month, the yield for that year was subsequently reduced by the percentage identified in Table 4.3. The percentage reduction used to determine final yield for the cropping season is the higher reduction factor of the months December, January or February. For example, if the January and February yield reduction factors were triggered, the yield for the season would be reduced by 70 percent of the potential yield if water was not a limiting factor.

Step 7: The yield and water delivery data from steps 3 and 4 are then used to calculate the gross margins per hectare for each crop for every year of the simulation.

The total gross margin for each year is then calculated by multiplying the gross margin per hectare by the area planted.

An important assumption in calculating the total gross margin is that no dryland crop is assumed to be planted in the irrigated cropping area when the area of irrigated crop planted in a year is below the maximum area available. This is based on Ritchie (2004) where anecdotal discussions with a number of irrigators in the Border Rivers Catchment highlighted that a range of practices in relation to planting dryland crops in irrigated areas takes place ranging from planting dryland crops to not planting dryland crops in the irrigated area.

Steps 8 and 9: The time series (1894–1994) of total gross margin for the irrigation district is then stored and the model returns to step 2, and the next irrigation district is selected.

Step 10: The regional gross margin for each year of the simulation is calculated by summing the total gross margin for each irrigation district. This provides a time series of RGM for further analysis as reported in the results section.

Step 11: The model is configured to permit a number of scenarios to be run consecutively and then summary results from each can be compared. This comparison forms the basis of the estimate of Δb in the trade-off analysis in Figure 4.11.

Completion of this part of the EETMM means that for each scenario there is now two pieces of data: data from the environmental analysis; and, data from the economic analysis. Judgment of the Pareto outcome from each scenario requires that these two pieces of data are integrated into a decision-making framework.

4.4. Integration of model output - testing the hypothesis

In this research a case study approach is applied to test the hypothesis and examine the potential use of SCF information in environmental flow management. This is achieved using two related assessments. In the situation of dual economic and environmental goals, a trade-off analysis incorporating the Pareto principle is used to assess outcomes in relation to these goals in one analysis. This trade-off analysis is carried out using the approach of a Production Possibility Frontier however it is extended such that instead of analysing trade-offs in production terms the trade-off is assessed between economic and environmental outcomes. The typical PPF for assessing trade-offs between environmental and in-stream benefits of water use identified in Figure 2.6 is amended to that shown in Figure 4.15. The variables used for assessing the trade-offs become hydrological indicators as the measure of in-stream use benefits and regional gross margin as the measure for consumptive use benefits. While this approach is outside of the strict PPF analysis it is important to note that changes to the RGM variables of measure remain driven by the physical relationship to changes in water supply as the prices used to convert to dollar terms are kept constant. Acknowledging that this analysis in strictly not a PPF analysis from here it will be referred to as the trade-off analysis while bearing in mind the linkages to the PPF approach and the Pareto principle.

A second analysis is undertaken by calculating the value of the forecast information in either improving the regional gross margin of irrigated agriculture or producing conditions that will lead to improved riverine environmental health. This analysis focuses less on the trade-off between the two outcomes and more on a one dimensional analysis targeted at each outcome individually. Understanding the value of information in achieving either economic or environmental outcomes can aid in understanding the dual goal assessment.

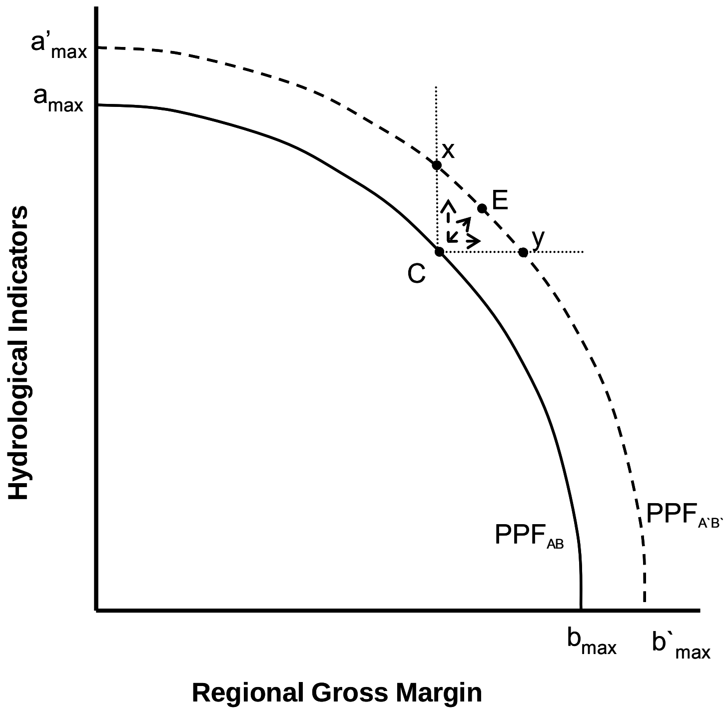


Figure 4.15 The trade-off analysis adopts units of measure that are extended from productive units.

4.4.1. Valuing the forecast information

The value of a forecast can be expressed in a number of ways. In simple terms it can be expressed as the increase (or decrease) in profit that can be attained through the use of the forecast information (Mjelde 2002; Letson et al. 2009). Mjelde and Cochrane (1988) equate the value of information to a premium (p) a decision-maker would be willing to pay to have access to the information. In this study the value is not easily defined because there is no metric for measuring all values e.g. values for environmental benefits. For irrigation use, dollar values such as regional gross margin can be used but there are no dollar values placed on environmental benefits which would allow a composite value to be determined. Therefore, 'value' merely represents a difference between various outcomes, however these outcomes may be defined.

The case study approach permits consideration of the implications of forecast accuracy on the water management process by assessing the value of forecast information through comparing outcomes for scenarios where a perfect forecast is available with the situation where the current standard of forecast is used. Where the value of forecast information is small this points to inaccuracy and visa versa.

Implementing the method for calculating the value of the forecast information involves comparisons amongst three scenarios which form part of the trade-off analysis. First, outcomes from a "perfect" forecast, ∂. Second, outcomes for the "baseline" or without forecast scenario (Ω). Finally, outcomes from scenarios developed using SCF information represented by χ. In this method it is expected that outcomes or value follow the pattern:

$$\chi \text{ has a positive value if } \Omega < \chi < \partial \qquad (1)$$

In the case where:

$$\Omega > \partial \qquad (2) \quad \text{or}$$

$$\Omega > \chi \qquad (3)$$

the forecast system has no value as better outcomes are achieved using the baseline scenario rules.

Where (1) represents the outcome hierarchy, the value of the forecast (VoF) can be represented by:

$$\mathrm{VoF} = \chi - \Omega \qquad (4)$$

In addition, the potential forecast value (PFV) remaining is represented by the difference between the perfect forecast scenario and forecast scenario:

$$\mathrm{PFV} = \partial - \chi \qquad (5)$$

Production of a perfect forecast is complicated by the issue of what variable should be forecast. One option given the variable off-allocation cap decision rule, explained using Figure 4.6, is for the perfect forecast to pertain to dry, medium or wet years. As the partition for these year types in this study are streamflow terciles for gauged (modelled) flow at Goondiwindi, then a perfect forecast might relate to knowing in advance which tercile the gauged flow in each year will be. The actual volume of off-allocation water able to be accessed by irrigators is strongly related to the volume of streamflow in a particular water year. However, there are a number of other factors that affect off-allocation availability, such as current on-farm storage levels and environmental needs, therefore the correlation between off-allocation availability and streamflow is expected to be less than 100 percent. A perfect forecast for that reason could be said to be one that predicted the amount of off-allocation water available to an irrigator in each year. Therefore, a scenario was developed where no restrictions were put on water use to represent the perfect forecast scenario.

This equates to a forecast made for consumptive uses as opposed to one that might be made to assist in achieving an environmental outcome. In this situation, where the proxy perfect forecast scenario outcome is represented by $\partial`$, equation (1) becomes:

$$\chi \text{ has a positive value if } \Omega < \chi < \partial` \qquad (6).$$

The value of the forecast remains equal to (4), but the potential forecast value is represented by:

$$PFV = \partial` - \Omega \qquad (7).$$

Incorporation of the value of forecast information into the trade-off analysis is explained using Figure 4.16 where it is assumed that the baseline or without forecast scenario (Ω) is within the frontier and the outcome from the perfect forecast scenario ($\partial`$) lies on the frontier. Furthermore a Pareto improvement is attained by shifting the outcome from point $\Omega$ into the quadrant x$\Omega$y. For (1) to be valid the outcome of the forecast scenario ($\chi$) must lie somewhere between $\Omega$ and the curve x$\partial`$y.

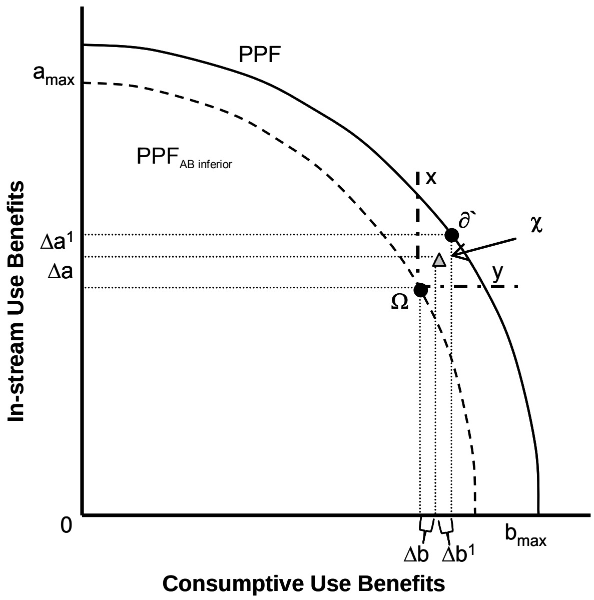


Figure 4.16: Incorporating the forecast value into the trade-off analysis.

The value of the forecast is then measured by the differences between these outcomes using the respective parameter of interest. From Figure 4.16, the value of the forecast is:

$$PFV = \partial` - \Omega \qquad \text{(7) is equivalent to}$$

$$PFV = (\Delta a + \Delta a^1) + (\Delta b + \Delta b^1)$$

However, focusing on the one dimensional analysis because of the lack monetised values for in-stream benefits, PFV for consumptive uses becomes:

$$PFV = (\Delta b + \Delta b^1)$$

Subsequently

$$VoF = \chi - \Omega \qquad \text{(4) is equivalent to}$$

$$VoF = \Delta b.$$

Should monetised values for in-steam benefits be available then:

$$\text{PFV} = \partial` - \Omega \qquad \text{(7) is equivalent to}$$

$$\text{PFV} = \Delta a + \Delta a^1.$$

Subsequently

$$\text{VoF} = \chi - \Omega \qquad \text{(4) is equivalent to}$$

$$\text{VoF} = \Delta a.$$

However as values for in-stream benefits are not used in this study the VoF or Δa is not able to be estimated. The environmental outcomes from $\partial`$ will however be examined.

4.4.2. Using the trade-off analysis to determine Pareto outcomes

The dual economic and environmental trade-off analysis highlights the combinations of economic and environmental outcomes possible from changing the usage pattern of water and provides insights into the relationships between the two outcomes (Doll and Orazem 1984; Wiggering et al. 2006; Scott et al. 1998). Combining the Pareto principle and TVA facilitates an assessment of the outcomes from scenarios using seasonal climate forecast (SCF) information in water management decisions against a baseline "without forecast" scenario.

To implement this assessment, data output from the economic and environmental models is used in the trade-off analysis to identify those scenarios which are in an efficient set (Pareto acceptable) and those not (Pareto unacceptable). Graphically this is demonstrated in Figure 4.16 where a Pareto improvement is attained by shifting the outcome from point C into the quadrant $x\Omega y$.

While the Pareto principle relates to the summation of individuals' or households' welfare, the justification for using it in this instance (across economic and environmental outcomes) comes from a basic implied premise of the water reform process. Judgements on whether a potential Pareto improvement is attained are based on whether a change to a policy results (at a minimum) in making someone better off without making anyone worse off (Johansson 1991). That is, any change in management or policy that leads to increased environmental degradation is unacceptable as someone who values the environment is made worse off. Put another way, society has placed a threshold value on the environment such that further negative changes in environmental health are implicitly valued so highly that no positive changes in outcomes from consumptive use will be permitted if this level is exceeded.

Assuming that a Pareto improvement from the baseline point Ω in Figure 4.16, the outcomes can be a point within the quadrant bound by $x\Omega y$ such as χ, or any point on the frontier xy. In the absence of data for indifference curves it is difficult to choose between points along the trade-off curve. It is however possible to apply TVA in conjunction with the principle of Pareto improvement to show that χ or any point inside the quadrant is superior to point Ω. In other words a gain in at least one outcome without loss to the other is an improvement. The rationale for this approach instead of using extended cost-benefits analysis were explained in chapter 2 and relate to issues surrounding placing monetary values on the environment (McMahon et al. 2000), and criticism on the use of surveys to elicit values and problems of bias (Lockwood and De Lacy 1992; Young 1996). With the TVA + Pareto approach the analysis only needs to show physical changes reasonably associated with environmental gains in order to make a judgement on the efficacy of water management changes.

The TVA + Pareto framework was therefore chosen due to an appropriate level of theoretical underpinning whilst maintaining an acceptable level of robustness. An additional benefit of this approach is that decision-makers will gain a stronger understanding of the trade-offs inherent in different levels of use of the water resource. Through the use of the trade-off curve, the implications of decisions are able to be understood more fully. Therefore the method is suited to informing decision-making processes as opposed to providing an answer indicating one option is better than another.

To illustrate this trade-off analysis in another way the data sourced from the economic and environmental models can be viewed as a decision matrix (Table 4.6) as referred to in the flow chart of the EETMM model (Figure 4.2). The matrix lists the possible outcomes from the comparison between the "baseline" or current management scenario and other scenarios developed using SCF information. It identifies whether the outcomes from a particular scenario represent a potential Pareto improvement or fail to meet the Pareto criterion.

Outcomes 1, 2 and 3 are cases where there is an increase in both economic and environmental health indicators or an increase in one without diminution of the other and consequently represent a movement from point Ω into the quadrant bound by $x\Omega y$ (Figure 4.16). By definition these outcomes are the efficient set as they result in potential Pareto improvements. On the other hand, outcomes 7, 8 and 9 represent cases where the movement from point Ω is to a point outside of the quadrant bound by $x\Omega y$. These outcomes result in a decrease in both economic and environmental health indicators or decrease in one without diminution of the other. Therefore they are judged to be non-Pareto improvement outcomes and subsequently sub-optimal. Outcomes 4 and 5, with an increase in one indicator and a decrease in the other, correspond to possible Kaldor-Hicks outcomes (Johansson 1991). The threshold value technique can be used by decision-makers in situations where this type of outcome occurs to assist in judgement of the usefulness of SCF information. Finally, outcome 6 depicts the situation where the policy change induces no change in either economic or environmental outcomes.

Table 4.6: Matrix of outcomes identifying those scenario outcomes that result in a potential Pareto improvement

			Environmental outcome		
			Proportion of flow duration percentile		
			Increase	No change	Decrease
Economic outcome	***Change in regional gross margin***	Increase	Outcome1*	Outcome2*	Outcome4**
		No change	Outcome3*	Outcome6	Outcome7
		Decrease	Outcome5**	Outcome8	Outcome9

* Outcomes for which there is a potential Pareto improvement (move towards trade-off curve on or within a north-east quadrant).

** Possible Kaldor-Hicks outcomes.

Graphically an understanding of the general direction of change from a starting point C for each of these outcomes is displayed in Figure 4.17. The lines aa and bb equate to the threshold values mentioned above and previously explained in Figure 2.7.

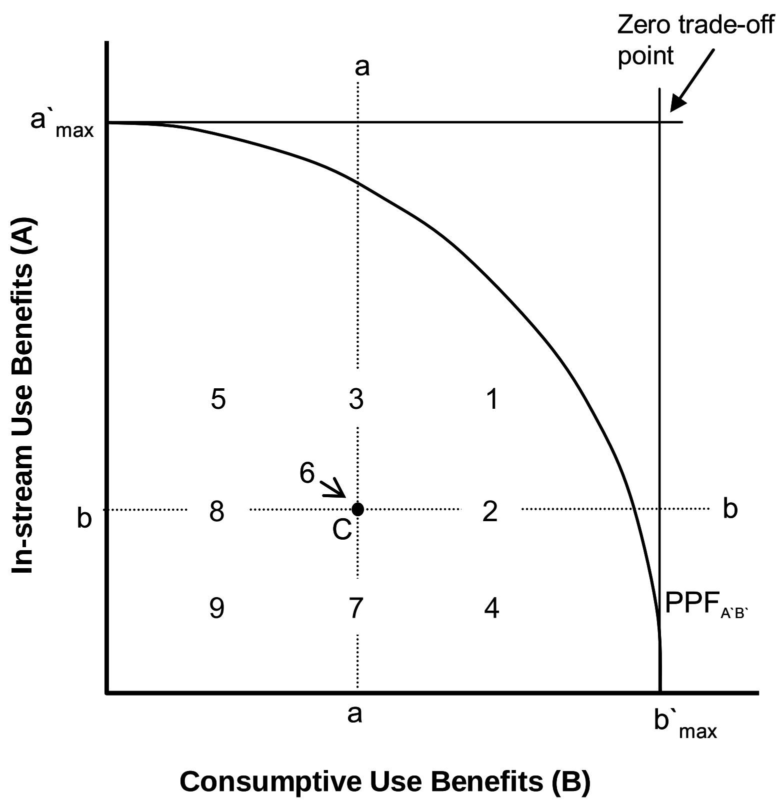


Figure 4.17: Graphical representation of the matrix of outcomes from a starting point *C*.

As noted above, an efficient outcome in the context of this research is restricted to those outcomes in the Pareto improvement set, that is outcomes 1, 2 and 3. An alternative (or extension) of this decision framework is the consideration of the implications of the scenarios which represent possible Kaldor-Hicks outcomes (4 and 5). In these cases where the environmental outcomes are at a point to the left of the line *aa* or below *bb* in Figure 4.17 TVA comes into play. Thus, there may be a case to adopt a scenario where the outcomes are found to be a win-loss pattern should the value of the gain in one outcome be higher than the loss of the other. That is, outcome 5 might be acceptable if it is judged that the loss in consumptive benefits is outweighed by the gain in environmental outcomes. This situation is displayed graphically in Figure 4.18 using the interaction between indifference curves and outcomes where point C is the baseline or starting outcome. While outcome D is outside of the quadrant cxy it is nonetheless on indifference curve μ_2, which because it is higher than μ_1, is the preferred option. Perhaps the simplest way of understanding the use of the threshold in this case is to frame the answer as a question similarly to Bennett (1999). Is the value of the slight decrease in consumptive use benefits (Δa) worth the increase in in-stream use benefits (Δb)? The value judgement by policy-makers in this case may well be that it is worth the slight decrease in environmental health indicator as this has not placed the riverine environment in an unacceptable position.

An extension to TVA is the notion of the Kaldor-Hicks criterion. In this study, as the issue of compensation under the Kaldor-Hicks criterion is not considered, the focus is on identifying Pareto improvements. In undertaking this however it is possible that a number of scenarios which produce outcomes where the Kaldor-Hicks criterion is applicable will be identified. A potential Kaldor-Hicks improvement is represented by point D in Figure 4.18.

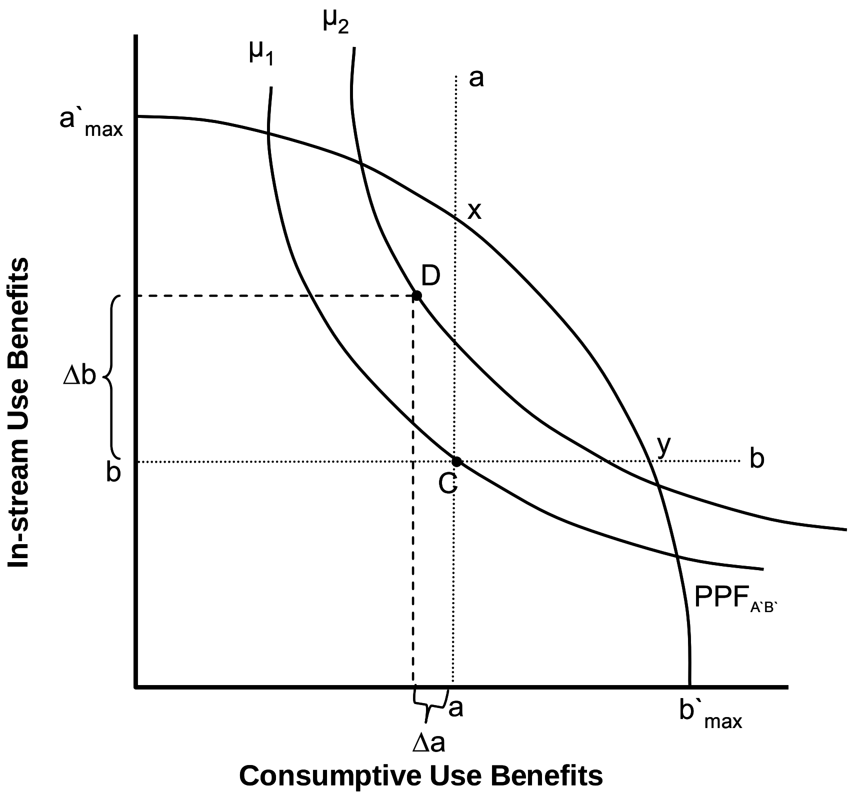


Figure 4.18: Threshold value analysis and the Kaldor-Hicks criterion.

This point (D) may represent an improvement in utility but as it lies outside of the thresholds set by society it is therefore considered unacceptable in this study. Instead the trade-off analysis incorporating the Pareto principle + TVA as presented forms the assessment framework to determine whether the use of SCF information makes society "better off". This will allow not only some inference to be made on the usefulness of the SCF information but also highlight how much there may be to gain from improvements in the ability to forecast streamflows.

4.5. Scenario development and seasonal climate forecasting

The scenarios tested in this study are based around the study hypothesis, that SCF can be used to assist in improving environmental flow management. Specifically, the method employed in the scenarios is to transfer the extraction of water from the river for consumptive use from dry to wet years resulting in higher streamflows (fewer "droughts") in the river system in the dry years. That is, increase the extraction of water in wet years and decrease the extraction in dry years. This straightforward set of scenarios is a step in determining first if the assessment framework meets the aim of developing an appropriate methodology and modelling framework. Secondly then, it allows inferences to be made on the efficacy of seasonal climate forecasting on environmental flow management in the Border Rivers catchment (testing the hypotheses).

Implementing the assessment framework (Figure 4.2) to undertake the assessment identified in Figure 4.16 requires a number of scenarios to be run to:

1. map out the trade-off curve;
2. identify the baseline scenario as a staring point for comparison. This point is identified as Ω in Figure 4.16 and is a key requirement for equations 4 and 7 to estimate the value of the forecast, potential value of the forecast and the TVA + Pareto assessment;
3. as part of equation 7, estimate the outcome of the proxy perfect forecast scenario (∂` in Figure 4.16); and,
4. a series of scenarios developed using SCF information. This will provide χ in equation 4 and Figure 4.16.

4.5.1. Estimating the trade-off curve

The combinations of output highlighted in the trade-off analysis alter as we move along the trade-off curve in response to changes in the allocation of one input variable to produce more of one type of outcome and less of the other. The input that is varied in this study is water, however specifically only one component of the total water supply, off-allocation, is being manipulated (see Figure 3.5). Earlier discussion in relation to Figure 3.5 noted that approximately 56 percent of water diverted in the New South Wales part of the catchment for the water years 1991 to 1999 was sourced by off-allocation (Abawi et al. 2001). This figure refers to diversions only and does not refer to the other parameters identified in Figure 4.5. While no overall estimate is available for the proportion of off-allocation water of the total water supply as defined in Figure 4.5 it is expected to be considerably less than 56 percent. Consequently, developing the trade-off curve for this study refers to only a portion of the frontier as the extremes of all water supply to either of the outcomes a_{max} or b_{max} (Figure 2.2) does not occur.

Nonetheless, to establish the efficacy of the SCF information a baseline, or without SCF curve, needs to be estimated for comparison purposes. To develop the trade-off curve a number of scenarios are run where more water is made available for one outcome, consumptive use, thereby reducing the outcome of environmental benefits as measured by hydrologic indicators. The lower bound for the trade-off curve is obviously zero. The upper bound was determined by finding the level of off-allocation availability such that no more off-allocation water was diverted in the simulation. I found changing the off-allocation cap from 100,000 ML to 200,000 ML did not change the level of off-allocation water diverted for a number of structural reasons primarily concerned with the level of on-farm storage and land developed for irrigated agriculture. Therefore the simulation to estimate the trade-off curve involves a gradual increase in the amount of off-allocation water that is made available to irrigators for each scenario from a minimum amount of 10 megalitres to 100,000 megalitres for each water year of the scenario period 1894 to 1994 (Table 4.7). *Ceteris parabus*, it is logical to expect that gradually increasing the availability of water will result in a gradual increase in economic returns and a decrease in environmental benefits. The outcome of this simulation may be used to map out the shape of the trade-off curve as shown in Figure 4.19.

Table 4.7: Setting the simulation off-allocation parameters for estimating the trade-off curve

Scenario number	Off-allocation cap
1	10 ML
2	10,000 ML
3	20,000 ML
4	30,000 ML
5	40,000 ML
6	50,000 ML (Baseline)
7	60,000 ML
9	70,000 ML
9	80,000 ML
10	90,000 ML
11	100,000 ML

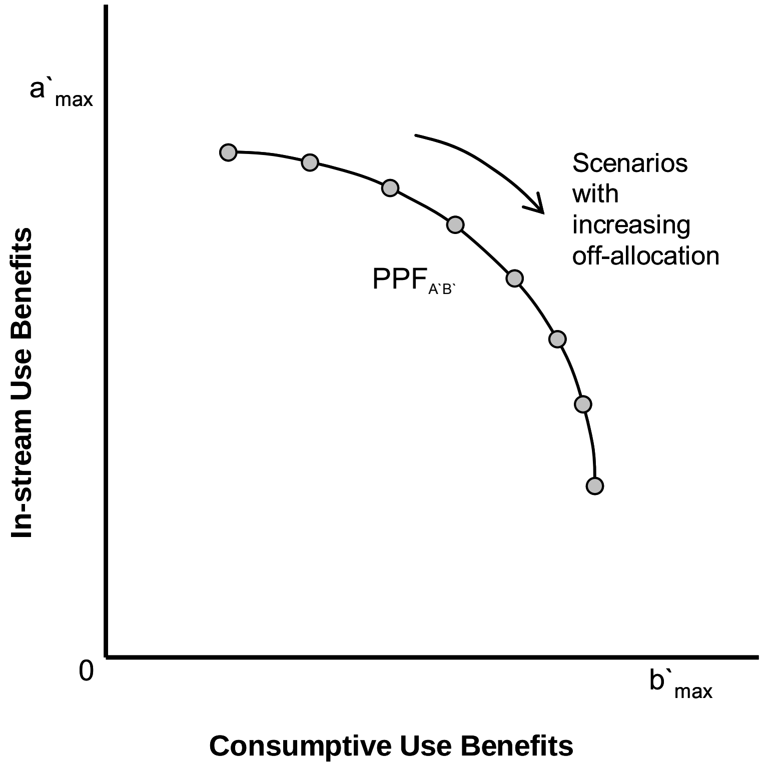


Figure 4.19: The results of the simulation gradually increasing access to off-allocation water.

The estimated trade-off curve in Figure 4.19 is developed assuming full technical efficiency, full use of available management skills, and the availability and use of information (see chapter 2). However this information does not include SCF information as it is currently being used only as qualitative background information (Long and McMahon 1996). It was also proposed in chapter 2 that the use of SCF information will result in a shift in the trade-off curve as portrayed in Figure 4.20 where the dashed line represents the starting "without forecast" trade-off curve and the solid line the new, but unknown, Trade-Off Curve$_{ab`}$. Given the estimated trade-off curve the next step is to estimate the baseline scenario for comparison against the SCF scenarios and then the SCF scenarios themselves.

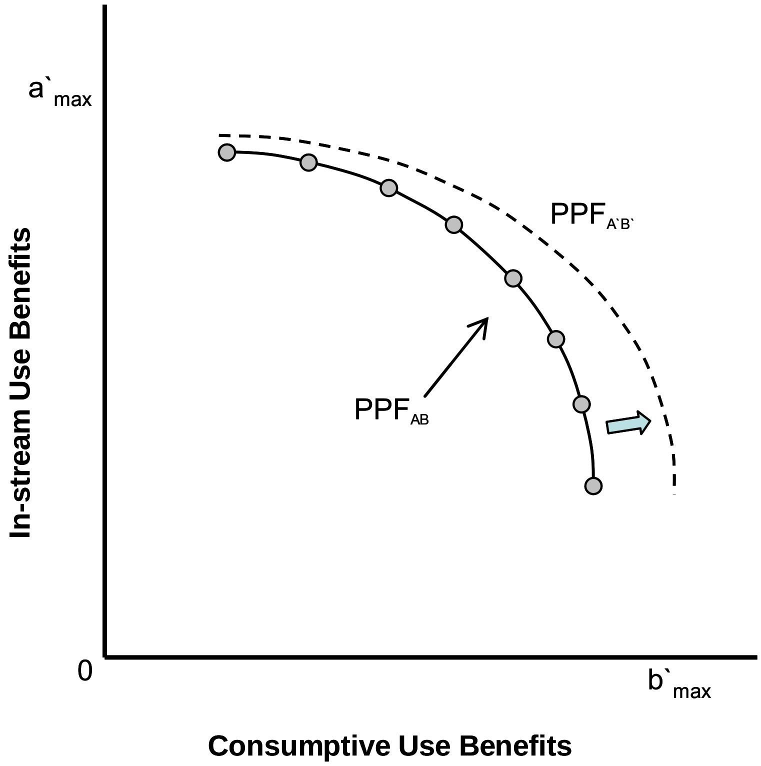


Figure 4.20: The stylised effect of successful use of SCF information on the trade-off curve.

4.5.2. Estimating the research scenarios

The study baseline scenario represents the status quo (without forecast) off-allocation decision and access settings used in the unmodified IQQM system file for the BRC. The settings include an off-allocation cap fixed for each water year of the entire simulation (1890–1996) of 50,000 ML for the NSW irrigation districts (Table 4.7). Justification for this choice is based on the fact that it is the mid-point in the availability of off-allocation scenarios assessed.

The ideal perfect forecast would entail having a perfect predictor for off-allocation access for each water year of the simulation. The fact that river management procedures influence whether a particular flow event is opened to off-allocation access makes this unachievable. Therefore the proxy perfect forecast developed was based on a high off-allocation cap for each year of the simulation. In effect this means that all flow events eligible for off-allocation can be harvested by irrigators if they have sufficient on-farm storage available. Should river management rules have a preset condition for an environmental or other requirement this would preclude off-allocation access being granted.

Seasonal climate forecasting methods have been and remain an area of active research. Consequently there are a range of forecast tools or indices that could be used to develop rules for managing off-allocation water access. The decision rules in the SCF scenarios are developed through considering the relationship between streamflow volumes and access to off-allocation, and the ability to forecast streamflow volumes.

A number of these forecast tools or indices have been subject to peer review and as such are scientifically valid. The tools tested are referred to as the SOI phases, SOI values and the ENSO and are outlined below. The forecast scenarios tested in this study are shown in Table 4.8 and an explanation of each major grouping follows.

Table 4.8: List of scenarios tested in this study

Scenario number	Seasonal forecast method	Scenario variable
1.	SOI Phases	October
2.		September
3.		August
4.		July
5.		June
6.		May
7.	SOI Values	-5 / +5
8.		-7 / +7
9.		-10 / +10
10.		-15 / +15
11.		-20 / +20
12.	ENSO	El Niño/La Niña

The following sections provide the justification of each scenario set as well as an outline of the scenario tested.

Using the SOI phases

The first set of forecast scenarios is based on the five phases of the Southern Oscillation Index (Stone et al. 1992) which are used to forecast the probable streamflow in the October to February period. Stone et al. (1992) used principal component and cluster analysis to group SOI values into five clusters or phases based on the magnitude of the SOI and on magnitude of the SOI over the current month and the direction of change from the previous month. Undertaking this analysis over the historical record results in each month over the period being assigned a phase which can then be used in forecasting, see (Abawi et al. 2001). The consistently negative and rapidly falling SOI phases correspond with below median rainfall (and streamflow) while the consistently positive and rapidly rising phases correspond with above median rainfall (and streamflow) in eastern-Australia (Abawi et al. 2001; Ritchie et al. 2004; Stone et al. 1992).

The table of SOI phases used in this study is in Appendix G [25]. Figure 2.18 demonstrated the forecast method through the use of a probability distribution where for a given probability the probable streamflow for an upcoming period can be forecast based on what occurred in the historical record in the years when the SOI was in a particular phase. An alternative way to view this forecast is through the use of pie charts (Figure 4.21). If the time series (1890 – 1996) of cumulative flows over

[25] Sourced from http://www.longpaddock.qld.gov.au/.

a particular period, say for example, October to February is split into terciles[26] then for the 107 records 36 would fall into the low tercile, 36 into the high tercile and 35 into the middle tercile.

The cumulative flows for October to February however can also be sorted into their respective terciles based on the SOI phase for a particular month. This is shown in Table 4.9 where the cumulative monthly flow is tagged according to which tercile it falls into where the terciles are identified at the bottom of the table. For example in 1890 the SOI phase is Rapid Rise (RR) and the cumulative flow is in the high tercile because it is greater than 453,476 ML. In pie chart form as shown by Figure 4.21 when the SOI phase in September is consistently negative or rapid fall this indicates lower flows for the period October to February because, more of the years had cumulative flows in the low tercile (< 193 024 ML) than the high tercile (453 476 ML). Conversely, when the SOI phase in September indicated higher flows for the period October to February, that is consistently positive or rapid rise, more of the years had cumulative flows in the high tercile than the low tercile.

Table 4.9 Sorting cumulative streamflows for Goondiwindi for October to February (1890 – 1996) into terciles based on the SOI phase in September.

	SOI phase in September					
Year	**CN**	**CP**	**RF**	**RR**	**NZ**	**Tercile**
1890				778,367		High Tercile
1891	252,254					Mid Tercile
1892		1,641,997				High Tercile
1893		356,549				Mid Tercile
1894		334,036				Mid Tercile
1895					307,689	Mid Tercile
1896	208,200					Mid Tercile
1897					766,420	High Tercile
1898					675,576	High Tercile
1899				266,330		Mid Tercile
1900			33,433			Low Tercile
1901			79,471			Low Tercile
1902			192,045			Low Tercile
Low Tercile			193,024			
High Tercile			453,476			

[26] Terciles are range of values of a physical variables (e.g. precipitation, temperature...) defined so as to sort into 2 sections 1/3 of the lower, of the average and higher values of a distribution that could represent a climatology, http://www.aviso.oceanobs.com/en/services/glossary/index.html.

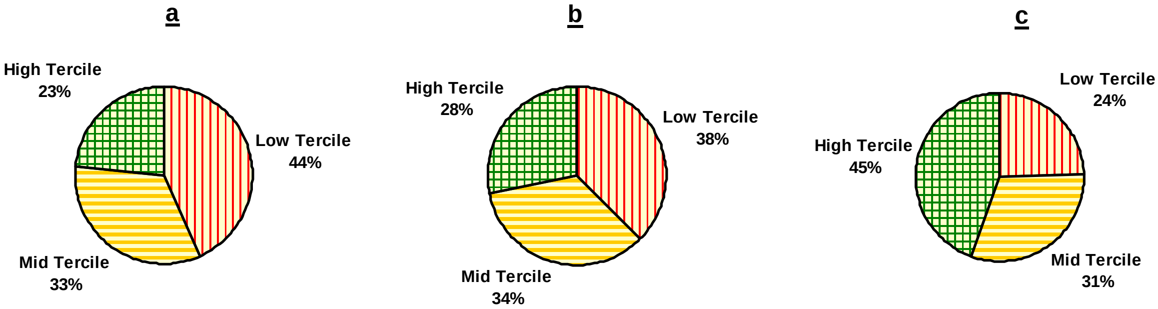


Figure 4.21: Comparing the occurrence of streamflow volumes at Goondiwindi by SOI phase types for September.

a: Consistently negative and rapid fall SOI phases
b: Near zero SOI phase
c: Consistently positive and rapid rise SOI phase

In addition to previous discussion in the literature review (see Figure 2.18) on the ability to forecast streamflow using the SOI phases, further rationale for the use of this forecast method is provided in Table 4.10. This table shows the relationship between the SOI phases and the river health indicators identified earlier (chapter 4.3.2). Recall that median annual flow ratio (MAFR) is the ratio of MAF volumes under "developed" conditions and MAF under "natural" conditions at a point in the catchment and provides an indication of the volume of water being diverted upstream of a particular point in the catchment. In this case the point is the Barwon River at Mungindi. Table 4.10 highlights that over all years (1890 – 1996), 47 percent of MAF of the Barwon River passes Mungindi, so 53 percent of the natural MAF volume has been removed from the river upstream of Mungindi. A relationship between forecasting of streamflow and the natural flow regime of the river is also highlighted in Table 4.10. Here proportionally less water, 41 percent, is removed in those years when the SOI phase was consistently positive (100 percent minus 59 percent), and considerably more, 65 percent, when the SOI phase was consistently negative (100 percent minus 35 percent). This corresponds with expectations of higher flows in years when the SOI phase was consistently positive and lower flows in years when the SOI phase was consistently negative.

Table 4.10: Comparison of flow statistics for the Barwon River at Mungindi (1890-1997) based on the SOI phase in May

SOI phase in May	SOI phases: Consistently negative	SOI phases: Consistently positive	All years
Median annual flow ratio (percent of Natural)	35 percent	59 percent	47 percent
Flow duration percentile	Proportion of flow duration percentile (PFlowDur)		
10 percent	0.46	0.89	0.63
30 percent	0.24	0.41	0.29
50 percent	0.39	0.64	0.46
80 percent	0.13	0.49	0.35

Source: adapted from Abawi (2001).

Flow duration percentiles indicate the percentage of time that flows of a certain magnitude (10 percent, 30 percent, 50 percent, 80 percent) are exceeded in the long-term over the 106 year timeframe based on the daily flows over the whole simulation period with no account being taken of the sequence or time of the year that they occurred. For example, high flows may be represented by the 10-percentile flow duration, (4 670 ML/day under natural conditions and 2 891 ML/day under developed conditions). The proportion of the flow duration percentile (PFlowDur) is the ratio of these flow duration percentiles under "natural" and "developed" conditions and reflects the change from the natural case. A value close to one indicates that irrigation development has had little impact on the proportion of time a particular flow percentile is reached, however a value close to zero indicates a situation where development of water resources is extensive with a consequential high impact on streamflows. For example, in Table 4.10 the 10-percentile PFlowDur for the full 107 year simulation is 0.63 which demonstrates a situation of considerable impact.

Similarly to MAFR, the ratios for the PFlowDur statistics for the subset of years when the SOI phase in May was either consistently negative (16 years) or positive (23 years) were calculated and are also reported in Table 4.10. The shifts in these statistics are consistent with the expectation of higher flows in years where the SOI phase in May is consistently positive, as opposed to when it is consistently negative. For example the 10-percentile PFlowDur is 0.89 in the subset of years when the SOI phase is consistently positive which is considerably closer to 1 (or natural) than when the SOI phase in May is consistently negative with a 10-percentile PFlowDur of 0.46. While this does not indicate that the riverine health will be improved when the SOI phase in May is consistently positive because over the 106 year period only 23 years have this SOI phase in May and they are separated by other year types. However the relationship is consistent with forecast flows and the differences observed here strengthen the case for examining the opportunity for use of SCF in this study.

SOI phase scenario outline

The SOI phase scenario is developed for use in the decision tree identified in Figure 4.6 whereby there are three year types forecast: dry; medium; and, wet. The three year types are defined by the terciles as shown in Table 4.11 which also identifies the off-allocation cap for the upcoming water year for each year type.

Table 4.11: SOI phase forecast decision rule

Year type	SOI phase	Predictor	Off-allocation cap
Dry	Consistently negative (CN) & rapid fall (RF)	Low tercile	10 megalitres
Medium	Near zero (NZ)	Mid tercile	50 megalitres
Wet	Consistently positive (CP) & rapid rise (RR)	High tercile	100 megalitres

Table 4.12 illustrates how the three scenarios are defined to obtain an estimate of the value of information. In the left hand column the proxy scenario for the perfect forecast, ∂ has a high off-allocation cap set that effectively allows irrigators to take

off-allocation water whenever it is available with a high annual limit. The without forecast scenario, Ω, has an off-allocation cap that again is fixed for the whole simulation and restricts off-allocation water supply to a baseline level. In the case of the forecast scenario, χ, a forecast is developed categorising each water year of the simulation into dry, medium or wet. The cap for each of these year types is then identified to allow more water to be taken in years that are forecast wet, no off-allocation water to be extracted in forecast dry years, and the baseline amount in the "other" years. It should be noted that off-allocation water is only able to be extracted if available in line with earlier notes regarding other uses.

The scenarios using the SOI phases were extended for this study by using an increasing lead time. Abawi (2001) concluded that streamflow volume for the next spring – summer period (October to February) could be forecast using SOI phases from May. Therefore, this study has tested a series of scenarios using lead times of four months (May SOI phase) to zero lead time (September SOI phase) as listed in Table 4.8.

Table 4.12: Example year type file

Year	Proxy perfect forecast (∂)	Without forecast scenario (Ω)	Forecast scenario (χ)			
	Off-allocation cap (GL)	Off-allocation cap (GL)	Forecast type	Year type category	Year type code	Off-allocation cap (GL)
1889	High cap	Baseline	CN or RF	Dry	1	Low cap
1890	High cap	Baseline	CP or RF	Wet	3	High cap
1891	High cap	Baseline	CP or RF	Wet	3	High cap
1892	High cap	Baseline	CP or RF	Wet	3	High cap
1893	High cap	Baseline	CP or RF	Wet	3	High cap
1894	High cap	Baseline	NZ	Medium	2	Baseline
1895	High cap	Baseline	CN or RF	Dry	1	Low cap
1896	High cap	Baseline	NZ	Medium	2	Baseline
1897	High cap	Baseline	NZ	Medium	2	Baseline
1898	High cap	Baseline	CN or RF	Dry	2	Low cap

Using the SOI values

The second set of forecast scenarios is based on the values of the Southern Oscillation Index (Stone et al. 1992; Chiew et al. 2000; Queensland Government 2005) which are used to forecast the probable streamflow in the October to February period. Implementing this set of scenarios entailed using the average SOI value for the August to September period[27] as the forecast variable which was partitioned in three year types for five values of the SOI (Table 4.13). Similar to SOI phases the year types used are dry, medium and wet. However for SOI value scenarios each year of the simulation is pooled by SOI values instead of SOI phases. For instance in (Table 4.13) years where the average SOI value for the August to September period is less

[27] The table of SOI values used in this study is in Appendix H .

than -5 are pooled into the dry year type, while those with average SOI values greater than +5 are pooled into wet years. Those years falling into the greater than -5 but less than +5 are pooled into the medium year type. Similar forecast scenarios are developed using average SOI value for the August to September period of 7, 10, 15 and 20.

Table 4.13: Defining the forecast scenarios and decision rules using the values of the SOI

Year type	SOI values					Predictor	Off-allocation cap
	-5 / +5	-7 / +7	-10 / +10	-15 / +15	-20 / +20		
Dry	x >-5	x >-7	x >-10	x >-15	x >-20	Low tercile	10 megalitres
Medium	-5< x* <5	-7 <x <7	-10 <x <10	-15 <x <15	-20< x <20	Mid tercile	50 megalitres
Wet	x >+5	x >+7	x >+10	x >+15	x >+20	High tercile	100 megalitres

* refers to the SOI value for the July to September period

A pie chart analysis shows that when the average SOI value indicated lower flows for the period October to February, that is x <-5, more of the years had cumulative flows in the low tercile (< 193,024 ML) than the high tercile (453,476 ML). Conversely, when the average SOI value indicated higher flows for the period October to February, that is x >+5, more of the years had cumulative flows in the high tercile than the low tercile. This supports the forecast decision rules listed in Table 4.13.

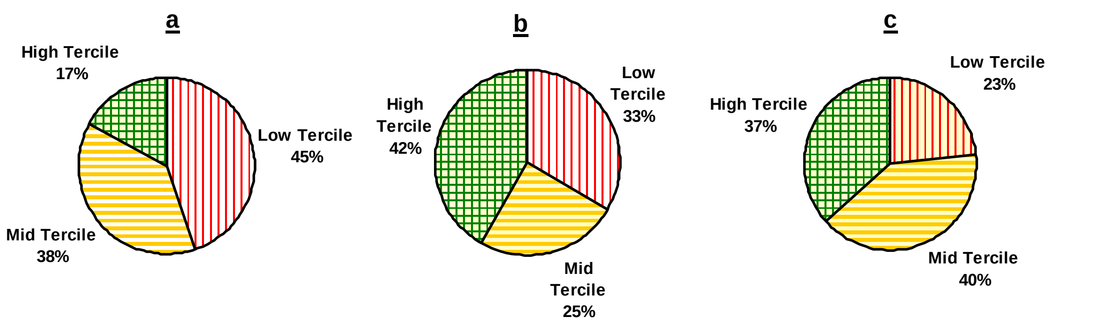


Figure 4.22: Comparing the occurrence of flow volumes by SOI value ranges.
a: <=-5
b: -5<x<+5
c: >=+5

The SOI values forecast scenarios were extended by progressively widening the bounds of the SOI value set from minus 5 to minus 20 as shown in Table 4.13 and listed as scenarios 6 to 10 in Table 4.8. The effect of this is to progressively increase the numbers of years falling into the medium band as the boundaries become more extreme. For example when the bounds for the scenario are SOI x<-5 there are 32 years in which the dry year type decision rule in Table 4.13 is triggered. However when the SOI bound is more restrictive such as x<-15 only 5 years trigger the dry year type decision rule.

Using the ENSO phases

A third scenario set tested was based on the partitioning of years into El Niño and La Niña years (Allan et al. 1996a). Rather than identifying a monthly phase each year is categorised as an El Niño or La Niña and each year type is associated with higher or lower rainfall and streamflow. In El Niño events expectations are for lower rainfall and streamflow while conversely during a La Niña event higher rainfall and streamflow is expected.

Figure 4.23 provides a probability of exceedence graph for years partitioned into ENSO types (Appendix I) analogous to Figure 2.18. To give these results similar context, when the ENSO year type is El Niño, the cumulative flow for October to February at Goondiwindi at 50 percent probability is approximately 223,000 ML as opposed to 440,000 ML when the ENSO year type is La Niña.

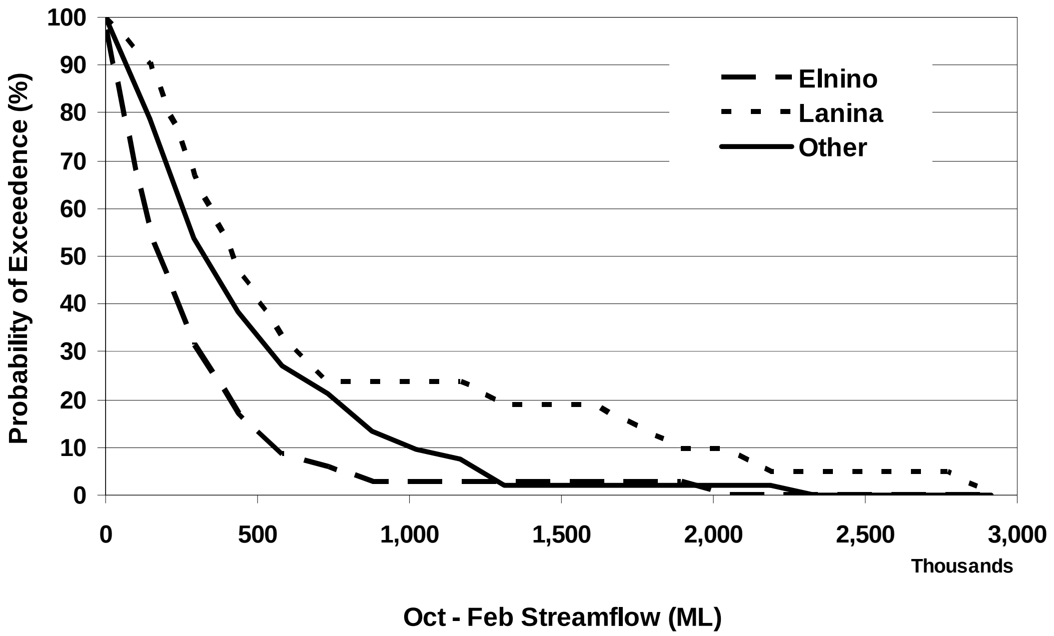


Figure 4.23: Forecast of streamflow for October to February (1890–1996) based on the ENSO years.

Similarly to the previous scenarios for SOI phases and values Figure 4.24 illustrates that when the ENSO year type indicated lower flows for the period October to February, that is El Niño, more of the years had cumulative flows in the low tercile (< 193,024 ML) than the high tercile (453,476 ML). Conversely when the ENSO year type indicated higher flows for the period October to February, that is La Niña, more of the years had cumulative flows in the high tercile than the low tercile. This supports the forecast decision rules listed in Table 4.14.

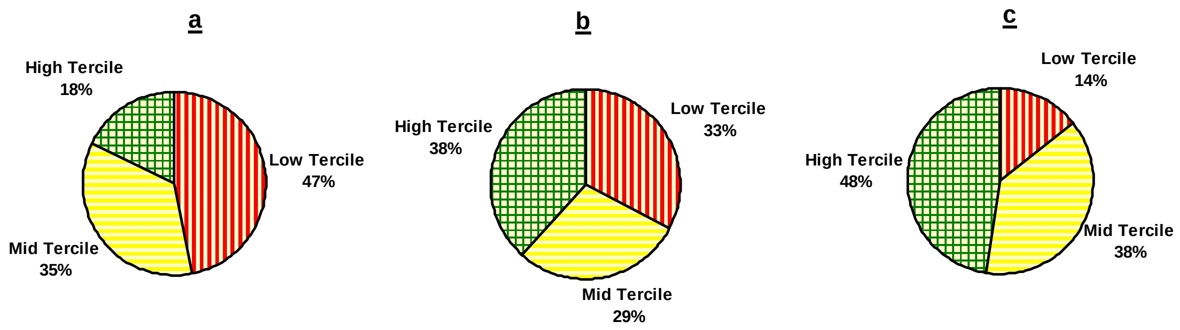


Figure 4.24: Comparing the occurrence of flow volumes by ENSO phase types.

a: El Nino
b: Other
c: La Nina

Table 4.14: Defining the forecast scenarios and decision rules using the ENSO phases

Year type	ENSO phase	Predictor	Off-allocation cap
Dry	El Niño	Low tercile	10 megalitres
Medium	Other	Mid tercile	50 megalitres
Wet	La Niña	High tercile	100 megalitres

Seasonal climate forecasting is a tool that can be used to provide information on the impact of climate variability on streamflows which in turn may assist in water management decision-making. The three SCF tools assessed in this study are all related to the ENSO phenomenon and are typically used to make probabilistic forecasts of cumulative streamflow volumes in upcoming months, October to February. That is, based on the ENSO indicator a certain volume of streamflow is expected at a certain probability over the October to February period. Given a particular forecast the challenge then becomes how to use the information in decision-making to improve outcomes.

4.6. Conclusion

ARMCANZ (ARMCANZ 1995) identified the goal of water reform as achieving the "highest and best value of the limited resource for community benefit whilst ensuring that use of the resource is ecologically sustainable" ARMCANZ (1995, p. i). This study assesses the efficacy of using SCF information to assist in achieving the goal of ARMCANZ using the trade-off model and the Pareto principle. The Pareto principle facilitates an assessment of the outcomes of using SCF information in water management decisions based on whether the outcome is an increase in either economic or environmental health outcomes (or both) without a decrease in the other.

Assuming a competitive relationship between outcomes, and a positive change to the environmental outcome is achieved, the size of this change is compared to the economic loss by the decision-maker to determine if it is sufficient for the policy to be deemed acceptable. The same approach is used should a positive economic outcome be achieved with a loss in environmental outcome (Bennett 1999; Streeting and Hamilton 1991; Jayasuriya 2004). In effect the decision-maker asks the question: "are the benefits of protecting the resource greater than the value of the extraction benefits that will be given up? (Bennett 1999, p. 2).

The TVA + Pareto assessment framework was chosen as having an appropriate level of theoretical underpinning and an acceptable level of analytical robustness. A key benefit of this approach is that decision-makers will gain a stronger understanding of the trade-offs inherent in different levels of use of the water resource making the implications of decisions more explicit. Therefore the method is used to inform decision-making processes as opposed to providing an answer indicating one option is better than another. The problem of subjective decision-making by policy-makers and/or politicians remains in a TVA approach; however, this can be partially overcome by an open process outlining analysis results which make plain the assumptions on environmental values used by decision-makers.

Implementing this assessment method requires estimation of the economic and environmental outcomes. This chapter has outlined the eco-environmental threshold meta-model (EETMM) and its components and has provided details of its implementation. The basis of the EETMM is the hydrologic model of river management and water use for irrigation which provides the data for the economic and environmental models. The economic model is used to estimate the regional gross margin as a proxy for of producer surplus. The environmental model draws on the relationship between hydrological processes and ecological factors to estimate impacts on the health of the riverine environment. Hydrologic indicators calculated by comparing the flow characteristics of the consumptive scenarios to that occurring in the natural or pre-development state are used as proxy for ecological outcomes. Both of these indicators are proxies for actual outcomes and as such have a level of uncertainty around them relating the achievement of actual outcomes.

The outcomes from the economic and environmental models are used to estimate the trade-off curve and the baseline or "without forecast" position. Against this is compared the series of scenarios developed using SCF information. This comparison is carried out at two levels using the notion of valuing forecast information and the Pareto principle and permits inferences to be made on the efficacy of SCF information use in achieving the study hypotheses.

5. Outcomes of the EET meta-model

5.1. Introduction

In the previous chapters I reviewed the literature relating to the hypotheses, issues surrounding the assessment of the use of forecasting information, and what methods have been used by others in undertaking related studies. In chapter 4, I outlined a method to undertake the impact assessment and introduced the meta-model for this assessment. Subsequently the components of the meta-model were explained and the decision framework for the assessment process outlined. This chapter reports the modelled results of the scenarios. These results are presented in two sections. The first section reports the results of the scenarios identified in Table 4.7 to simulate the trade-off curve for the baseline or without forecast scenario in a further two stages where each stage represents the x and y axes that comprise the trade-off curve.

The second section reports the results of the scenarios using the rules based on SCF information identified in Table 4.8. The chapter concludes with sensitivity testing of key assumptions within the production model and the results assessment using the decision framework developed in chapters 2 and 4.

5.2. Assessing the model and identifying the baseline

The first simulation comprises a number of scenarios where, step-wise from a minimum amount of 10 to a maximum 100,000 megalitres, more water is made available for one outcome, consumptive use, in order to estimate the trade-off curve (Table 4.7). The following sections present the results of this simulation on each outcome separately, before they are brought together in the frontier trade-off analysis in chapter 6.

5.2.1. The relationship between off-allocation and streamflow

The basis for modifying access to off-allocation is its relationship with streamflows. When high flow events occur, subject to a number of rules, irrigators may be given access to additional water supplies, namely off-allocation. Therefore an important first step is to test the strength of this relationship using the hydrologic model. Figure 5.1 illustrates the high correlation (r = 0.98) between modelled natural streamflow[28] volumes for the October to February period and access to off-allocation water. The implication of this correlation is that when high flow events occur off-allocation access is highly likely to be granted to irrigators. It follows therefore, that if a forecast system can accurately predict streamflow volumes for an upcoming period then we would also have a reliable prediction of off-allocation volumes. While this link is strong, the potential presence of other demands for water in the system means that a particular medium or high flow event does not necessarily lead to off-allocation access. These other demands include water that may have been ordered from the state-owned dam by other irrigators, or water that could potentially be set aside to ensure the end of system flow objective is met (DNRM 2003).

[28] Natural streamflow refers to flows modelled for "natural" or pre-development conditions where infrastructure and water extraction data are removed.

In addition, the volume of off-allocation water actually taken is restricted by factors such as pump size for extracting water from the river, on-farm storage size and the proportion of the on-farm storage available for water storage. For example, if the on-farm storage is already full, the off-allocation water will not be able to be captured.

This strong correlation, coupled with the importance of off-allocation water supply to irrigators (Figure 3.5); and the ability to forecast streamflows (Figure 2.18), justifies using off-allocation as the independent variable in this study. In order to assess the efficacy of the SCF information based rules to achieve the desired outcomes the trade-off curve and baseline scenario need to be identified for comparison purposes. The first step in this process is to determine how the consumptive use benefits change as access to off-allocation access is increased.

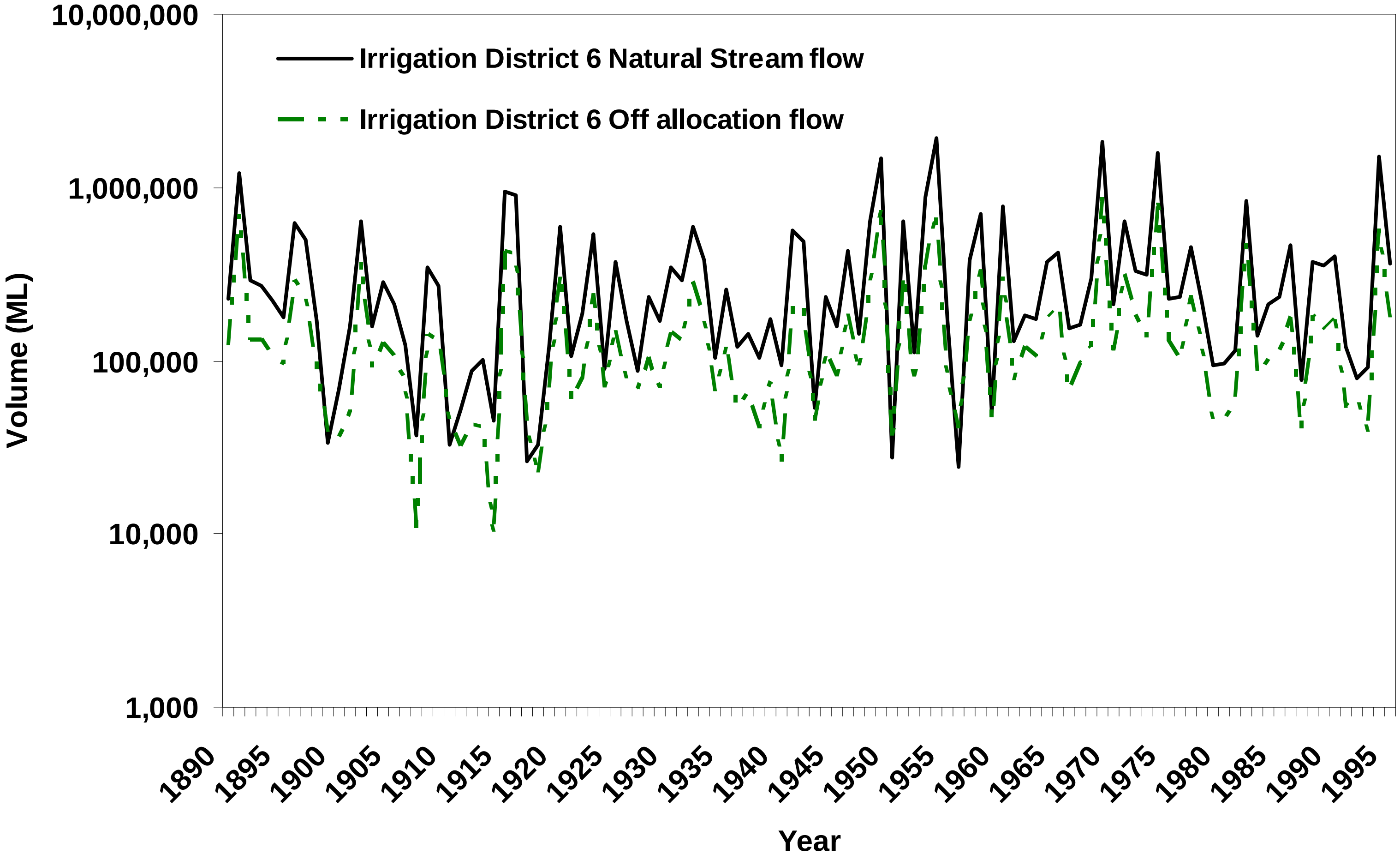


Figure 5.1: Relationship between streamflow and off-allocation water supply for October to February in the Border Rivers catchment, Australia.

(Data: Appendix J).

5.2.2. Impact on regional gross margin from increasing off-allocation availability

Regional gross margin (RGM), a proxy for producer surplus, is used in this study to represent regional economic impacts. In threshold value terms it represents the opportunity costs of agricultural production which can be incorporated into a trade-off framework and compared to the estimate of environmental change from each scenario. The first simulation was undertaken to test the logic of the production model against the results and to identify the baseline for comparison with the forecast scenarios. The simulation involves a gradual increase in the amount of off-allocation water that is made available to irrigators for each scenario from a minimum volume of 10 megalitres to a maximum of 100,000 megalitres for each water year of the scenario period 1894 to 1994. *Ceteris parabus* it is logical to expect that gradually increasing the availability of water will result in a commensurate increase output and therefore regional gross margin, assuming variable costs are constant.

The increase in returns is expected because as more water is made available there should be less crop stress, and therefore less crop yield losses in dry periods, leading to increased returns. In addition, there may also be a lagged effect of an increase in the size of planted areas in subsequent seasons which may also increase returns. This seasonal lag is due to the fact that the increased availability of off-allocation water in the current season is not taken into account when the plant area decision is made in the IQQM. This is because increasing availability does not necessarily translate to increased water supply unless sufficient streamflows are experienced. To account for this, when making the plant area decision, the IQQM does not directly take into account the off-allocation cap for that year. While the off-allocation cap does not impact on plant area decisions for that year, it does have an impact on subsequent years through a substitution effect. Where in a particular year the off-allocation cap is high, and the volume extracted is also high, this permits off-allocation water to be substituted for on-allocation water that is then stored for use in subsequent years. This has the effect of increasing the starting volume of supply in subsequent years, allowing more area to be planted in those years.

The results of this simulation, reported in Table L1 of Appendix L and shown graphically in Figure 5.2, indicate that the expectation of increased economic returns from increasing the availability of off-allocation water was largely met. The trend of RGM increases along with increased access to off-allocation as is shown by the mean (shaded box and values quoted on graph) and the median (cross bar on the vertical line). In addition the mean plus and minus 1 standard deviation (represented by the solid vertical line) highlight high but consistent variability. Close examination of Table L1 (Appendix L) however, reveals that there is a small sinusoidal trend after the 70,000 ML point. There is a small change in RGM downwards for the next two scenarios (80,000 ML and 90,000ML) before a recovery, but not to the same level of the 70,000 ML scenario.

While the overall trend is up, the last three scenarios outcomes are contrary to the expectations identified above. The explanation of this result is related to the rules incorporated into the calculation of RGM. The first rule explained in chapter 4 is the adjustment to yields based on the crop water depletion (refer to Table 4.3). As the plant areas get larger there are more stress periods placed on the crop because in an increasing number of years the amount of water available fails to match the requirement. If there is no access to water in a particular year then the crop yield will be decreased. In each scenario there are a number of years for an irrigation district where water demand is not matched by supply. In these years, depending on the penalty value (from Table 4.3), the crop yield decreases which in turn decreases RGM. The impact of this assumption is considered in the sensitivity analysis section of the results.

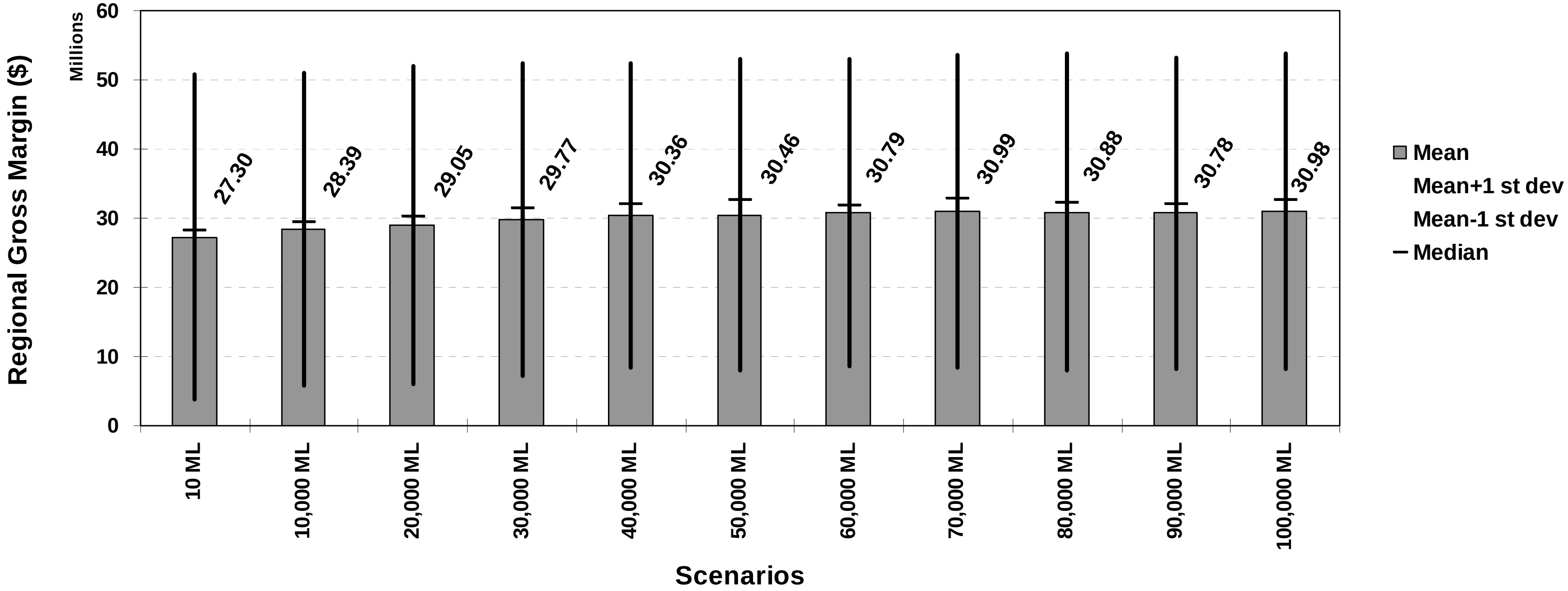


Figure 5.2: Simulation to test the outcome of gradually increasing access to off-allocation.

The second issue that is observed (from Table L1 Appendix L and Figure 5.2) is the relatively small change in mean regional gross margin as the access to off-allocation water supply increases. The mean result of $27.30 million from the first scenario where the off-allocation cap is restricted to 10 ML for each water year of the simulation period is quite close to the $30.98 million for the 100,000 ML off-allocation scenario. This difference of 12 percent could be considered small when compared to the large increase in potential access to off-allocation water.

There are several reasons for this relatively small difference. In the first instance there is a constraint on the area able to be planted to irrigated crops. An assessment of the simulation results identifies that the area planted in the IQQM model are close to maximum in a large number of years during the simulation for each scenario. Recall that in this study the irrigation districts account for 30,962 hectares of irrigated crop (Table 4.2). The summary plant area results presented in Figure 5.3 reveal that the large increases in potential water supply did not result in commensurately large increases in mean area planted to irrigated cotton. The reason for this is the relatively high level of water supply per hectare available for irrigation. In the study, irrigation districts' total irrigable area of 30,962 hectares has a corresponding 212,148 megalitres of on-allocation water supply. At full allocation this equates to a water availability of 6.8 ML/ha. The results in Figure 5.3 report the mean plant area and water for each scenario which when converted to ML/hectare at 60 percent efficiency similarly indicate that between 6 and 7 ML was applied to the irrigated cotton crops in the IQQM[29]. This indicates a much reduced reliance on off-allocation to grow a crop.

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[29] Crop water efficiency refers to the difference between water extracted from the river and that applied to the crop roots. Therefore if 10 ML is extracted from the river and 6 ML is applied to the crop roots the crop water efficiency is 60 percent. In this case, 4.15 ML converts to 6.9 ML at 60 percent efficiency.

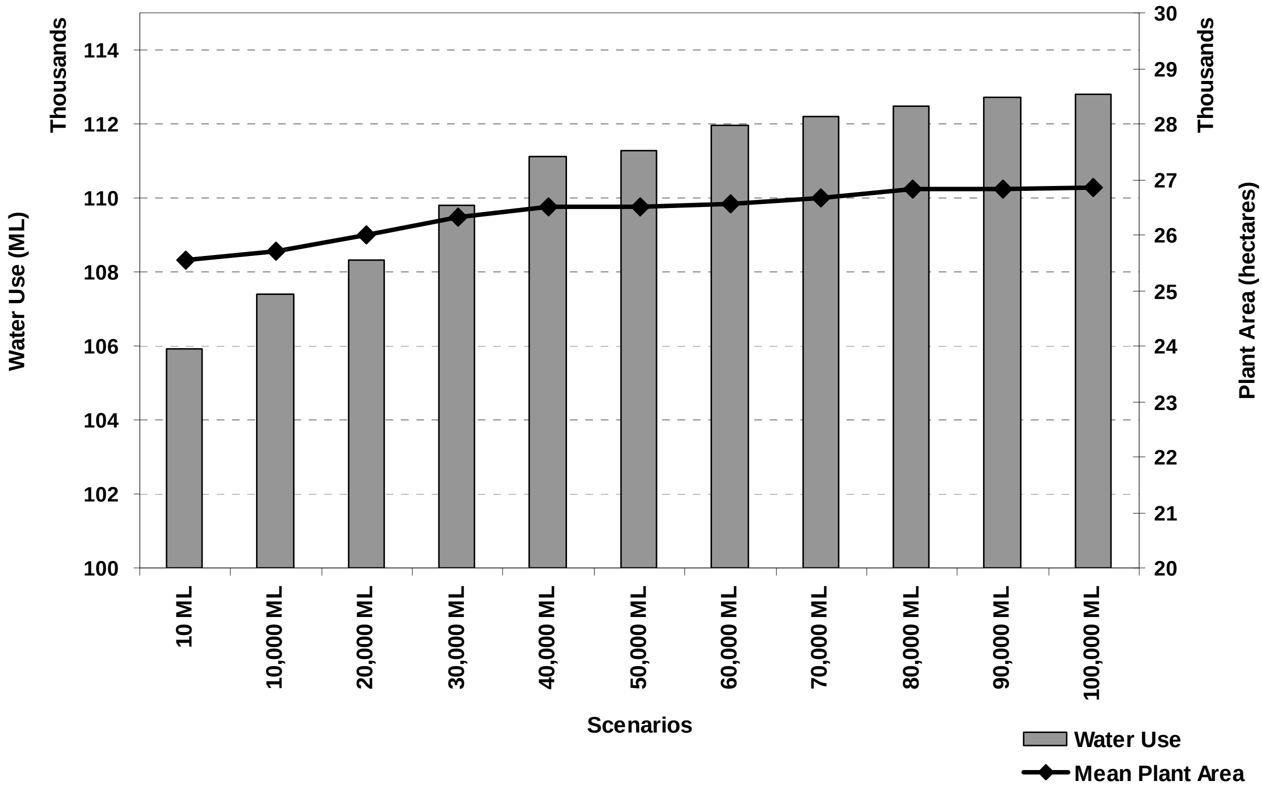


Figure 5.3: Mean areas of irrigated cotton planted and off-allocation water used for each scenario (1894 to 1994).

A further noteworthy feature of the results is the level of variability. This is highlighted in the box-plots in Figure 5.2 showing the variability by using one standard deviation around the mean. The increasing availability of water does not however lead to decreasing variability of RGM which might be expected. This outcome is related to the fact that increasing the off-allocation cap only increases the potential supply of water, not the actual supply of water. Therefore in very dry years, even if the off-allocation cap is 100,000 ML, there is a possibility that no additional off-allocation water will be available. For example in the 100,000 scenario when the off-allocation cap is set at 100,000 ML for each year of the simulation, the full amount of off-allocation is used in only 12 years out of 101 years. Due to lack of availability and bottlenecks between 1894 and 1994, less than 10,000 ML of off-allocation is used in 30 of the years.

A final conclusion from Figure 5.2 is that there are diminishing marginal returns of RGM to increasing availability of off-allocation water. This response, revealed in Figure 5.4 which shows the marginal RGM difference between scenarios, and confirmed by the fitted trendline[30], is largely as expected although somewhat variable due to the effects of bottlenecks in the system. Physical constraints include the fixed available area for planting irrigation crops which places a limit on the level of demand for water determining the amount extracted from the on-farm storage. Second the size and current levels of the on-farm storage can limit the amount of off-

[30] Derived using a logarithmic trendline in Microsoft Excel™ for the mean results with an R^2 of 0.83.

allocation water able to be extracted. If the on-farm storage already has a significant volume of water, then off-allocation water is unable to be accessed.

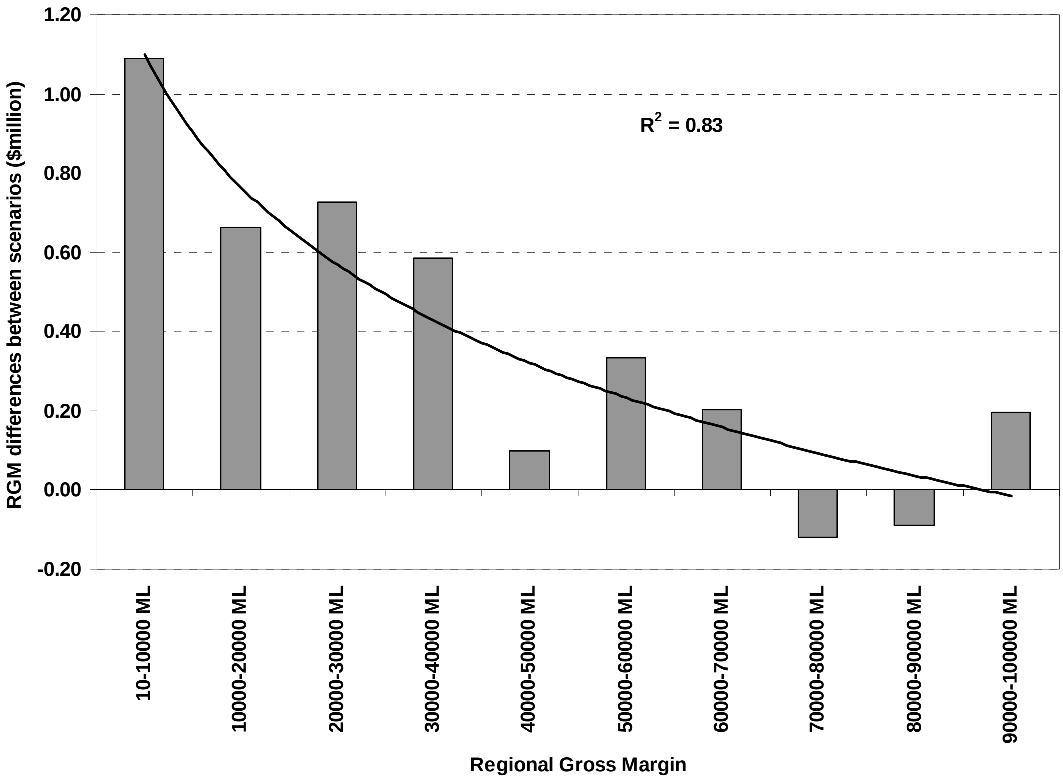


Figure 5.4: Trend of diminishing marginal returns for regional gross margin as access to off-allocation increases.

A different way of viewing the variability of RGM results shown by the standard deviation bars of Figure 5.2 is via the 10-year moving average of regional gross margin for three scenarios, the 10 ML, 50,000 ML and 100,000 ML off-allocation caps (Figure 5.5). The striking feature of the relationships shown in Figure 5.5 is the correlation between regional gross margins for all water availability scenarios. Additionally, there is a high variability of the moving average results across the scenario period and an inconsistent difference or spread between the 10 ML and the 100,000 ML off-allocation scenarios. This spread between the scenarios ranges between -1 and 30 percent over the simulation period identifying that at some point the 10-year rolling average of RGM for the 10 ML scenario is higher than that for the 100,000 ML scenario. This suggests that making additional water available does not necessarily translate into additional crop production and gross margin. There may be a number of causes for this but the primary ones applicable to this modelling are likely to be an inability to take advantage of the increased water supply because crops have already been planted and bottlenecks in the system stopping water being accessed. Finally, the 10-year moving average results emphasise that within the bounds identified in Figure 5.2 there are instances where RGM is low for extended periods of time. Use of the 10-year moving average of regional gross margin represents a summary of results over what is considered to be a realistic planning timeframe, but at the same time presents difficulties for deciding which scenario might be preferred.

The aim of simulating the impact on regional gross margin from increasing the availability to off-allocation from 10 to a maximum 100,000 megalitres was to estimate the x-axis of the trade-off model. Figure 5.6 presents the results of these scenarios in partial trade-off space in the absence of the results of the environmental impact analysis highlighting the pattern of increasing RGM as access to off-allocation water is increased. The next step in developing the trade-off curve is to estimate the environmental impacts from each scenario which will be combined with the results reported in Figure 5.6.

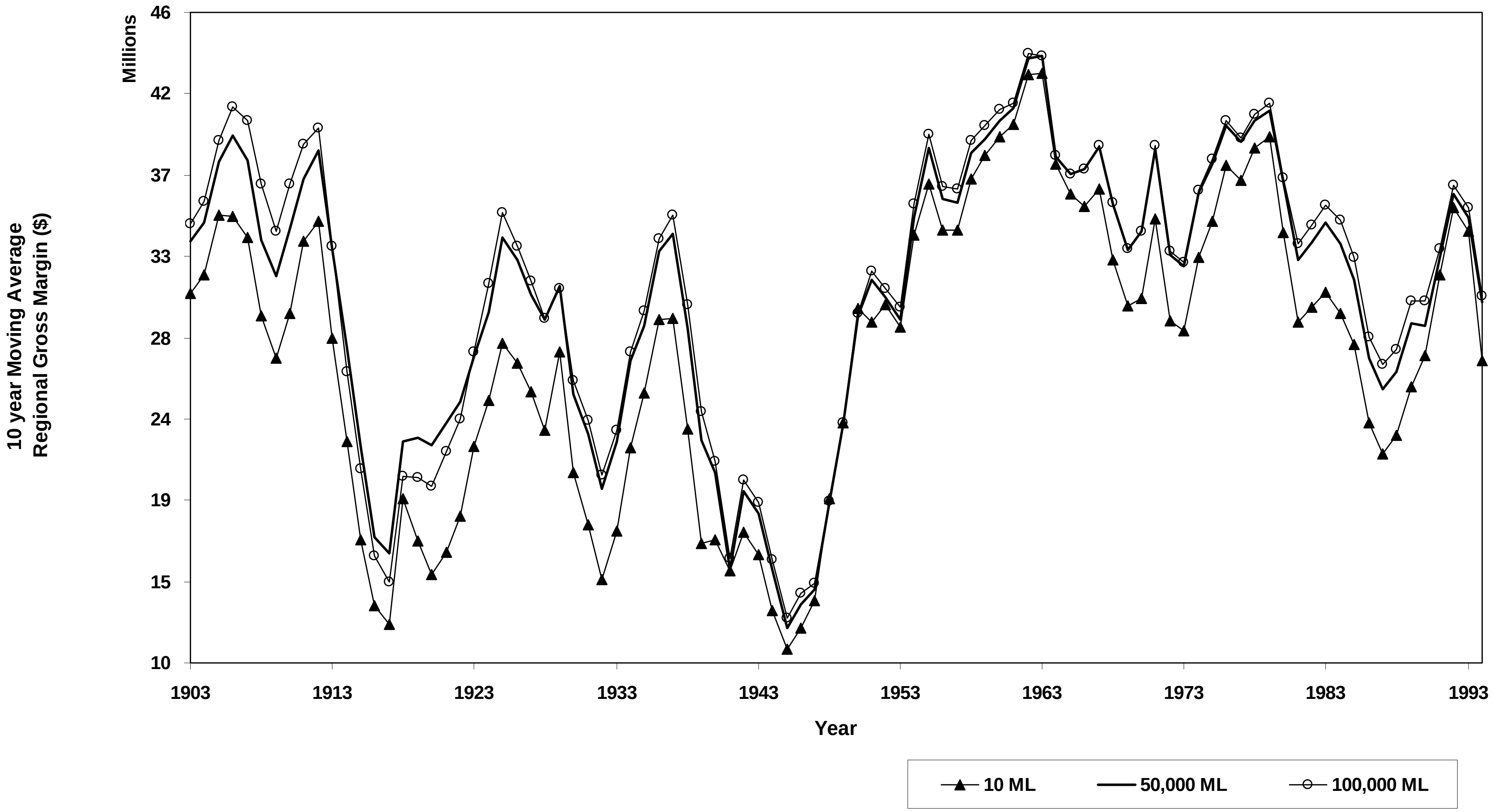


Figure 5.5: Assessing the variability of economic results by comparing the moving average of results.

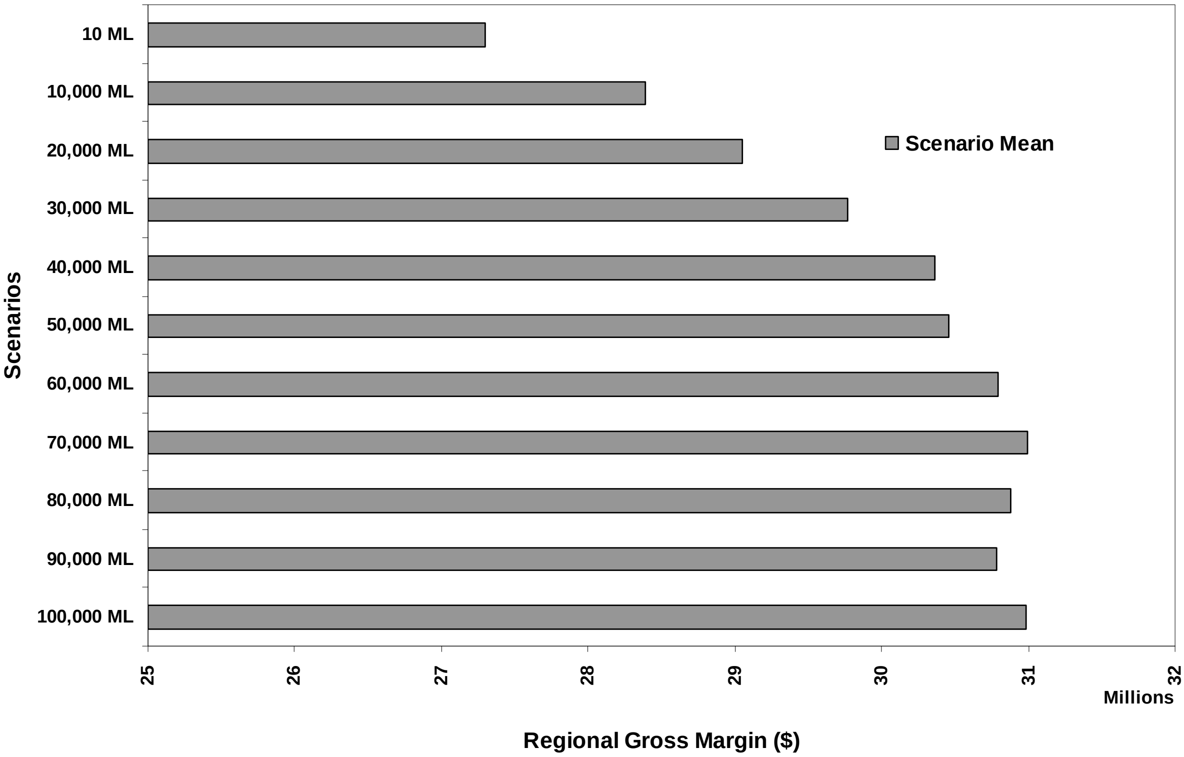


Figure 5.6: Regional gross margin results which form part of the trade-off curve.

5.2.3. Impact on environmental indicators from increasing off-allocation availability

The second major component of the modelling framework is used to assess the impacts of the various scenarios on riverine health. In relation to the production possibilities frontier this model is used to assess whether the environmental outcome from a scenario is a movement up or down the Y axis in Figure 4.19. The method selected to assess environmental outcomes is based on the natural flows paradigm and hydrologic indicator approach canvassed in the literature review. In this approach, judgements of the impact on the riverine environment from any scenario is undertaken by assessing changes in hydrologic indicators where the closer the hydrologic indicators are to the natural state (one) the healthier the river is assumed to be.

The assessment of environmental impacts was undertaken using a number of indicators - Mean Annual Flow Ratio (Mean AFR) and Median Annual Flow Ratio (MAFR) and the Proportion of Percentile Flow Durations (PFlowDur) (refer to section 4.3.2). Both Mean AFR and MAFR indicate the volume of water being diverted upstream of a particular point in the catchment while PFlowDur reveals the change in daily flows of certain magnitudes. The change in PFlowDur is assessed for the 2, 10, 50 and 80 percentile daily flows. An explanation of the calculation of the hydrological indicator that also provides an indication of the scale of each hydrological indicator is provided in Appendix C . In the case of the 0.50 PFlowDur the developed flow for the baseline (50,000 ML) scenario is 358 ML/day resulting in the ratio of 0.84 (358/426). For the 0.80 PFlowDur the developed flow for the Baseline scenario is 84 ML/day giving a ratio of 0.88 (84/95). The second feature of this table is the considerable difference in daily flows for flow duration indicators. This highlights the ephemeral nature of the river system in the BRC where flow magnitudes even under a natural state are zero for periods of time.

Table 5.1: The percentage of natural results are calculated by the ratio of the natural flow indicator and the developed flow indicator, in this case the baseline

Indicator	Flow (ML)		Ratio = percent of natural
	Natural	Baseline	
Mean AFR	737,794,	573,699,	0.78
MAFR	540,926,	397,090,	0.73
PFlowDur 2 percent	15,652,	12,990,	0.83
PFlowDur 10 percent	5,118,	3,727,	0.73
PFlowDur 50 percent	426	358	0.84
PFlowDur 80 percent	95	84	0.88

The indicators calculated in Table 5.1 can be used to assess the impact of scenarios developed using SCF information where if the SCF scenario results produce indicator scores below these the scenario is considered to have worsened environmental outcomes. Where the indicator scores are above these, the scenario is considered to have improved environmental outcomes.

For all the indicators, values close to one indicate little change from the natural case, while values further from one, either higher or lower, suggest greater change from the natural or pre-development flow conditions. The indicators are measured at the location of irrigation district 10 which is the final decision point in the catchment for this study and represents the end-of-valley location.

The impact of the increasing access to off-allocation on the health of the riverine environment is shown in Figure 5.7[31] while the table of results are reported in Table L.2 (Appendix L). The key points to note from the results are the relatively small level of change in indicators, the downward trend for some indicators and upward trend in others, and the level of change in annual flows.

The first feature to note from Figure 5.7 is that the changes in the values of the indicators are relatively small (see Appendix L Table L.2). In the cases of Mean Annual Flow (Mean AF) and Median Annual Flow (MAF) the percentage changes are of the order of two percent between the scenarios where off-allocation is severely restricted (10 ML scenario) and not restricted (the 100,000 ML off-allocation scenario). The small increase in overall water use shown in Figure 5.3 is a key factor in the quantum of change in these indicators. While small, for Mean AF and MAF, these changes nonetheless indicate that diversions above the point of irrigation district 10 have increased. In addition, for both of these indicators, as well as for PFlowDur 0.02 and 0.10, the trend is downward as the access to off-allocation water is increased.

[31] Note the scale is reduced from 0 to 100 because of clustered results. Changes in indicators are not as large as implied in the graph.

This downward trend is logical as in the lower off-allocation scenarios the off-allocation cap would have been reached relatively early in the water year leaving a number of flow-events to pass by without off-allocation access occurring. As the off-allocation cap is increased these flow events are able to be accessed for off-allocation extraction thereby reducing flows to downstream. This is highlighted in an example (Figure 5.8) where off-allocation is pumped for a longer period at irrigation district 8 in the 100,000 ML scenario when compared to the 50,000 ML scenario resulting in higher overall diversions for that year. In this example the flows downstream of irrigation district 8 will have been reduced by the area under the curves. Flows for the 100,000 ML scenario will have been reduced by a greater amount than the baseline with the size of the reduction being equal to the area between the two curves.

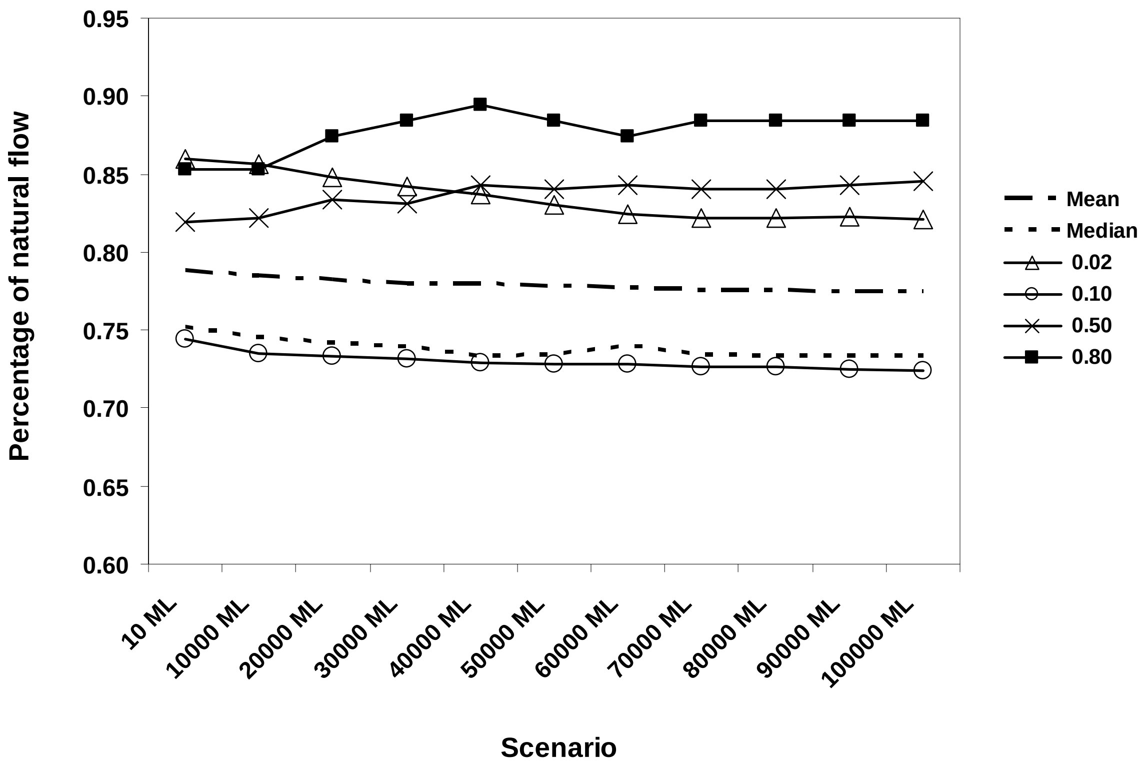


Figure 5.7: The impact on an hydrologic indicator from increasing access to off-allocation water.

On the other hand the trend for median daily flows, represented by the flow duration percentile of 0.50, is up over the scenarios and low flow (0.80 PFlowDur), increases up to the 40,000 ML scenario before decreasing marginally and plateauing as off-allocation access increases. The natural daily flow for these percentiles is 426 ML/day and 95 ML/day respectively. Again this result is largely as expected. When the access to off-allocation is restricted, irrigators use a high proportion of their on-allocation water. This increases the air-space in the state owned dams where licensed allocation is stored. Consequently, a larger proportion of the smaller flows are captured in these dams as opposed to passing over the dams when full. This results in the flow being lower for a given percentile. Hence there is a decrease in the flow duration indicators for the lower percentile flows of 0.50 and 0.80. Conversely, for the high off-allocation access scenarios, less on-allocation is used resulting in less airspace in the dams. Thus there are more dam spills and a higher proportion of the low flows being passed through the dams leading to higher flow duration indicators for the lower percentile flows of 0.50 and 0.80.

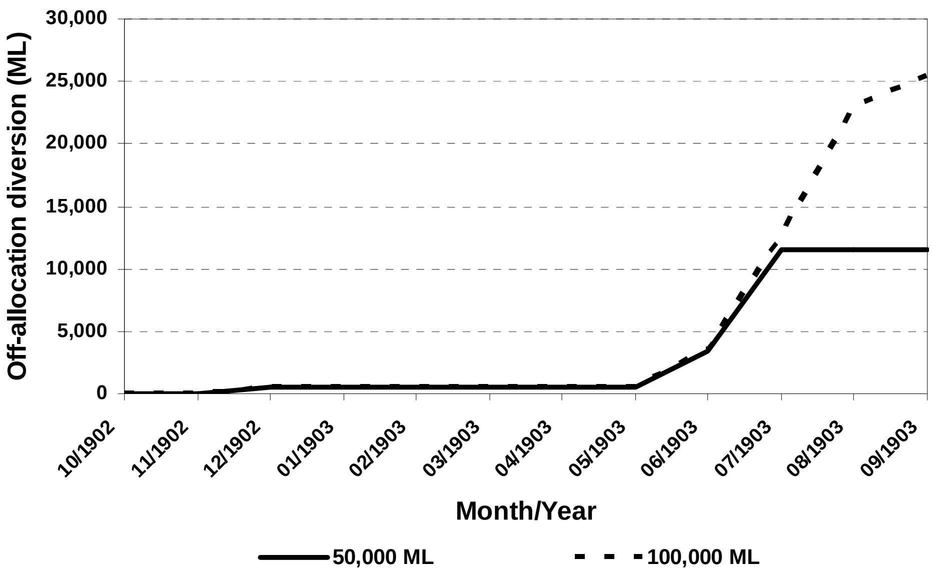


Figure 5.8: Cumulative diversions of off-allocation water for irrigation district 8 for the 1902-03 water year.

Some perspective of the scale of changes between scenarios can be gained by the use of a series of comparisons between simulations. Figure 5.9 provides an illustration of the change in flow from natural by highlighting the difference in total annual flow between the natural flow and 10 ML scenario at the location of irrigation district 10. In this graph the black columns show the total annual natural flow while the grey columns represent the difference between the natural flow and the flow when irrigation and other consumptive use is included in the simulation for the 10 ML scenario. The decrease in annual flows is both variable and at times proportionally significant. For example in 1939 the annual total natural flow of 128,910 ML is reduced by 37 percent to 81,177 ML.

Further perspective is gained by comparison of the total annual flows for the 10 ML and 100,000 ML scenarios in Figure 5.10. In this graph the total annual flow for the 10 ML scenario is taken away from the 100,000 ML scenario such that a negative data point indicates that the flow in the 100,000 ML scenario is less than the 10 ML scenario. It also shows that in a number of years the 100,000 ML scenario has total annual flows greater than that of the 10 ML scenario.

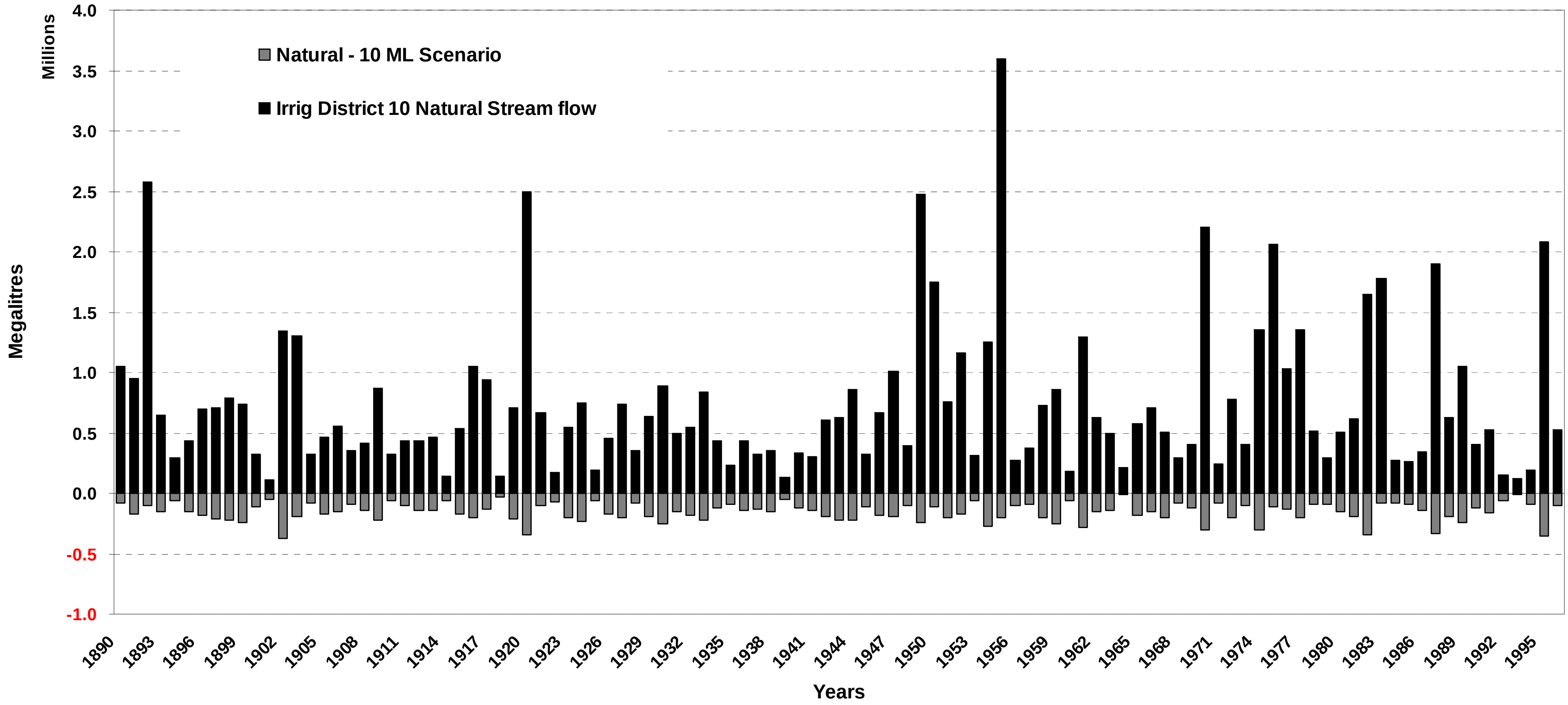


Figure 5.9: The difference in total annual gauge flow between the natural and 10 ML scenarios for irrigation district 10 highlighting the scale of changes between scenarios.

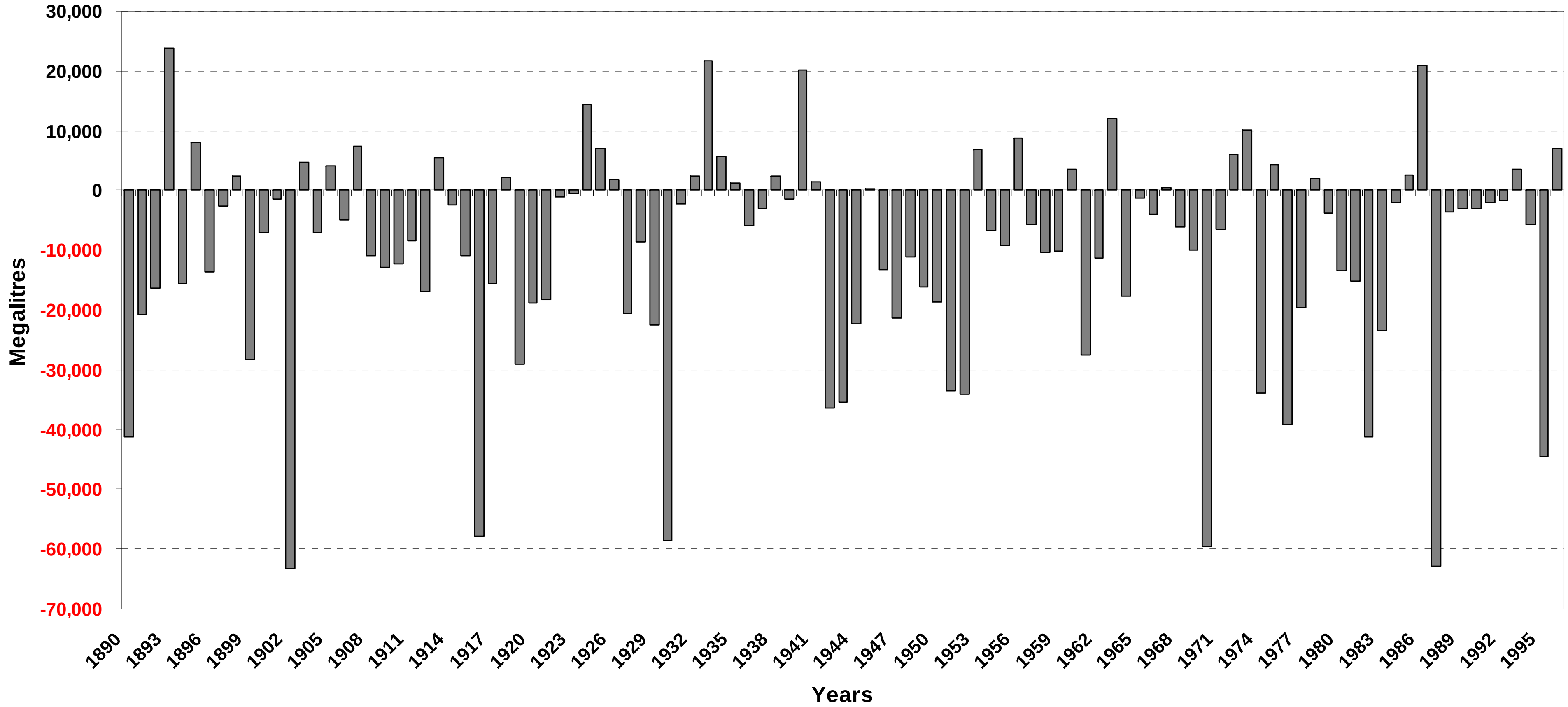


Figure 5.10: Difference in total annual flow between the without forecast 10 and 100,000 ML scenarios at irrigation district 10.

The results of the environmental analysis provide the second (y) axis to complete the trade-off curve referred to in Figure 4.19. Data from Table L.2 (Appendix L) is combined with the regional gross margin results from Figure 5.6 to estimate the frontier. Two examples highlighting different trade-off outcomes are shown in Figure 5.11 and Figure 5.12.

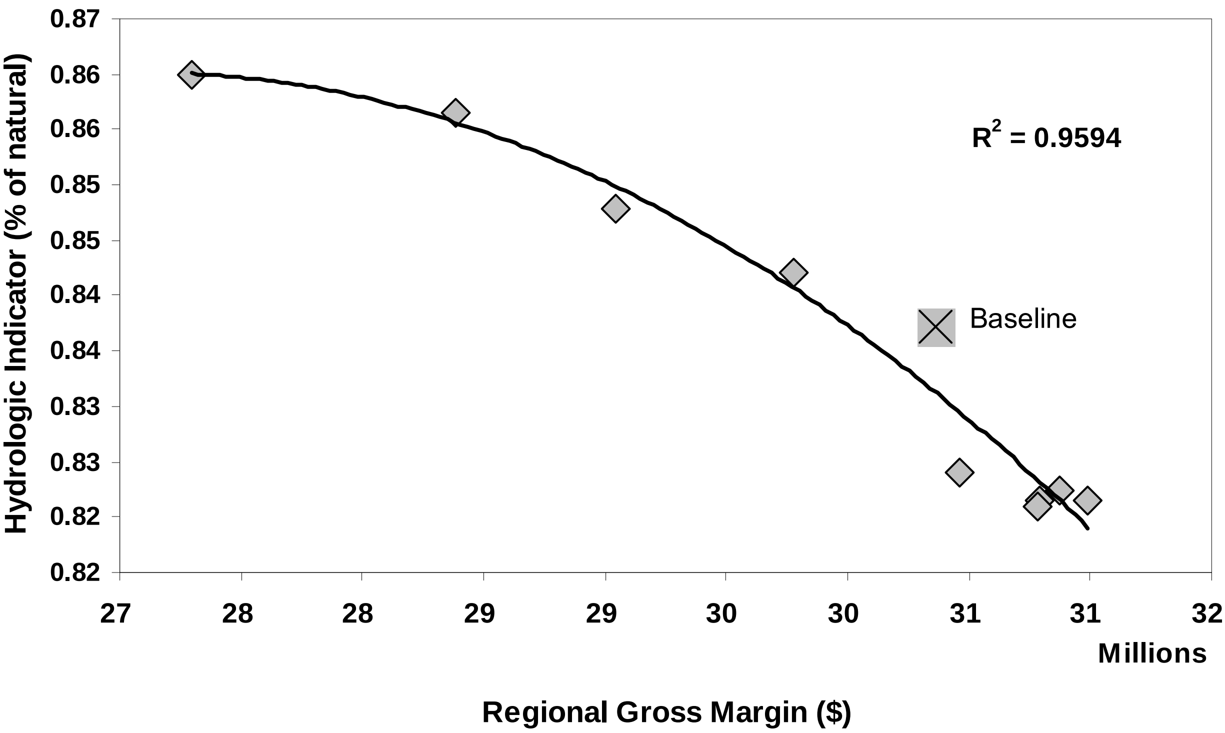


Figure 5.11: Trade-off curve using RGM and PFlowDur 0.02 depicting the expected trade-off between outcomes.

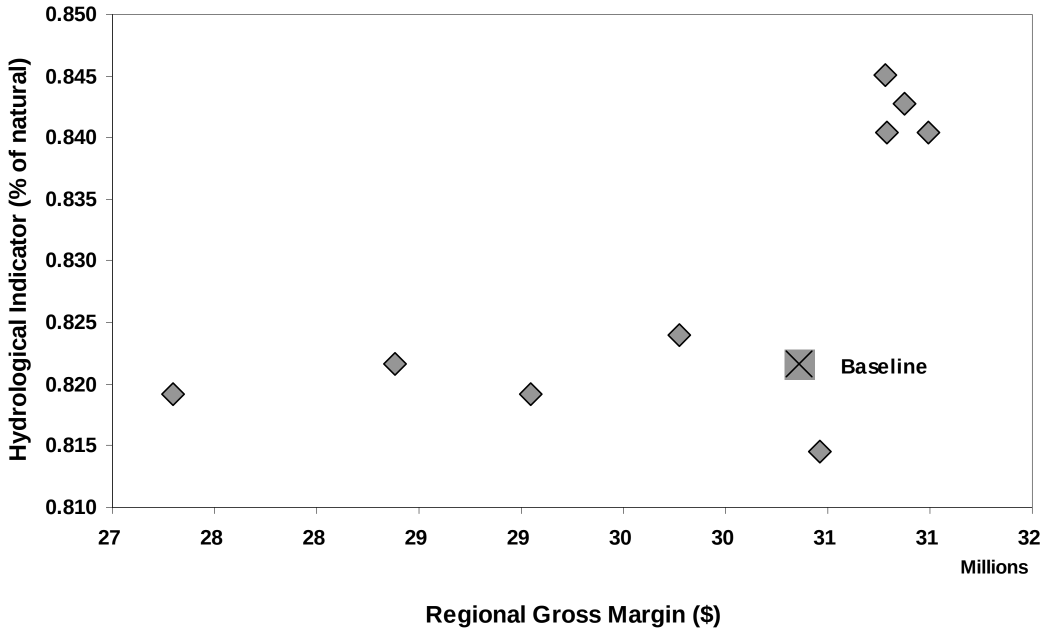


Figure 5.12: Trade-off curve using RGM and PFlowDur 0.5 showing no that trade-off in outcomes exists.

While it is noted that reducing the y-axis scale because of clustered results serves to highlight the relationship between the two outcomes (due to the small changes in the results in the hydrologic indicators), it is clear that the trade-off relationship exists for the RGM- PFlowDur 0.02 frontier, but not for the RGM-PFlowDur 0.5 frontier. This highlights that increasing access to off-allocation water has more of an impact on high flows (for which PFlowDur 0.02 is an indicator) than median flows (which are indicated by PFlowDur 0.5). As the actual announcement of access to off-allocation water in any given water year is related to higher levels of streamflow this result is not unexpected. The remainder of the trade-off curves are reported in Appendix K , and of the six hydrologic indicators tested, four show competitive relationships between the two outcome types (economic and environmental) and two do not show competitive relationships. These frontiers are estimated in this case using the trend line feature in Microsoft Excel with the R^2 reported to show the strength of the relationship. In testing hypothesis 1 the trade-off curve estimated in Figure 5.11 is used to compare outcomes of the scenarios based on SCF information. However, before this is undertaken a baseline scenario representing the “without” forecast case must be adopted.

5.2.4. Identifying the baseline for assessing SCF scenarios

The final use of the first simulation is to identify the baseline scenario which will be used as a benchmark from which to compare the scenarios based on seasonal climate forecasting and to test the study hypotheses. For the purposes of this analysis the 50,000 ML off-allocation cap scenario was chosen as the baseline.

This point is the mid-point in the availability of off-allocation scenarios assessed as increasing the off-allocation access above 100,000ML had a minimal impact on areas planted, off-allocation water used and RGM. When combined with the fact that because the Border Rivers system file for the IQQM model used in this study needed to be adapted to allow changes to the annual off-allocation cap, the mid point was the logical baseline point.

5.3. Simulation 2 – assessing the efficacy of the seasonal climate forecast scenarios

Two methods were used to assess whether the impact of forecast information will either improve the regional gross margin of irrigated agriculture or produce conditions that will lead to improved riverine environmental health. For the first method a forecast (χ) is said to have value if the outcome is less than gains from the use of a perfect forecast ($\partial`$), but an improvement on the baseline scenario (Ω), that is when:

$$\Omega < \chi < \partial`.$$

The greater is x, the greater the value of the forecast. For RGM, x can be estimated as a real number, while it is conceptual for environmental outcomes.

The second assessment method introduced in chapter 4 calls for each of the evaluations, economic and environmental, of the scenarios to be scored as a win or loss (Table 5.2). Then adopting the Pareto principle, scenarios with either a win-win or a win-no change result are identified as potential Pareto improvements and therefore potentially useful. Outcomes 1, 5 and 6 represent Pareto improvements. On the other hand, outcomes 2, 3, 4, 8 and 9 are sub-optimal, because the increase in one outcome results in a decrease in the other.

Table 5.2: Matrix of outcomes identifying potentially useful scenarios

		Environmental outcome *(Proportion of flow duration percentile)*		
		Increase	**No change**	**Decrease**
Economic outcome *(Change in regional gross margin)*	**Increase**	Pareto improvement	Pareto improvement	Possible Kaldor Hicks
	No change	Pareto improvement	–	–
	Decrease	Possible Kaldor Hicks	–	–

5.3.1. Impact on regional gross margin from using SCF based scenarios

Similarly to the process for assessing the logic of the model, the SCF scenarios were assessed through the use of regional gross margin. In this study an increase in average RGM for a scenario over the baseline scenario represents an economic improvement. However, as has been noted in Figure 5.5 which showed results from shorter 10-year periods, the use of an average across the complete scenario period is not without its issues.

The second simulation tested a range of SCF scenarios where the off-allocation access rules were based on the years categorised into:

1. SOI phases;

2. SOI values; and,
3. ENSO phases - El Niño and La Niña.

The specification of these scenarios can be found in Table 4.8 and are explained in section 4.5.

The summary economic results of ENSO and SOI phase scenarios are presented in Figure 5.13. Table L.3 (Appendix L) also presents results for the 10 ML, 30 000 ML, 40,000 ML, 50,000 ML and 100,000 ML scenarios for comparison with these results . Note that the 50,000 ML scenario represents the baseline scenario (Ω) and the 100,000 ML represents the proxy perfect forecast scenario (∂`). Perusal of Figure 5.13 (where the shaded box and reported numbers refer to the mean and the vertical lines to one plus and minus standard deviation), reveals that the SOI phase scenario closest to the baseline scenario is the June SOI phase scenario which at $30.14 million is 1.0 percent less than the baseline scenario (Table L.3). At $29.60 million the August SOI phase scenario is 2.9 percent less than the baseline scenario. The ENSO scenario at $30.60 million is slightly above the baseline by approximately 0.5 percent.

The first point to note is that the mean RGM results for the SOI phase scenarios lie within the bounds of the outcomes for the 10 ML and 100,000 ML scenarios. This is as expected because the 10 ML scenario restricts the ability of irrigators to access off-allocation water to effectively nil. As access to off-allocation water increases we expect that higher levels of production and RGM will result. At the upper end the 100,000 ML scenario permits irrigators to access what is effectively as much off-allocation water as they can take with a constant set of capital such as on-farm storage and area developed for irrigation. Therefore it is expected that the 100,000 ML scenario would provide a ceiling or upper limit of RGM outcomes, the 10 ML scenario would provide the floor for RGM outcomes and the SCF scenario outcomes would lie between these two outcome levels.

Second and more importantly, the outcomes of the SOI phase scenarios are all less than the baseline scenario i.e. the 50,000 ML scenario. The analysis of the mean RGM outcomes for the SOI phase scenarios in line with the theoretical value of information (VOI) indicates that there is no value in adopting any of the SOI phase forecast decision rules. In this analysis

$$\Omega < \chi < \partial` = \text{False}.$$

However for the ENSO scenario

$$\Omega < \chi < \partial` = \text{True}$$

as there is small value of forecast of approximately $145,000.

The next set of scenarios based on the SOI values has a mixed set of outcomes. In this simulation no lead time was assumed as the rules were based on average SOI values for the August–September months for each year of the simulation.

The summary economic results of this simulation for the first three scenarios SOI <-5,>+5, SOI <-7,>+7 and SOI <-10,>+10 similarly to the SOI phase scenarios show zero value of information, i.e.

$$\chi < \Omega < \partial` = \text{False}.$$

However, two scenarios (SOI <-15,>+15 and SOI <-20,>+20), present results, \$30.92 million and \$30.81 million respectively. These indicate a positive value of information when compared to the baseline scenario result of \$30.46 (see Figure 5.14[32]). In these cases the mean RGM for the forecast scenarios (χ) are above the Baseline scenario (Ω). While these results are relatively small at \$463,839 and \$355,858 above baseline or (1.5 and 1.2 percent respectively), they are nonetheless positive. Therefore the relationship:

$$\partial` > \chi > \Omega = \text{True}.$$

[32] In the figure the shaded box and reported numbers refer to the mean and the vertical lines to one plus and minus standard deviation.

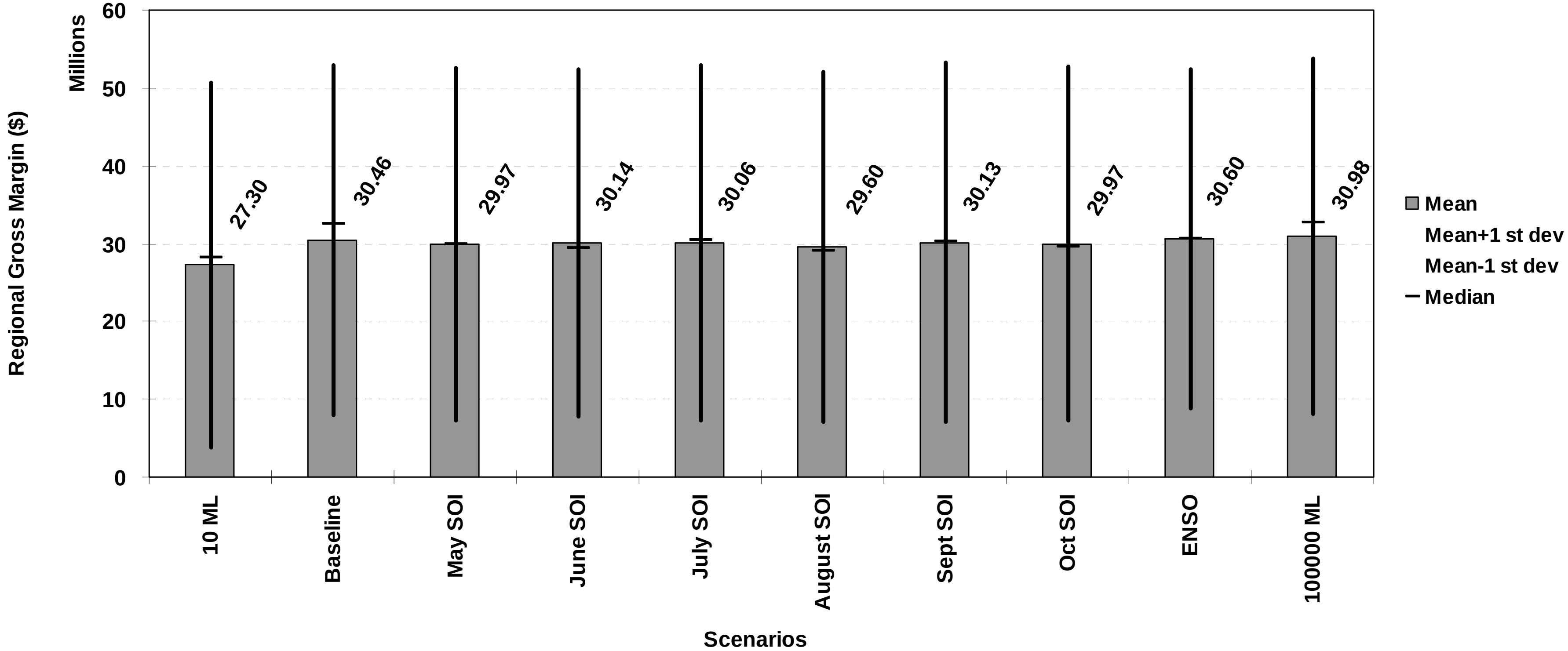


Figure 5.13: Economic results of simulation testing the outcome of regimes based on the SOI phases.

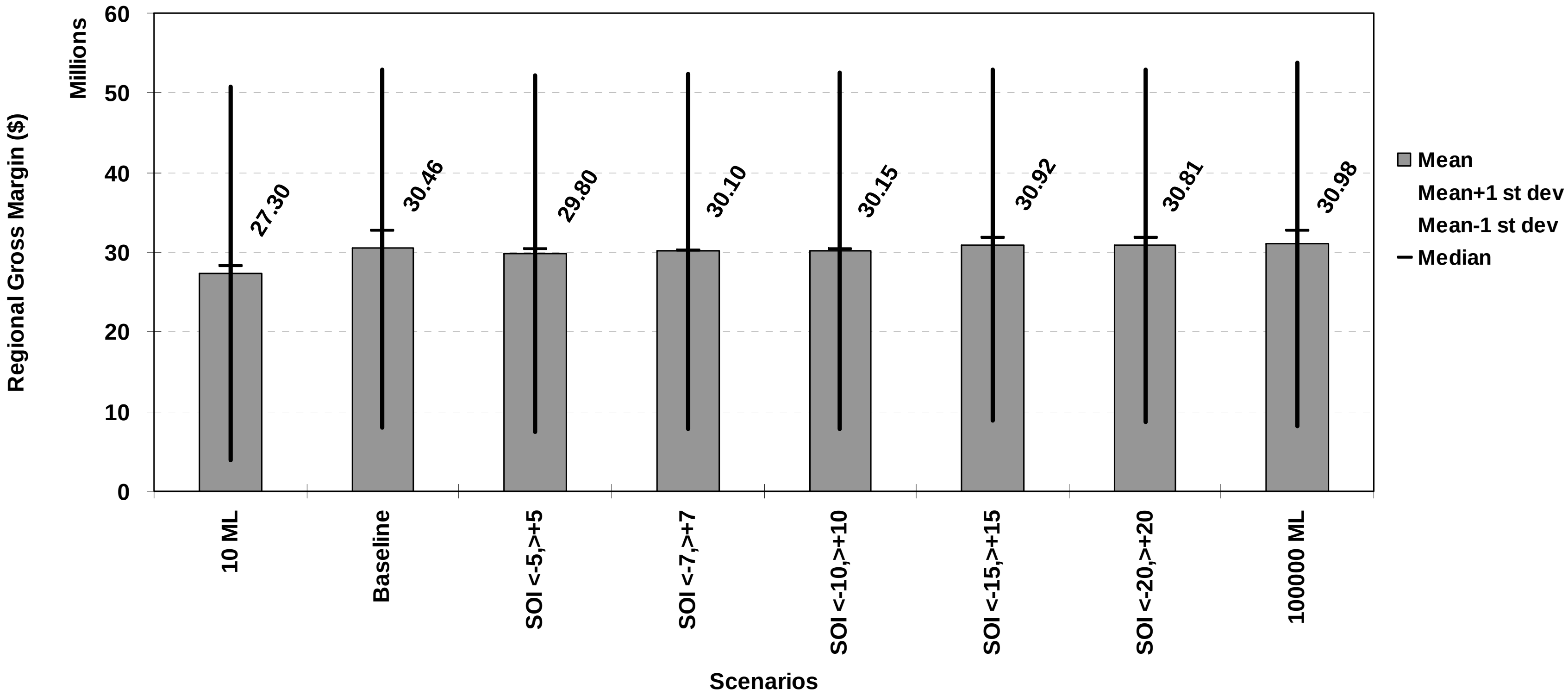


Figure 5.14: Economic results of simulation testing the outcome of regimes based on SOI values.

The results presented in Table L.3 and Table L.4 can also be interrogated based on the strength of the SOI signal. Table 5.3 identifies the number of years out of the 101-year scenario of each year type based on the bounds of the SOI values. It emphasises that as the SOI bounds become more restrictive, i.e. the SOI value bounds become larger as they move from the -5, +5 bound to the -20, +20 bound, the number of years when the off-allocation decision falls into the medium or the baseline bound (e.g. -5< x <5) increases. Therefore when the bounds for the scenario are SOI <-5,>+5, in 32 years the decision is to restrict off-allocation access to 10 ML, in 39 years off-allocation access is restricted to 50,000 ML and in 30 years off-allocation access is set at 100,000 ML. However when the SOI bound is more restrictive, as in the SOI <-15,>+15 and SOI <-20,>+20 scenarios, few years have the off-allocation restriction at either 10 ML or 100,000 ML. Perusal of Figure 5.14 demonstrates that as the SOI bound becomes more restrictive, the RGM results tend to increase, possibly due to a minimising of the number of scenarios when the streamflow outcome is not as forecast.

Table 5.3: Comparing the number of years by category for the SOI value scenarios

	SOI <-5,>+5	**SOI <-7,>+7**	**SOI <-10,>+10**	**SOI <-15,>+15**	**SOI <-20,>+20**
Dry	32	24	18	5	2
Medium (baseline)	39	56	70	91	97
Wet	30	21	13	5	2

A further important issue to be considered in analysing the results of a long-term simulation exercise, e.g. 100 years, is the variability of the results within the simulation period. The 10-year moving average of RGM (Figure 5.5) was used to consider this issue when analysing the results of the scenarios for developing the trade-off curve. A modified version of this approach is presented in Figure 5.15 where a comparison of the RGM for the May SOI phase and the baseline scenario for nineteen 10-year periods within the simulation period is shown. Each bar in Figure 5.15 represents the difference between the sum of RGM for each of the May SOI phase and the baseline scenarios for the 10-year period starting in 1894. That is:

$$\sum_{i=1}^{n=10} Baseline_i - \sum_{i=1}^{n=10} MaySOI_i .$$

Nineteen periods are obtained by staggering the starting point of each period by five years as in 1894, 1899, 1904, 1909 and so on until the last period starts at 1984. Therefore in Figure 5.15 the result for 1894 represents the difference for the period 1894 to 1903 and the result for 1899 represents the period 1899 to 1908. Two findings are immediately obvious. First, the number of 10-year periods where the RGM for the May SOI phase is greater than that of the baseline scenario is low at six from a total of 19 occurrences. Second, when the baseline scenario RGM is greater than the May SOI phase RGM it is generally by a considerable margin. In fact the total sum of the differences of the periods is in excess of -$96 million indicating the overall losses greatly outweigh the wins. Table 5.4 and Table 5.5 summarise the analysis depicted in Figure 5.15 for all of the SOI phase and SOI value scenarios. The pattern shown in Figure 5.15 is consistent across all SOI phase scenarios.

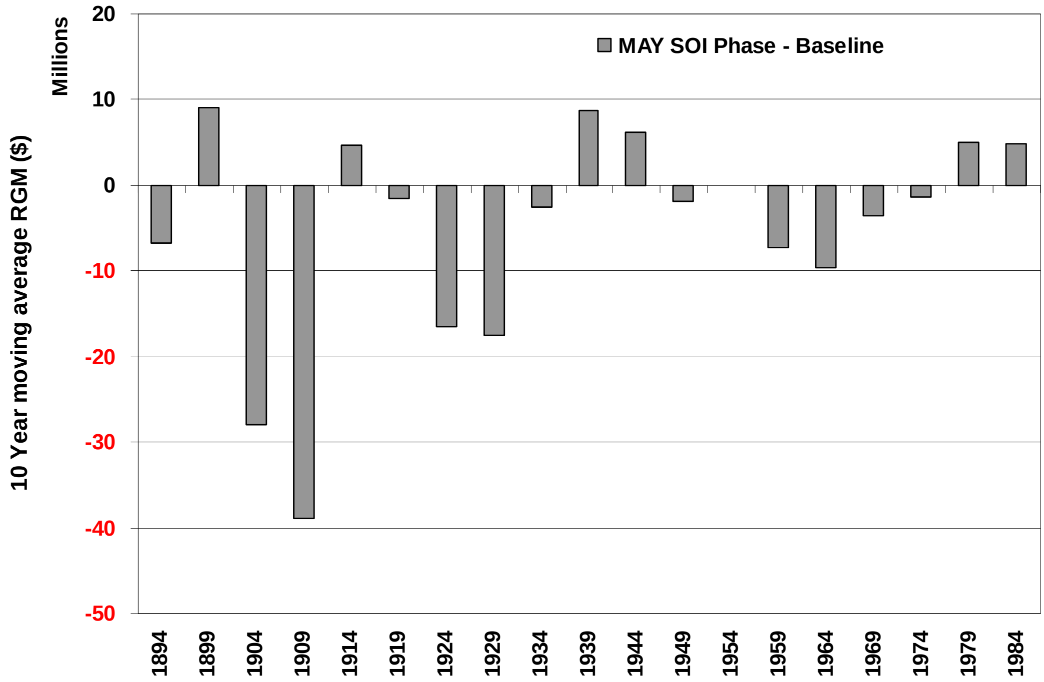


Figure 5.15 Assessing RGM results over shorter 10-year periods at 5 year starting increments.

While these results confirm the mean results reported in Figure 5.13 they also show that in some 10-year periods a scenario based on the SOI phases may give better outcomes from an economic perspective. The results of this analysis for the remainder of the scenarios reported in Table 5.4 and Table 5.5 all have similar patterns to that of Figure 5.15, excluding the two SOI value scenarios, SOI <-15,>+15 and SOI <-20,>+20 , that reported 100-year mean RGM above the Baseline scenario. These findings reiterate that a 100-year mean result should not be used without considering the implications for periods within the 100-year timeframe. Further research into this finding may provide information on why some 10-year periods give positive results and some negative, which in turn could lead to fine tuning the application of SCF information.

Table 5.4: Number of times the 10-year assessment period of each SOI phase scenario is above and below that for the 50,000 ML scenario

	May SOI	**June SOI**	**July SOI**	**August SOI**	**Sept SOI**	**Oct SOI**	**ENSO**
Number of times SOI phase RGM < 50000 ML scenario	13	12	10	15	10	11	11
Number of times SOI phase RGM > 50000 ML scenario	6	7	9	4	9	8	8

Table 5.5: Number of times the 10-year assessment period of each SOI value scenario is above and below that for the 50,000 ML scenario

	SOI <-5, >+5	**SOI <-7, >+7**	**SOI <-10, >+10**	**SOI <-15, >+15**	**SOI <-20, >+20**
Number of times SOI value RGM < 50000 ML scenario	15	12	12	7	6
Number of times SOI value RGM > 50000 ML scenario	4	7	7	12	12

In summary, of the 12 SCF scenarios assessed, the economic analyses show that only three scenarios reported mean RGM results greater than that of the baseline over the 100-year timeframe. When a set of 19 x 10-year assessment periods was considered, a number of the 10-year periods within each scenario reported results where the RGM from the seasonal climate forecast scenarios was greater than the baseline.

However, in only two scenarios, SOI <-15,>+15 and SOI <-20,>+20, did

$$\sum_{i=1}^{n=10} Baseline_i < \sum_{i=1}^{n=10} SCFscenario_i .$$

The data behind the results reported in Table 5.4 and Table 5.5 relating to these two SOI value scenarios and the ENSO scenario is reported in Table 5.6. The values in this table represent the difference between the 10-year total RGM Baseline scenario and the forecast scenario results. For example the in Table 5.6 the 1894 value of $7.91m is the difference between total RGM for the baseline and SOI <-15,>+15 scenarios for the 10-year period 1894 to 1903. The table therefore highlights that for these scenarios the size of the positive 10-year total differences strongly outweigh the negatives (note 0.00 outcomes represent small positive or negative outcomes but are not shown due to rounding). The table also shows that for the ENSO phase scenario which reported a mean RGM greater than baseline, although the total is positive, the number of negative 10-year assessment period results outweigh the positives. This is due to the two large positive outcomes for the 10-year periods 1918 and 1923.

This shorter period analysis indicates that there are periods within the 100-year simulation period for all scenarios examined where the SCF regional gross margin results are greater than that of the baseline scenario. It also reiterates the finding that in general the RGM results of the SCF scenarios are lower than that for the baseline.

Similarly to the process used in section 5.2, the next step in undertaking the trade-off analysis is to determine the impacts on riverine health from the SCF scenarios. This is also carried out using the approach where judgements of the impact on the riverine environment from any scenario is undertaken by assessing changes in hydrologic indicators where the closer the hydrologic indicators are to the natural state (one) the healthier the river is assumed to be.

Table 5.6: Differences between the 10-year total RGM of the baseline and selected SCF scenarios across the simulation period

	SOI <-15,>+15 – baseline $m	SOI <-20,>+20 - std- baseline $m	ENSO Phase - std- baseline $m
1894	7.91	7.91	-8.67
1899	18.46	18.46	-13.69
1904	8.74	8.74	-4.52
1909	6.60	6.60	31.23
1914	6.55	6.55	29.20
1919	-0.34	0.18	-2.16
1924	5.56	10.39	-3.42
1929	12.17	10.57	-0.30
1934	16.65	0.96	0.93
1939	15.29	5.47	3.99
1944	4.97	4.92	4.29
1949	0.00	0.00	0.50
1954	0.00	0.00	-0.09
1959	-0.01	0.00	-0.39
1964	0.00	0.00	-2.71
1969	0.00	-0.01	-4.27
1974	-3.06	-3.06	-5.02
1979	-3.51	-3.51	2.25
1984	-0.45	-0.45	4.67
Total	**95.51**	**73.70**	**31.83**

5.3.2. Impact on environmental indicators from using SOI based scenarios

The results of the SOI phase scenarios from the environmental perspective, reported in Figure 5.16 and Table L.5 (Appendix L), highlight the complexity of using a proxy for riverine health such as changes in hydrological indicators. The complexity is manifest in the variation of the response to the SOI scenarios where some of the indicators exhibit a beneficial change, i.e. moving closer to one, while others show a detrimental impact, i.e. moving closer to zero.

For instance, consider the June SOI phase scenario which in economic terms had the result closest to the baseline. The Mean AF indicator has not changed while the MAF has changed in a positive direction, albeit by only one percent. The PFlowDur indicators have all oscillated one, or at maximum two, percentage points either side of the baseline. Given these changes to environmental indicators it is difficult to ascertain whether there is a net positive or negative environmental outcome. If a single indicator such as MAF were used, then a number of scenarios would represent a positive outcome with the remainder representing no change. However use of another indicator, or combination of indicators, may lead to the conclusion that scenario outcomes represent either negative or positive outcomes. In other words, no clear conclusion can be drawn from these results.

The second simulation assessed the usefulness of SOI values to manage off-allocation access. Impacts on hydrologic indicators are shown in Figure 5.17 while tabular results are reported in Table L.6. In the same way as the SOI phase scenarios, the impacts on hydrologic indicators are very minor, in the realm of one to three percent[33]. Examination of the tabular results shows only a slight worsening of the PFlowDur 0.80 results ranging from one to three percent.

[33] Again noting the reduced scaling to permit visual comparison of results which has the tendency to make small changes appear larger.

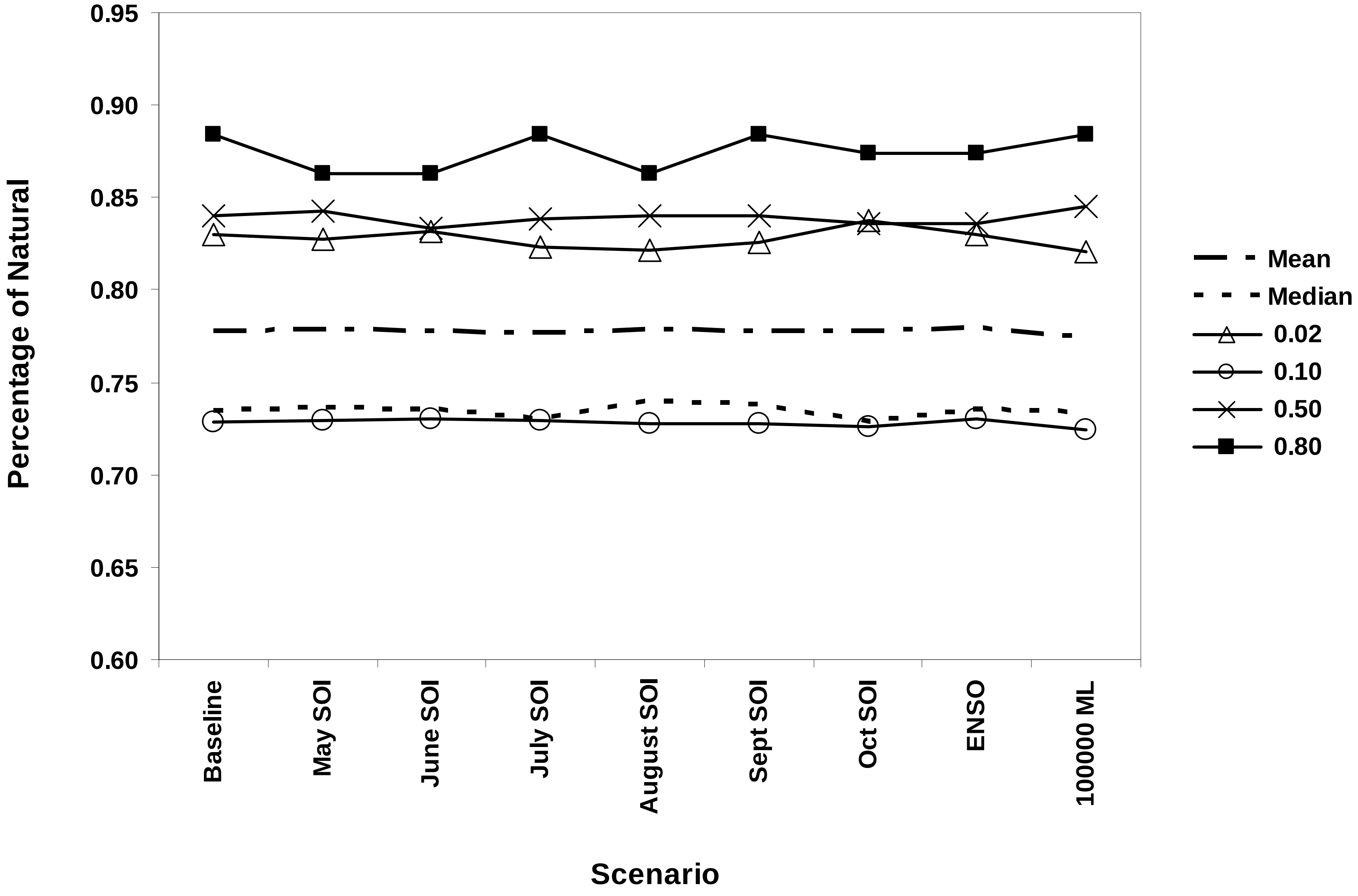


Figure 5.16: Assessing the impact on hydrological indicators of scenarios based on SOI phases.

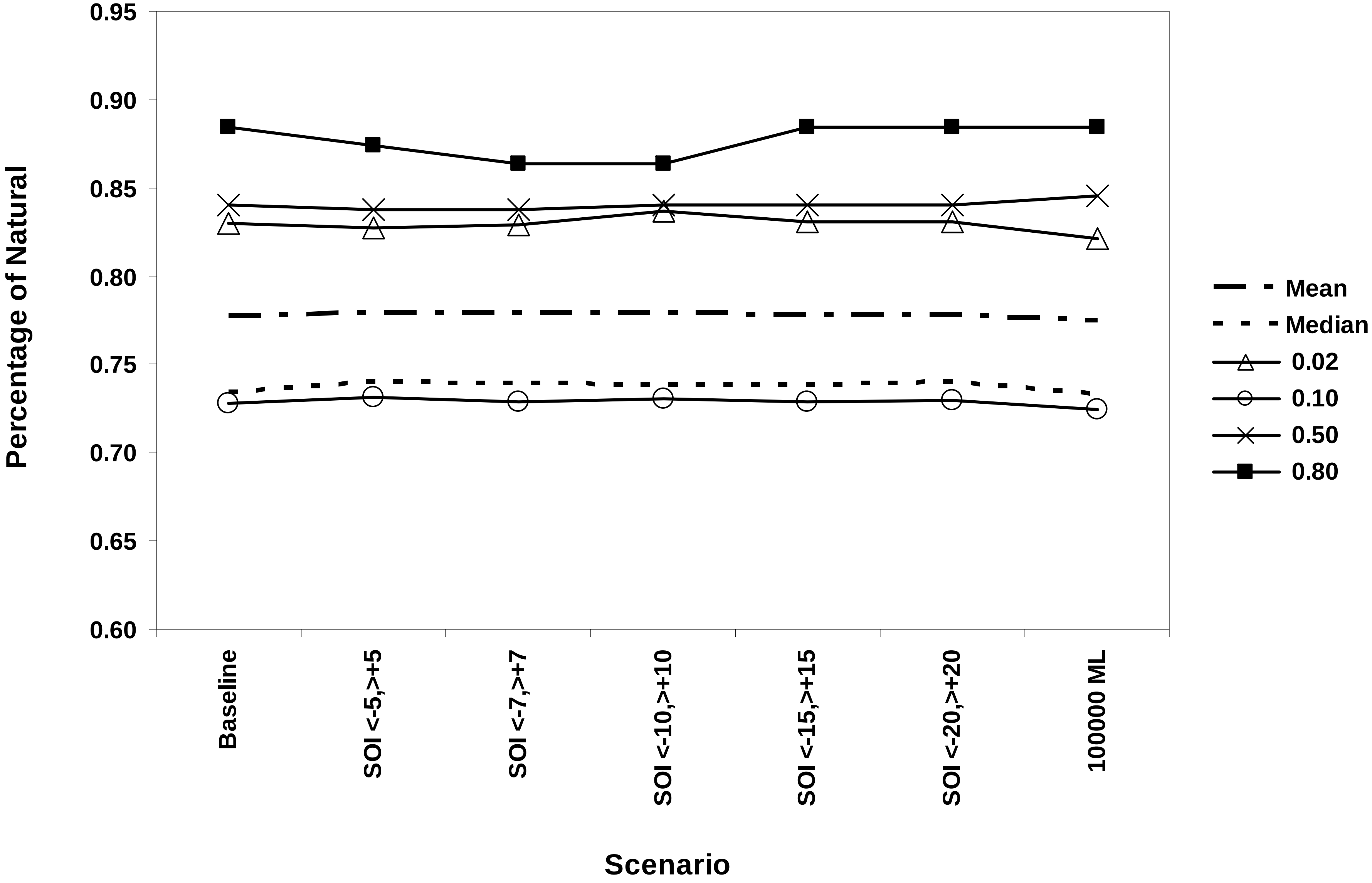


Figure 5.17: Assessing the impact on hydrological indicators of scenarios based on SOI values.

5.4. Sensitivity testing the production model

A key aim of this study has been the development of an appropriate methodology to assess the impact of using SCF in environmental flow management on both the regional output and the health of the riverine environment. This has been undertaken by the building of a model (called the eco-environmental threshold META-model, EETMM) combining a number of different model components covering river flow management, economic and environmental analysis, and finally SCF information. The EETMM is a series of models are used and run decoupled with data from one being used as input to the next (Figure 4.2).

The environmental component is undertaken using the IQQM and the Post Processor model to calculate hydrological indicator values and changes to the indicator values under the various scenarios. The Post Processor model (Bennett et al. 2002) is a "third party" component that is used in its entirety without amendment. While the adoption of the natural flows paradigm, and the analysis of hydrologic indicators, may be subject to scrutiny, this is aside from issues surrounding the EETMM itself. For this reason it is not feasible to undertake sensitivity analysis on these indicators as the uncertainty relates to the relationship between hydrologic indicators and riverine health. Over time as the level of knowledge about riverine health increases some of the indicators may be found to be less related to health and others more important. One rationale for using the methodology selected was to allow for different hydrologic indicators to be brought into the analysis as knowledge improves. Therefore there are no assumptions made during this study for this component that will impact on the results.

In contrast, the production model is the component that required the majority of separate inputs and includes a number of assumptions or rules that do impact the results. Therefore, the inputs and assumptions in this component represent the most risk to the validity of the analysis. The following sections consider the implications of assumptions made for two of the production model sub-components: soil moisture depletion; and, cotton yield.

5.4.1. Sensitivity of water use infrastructure settings

The analysis reported in Figure 5.2 shows increased output from increasing the availability of off-allocation. It also reveals that the large increase in potential access to off-allocation water resulted in a relatively small change in mean regional gross margin: 12 percent from the lowest off-allocation to highest off-allocation scenario. A key reason identified for this result was the constraint on the area able to be planted to irrigated crops.

The sensitivity of the results in Figure 5.2 and the key reason identified were assessed by increasing the area available for irrigated crops and running the same simulation. In this case the total irrigable area, on-farm storages and pump capacities were increased to 150 percent of the baseline scenario. The 50 percent increase was chosen for two reasons. First, to ensure the increase was significant enough to have an impact on study variables, and secondly, because in a survey of irrigators across the Border Rivers, Namoi, Gwydir and upper Condamine catchments Abawi et al. (2001) found that irrigators believed they could expand irrigated area by 47 percent. In this simulation all of these variables were increased in order to minimise the impact of bottlenecks in the system. The trend in regional gross margin remains similar to Figure 5.2 with the difference between the mean RGM 10 megalitre and 100,000 megalitre off-allocation access scenarios increasing to 24 percent (see Figure 5.18). The summary results of area planted and water applied provided in Table 5.7 show higher plant areas and water use, but again this does not reflect the proportional increase in area and water available. The fitted curve (R^2 = 0.97) again displays a pattern of diminishing marginal returns for mean RGM.

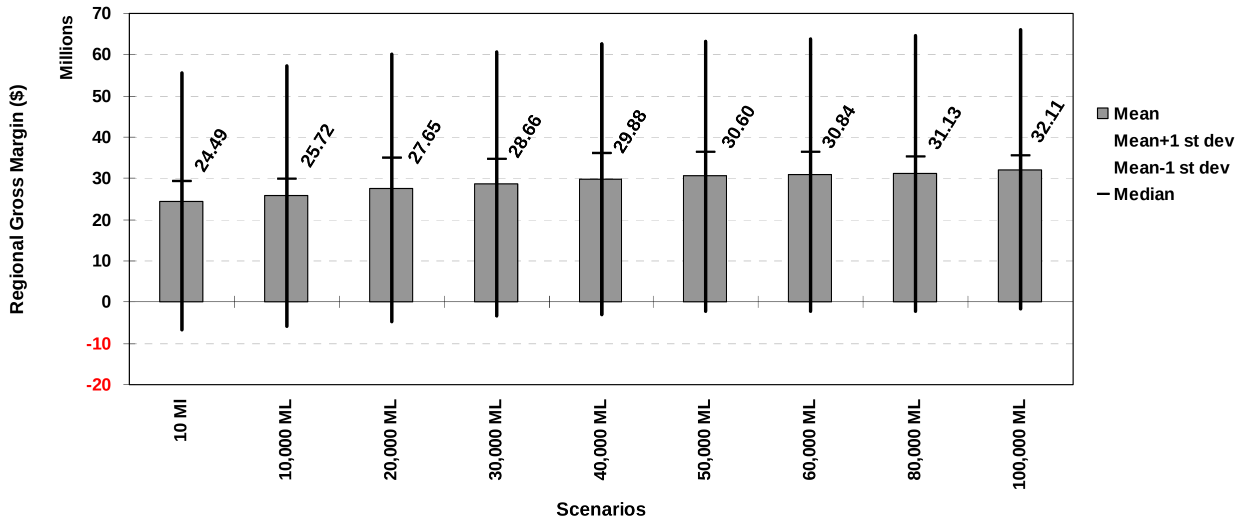


Figure 5.18: Assessing the regional gross margin for increasing access to off-allocation with total irrigable area, on-farm storages, pump capacities increased to 150 percent of the baseline scenario.

Table 5.7: Mean areas of irrigated cotton planted and water applied for the 150 percent scenarios (1894 to 1994)

	10 ML	10,000 ML	20 000 ML	30 000 ML	40,000 ML	50,000 ML	60,000 ML	80 000 ML	10,0000 ML
Plant area	28 038	28 522	29 239	29 667	30 220	30 486	30 704	30 955	31 311
Water applied	108 422	110 891	114 347	117 609	119 223	120 606	121 665	124 207	125 988

Again, the effect of bottlenecks is apparent. This means that although a particular decision rule may increase the potential availability of water, say from an off-allocation cap of 50,000 ML to 100,000ML per water year, there is a limit to how much water is able to be taken at any given time. For example, in a particular high flow event where there may be considerable volumes of off-allocation water available, the physical ability to take that water is constrained.

5.4.2. Impact of the soil moisture depletion rules in the EETMM model

As explained in the methodology, the IQQM soil moisture depletion (SMD) variable is used to determine a yield penalty if the soil moisture level drops below the critical threshold (180 mm). This penalty ranges from 40 to 100 percent of potential yield depending on the stage of the irrigation season (see Table 4.3). An example of the pattern of SMD in Figure 5.19 shows the impact of a lack of available water supply on SMD. In this example, due to a lack of water supply, the soil moisture level decreases (i.e. depletion gets higher). This will have implications for the subsequent yield of the cotton crop which in this case has been deprived of sufficient water to meet crop demands for most of the peak irrigation season. In the IQQM crop model there is no routine that identifies if the irrigated crop is dead and should not be irrigated if water becomes available later in the cropping season. In the absence of a rule or yield penalty accounting for this occurrence, a yield calculation based on the volume of irrigation water applied to the crop may overestimate yield in those years where water supply is limited for a period, but subsequently becomes available later in the cropping season. This is analogous for example to a case where five irrigations each of 1 ML are required to grow a crop to full potential. Therefore, the total irrigation supply of 5 ML translates to a yield of 10 bales per hectare from the OZCOT lookup table for 1910 (random year). If however the situation shown by Figure 5.19 occurred, but on a smaller scale, and the same amount of water was applied but with a gap during the peak demand period, then the yield from the lookup table would remain at 10 bales per hectare. But in this case the crop would potentially have reached wilting point and died, or at least suffered a yield decline.

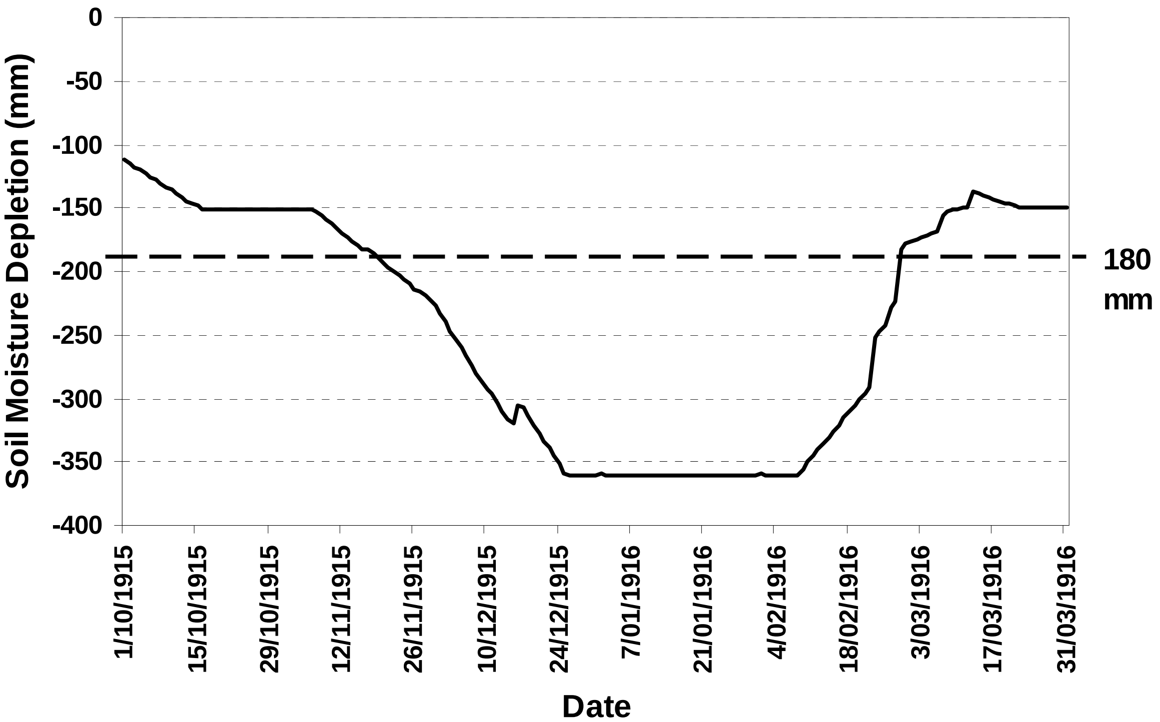


Figure 5.19: Tracking the level of soil moisture depletion past the wilting point of 180mm.

In the EETMM the SMD level is set at 180 mm such that the yield penalty is triggered in the applicable months when soil moisture depletion exceeds this level. The simple means of testing the impact of this rule is to compare two model runs where the first uses this rule and the second does not. The results of this analysis reported in Table 5.8, highlight two things. The first relates to the reasonably consistent percentage decrease in the average RGM from introducing the SMD rule. The second finding is that this rule did not change the ranking of the outcomes for the SOI phase and first three SOI value scenarios. For the ENSO and two SOI value scenarios that did show a positive value of information (SOI <-15,>+15 and SOI <-20,>+20) eliminating the SMD rule eliminated the VOI.

Table 5.8: Instituting the SMD rule results in a reasonably consistent increase in RGM in a range of 14-21 percent.

	Baseline	May SOI	June SOI	July SOI	August SOI	Sept SOI	Oct SOI	ENSO	100,000 ML
SMD set at 180mm	30.46	29.97	30.14	30.06	29.60	30.13	29.97	30.60	30.98
SMD rule excluded	35.04	34.80	34.83	34.92	34.63	34.88	34.75	34.84	35.48

	Baseline	SOI <-5,>+5	SOI <-7,>+7	SOI <-10,>+10	SOI <-15,>+15	SOI <-20,>+20	100,000 ML
SMD set at 180mm	30.46	29.80	30.10	30.15	30.92	30.81	30.98
SMD rule excluded	35.04	34.53	34.54	34.55	34.97	34.81	35.48

The second aspect of RGM results that needs to be considered is the impact on variability. This is achieved through a similar comparison, however this time the comparison variable is the coefficient of variation. The coefficient of variation (CV) is a ratio of the standard deviation over the mean and is a relative measure of variation. The impact on CV reported in Table 5.9 also exhibits a constant change. In this case, the consistently lower CV indicates that the exclusion of the SMD rule decreases the variability of the RGM results across the board. This outcome is explained by the removal of the penalties which can range up to 100 percent of the yield in any given year.

Table 5.9: Assessing the change in variability from the use of the SMD rule

	Baseline	May SOI	June SOI	July SOI	August SOI	Sept SOI	Oct SOI	ENSO	100,000 ML
SMD set at 180mm	0.74	0.76	0.74	0.76	0.76	0.77	0.76	0.71	0.74
SMD rule excluded	0.53	0.53	0.53	0.53	0.53	0.53	0.53	0.52	0.53

	Baseline	SOI <-5,>+5	SOI <-7,>+7	SOI <-10,>+10	SOI <-15,>+15	SOI <-20,>+20	100,000 ML
SMD set at 180mm	0.74	0.75	0.74	0.74	0.71	0.72	0.74
SMD rule excluded	0.53	0.53	0.53	0.53	0.52	0.53	0.53

5.4.3. Impact of the use of OZCOT yield data in the EET meta-model

The second component of the EETMM where study assumptions may influence results is that of cotton yield. In chapter 4 (methodology) two methods of using the OZCOT data to calculate yield for the EETMM were introduced. The method chosen was to use the yield matrix as a lookup table (reported in Table D.1, Appendix D) for determining cotton crop yield for each year of the simulation based on the water supply per hectare calculated in the IQQM simulation (referred to below as 'predicted yield'). The alternative method was to calculate the mean yield per hectare over the simulation for each allocation level (see Table 4.5 for information on how this mean yield is calculated). This approach would smooth the yield curve to a shape closer to that of the classic production function. An indication of the difference in yields between the two methods was presented in Figure 4.13.

Likewise with the SMD, the way to assess the difference between the two methods is to compare results from simulations where each method is used. Results of this analysis (Table 5.10) also show a reasonably consistent shift between scenarios. The results of the mean yield per hectare over the simulation for each allocation level are consistently lower than that of the predicted yield method. This translates to lower RGMs and is due to the tendency for the mean yield method to remove some of the variability in yield, particularly the yield spikes as seen in Figure 4.13.

Table 5.10: Assessing the impact on RGM of different methods of calculating cotton yields

	Baseline	May SOI	June SOI	July SOI	August SOI	Sept SOI	Oct SOI	ENSO	100,000 ML
Predicted yield	30.46	29.97	30.14	30.06	29.60	30.13	29.97	30.60	30.98
Mean yield	26.33	25.95	25.95	25.93	25.64	26.03	25.98	26.63	26.70

	Baseline	SOI <-5,>+5	SOI <-7,>+7	SOI <-10,>+10	SOI <-15,>+15	SOI <-20,>+20	100,000 ML
Predicted yield	30.46	29.80	30.10	30.15	30.92	30.81	30.98
Mean yield	26.33	25.83	26.15	26.20	26.83	26.77	26.70

The evaluation of the impact of the two rules on variability through the CV is shown in Table 5.11. In this case the trend is similar to that of the average RGM results: the introduction of the mean yield method reduces the variability of the RGM results in a reasonably consistent manner. The decrease in variability however is not as large as that for the SMD rule. This outcome is also due to the tendency of the mean yield method to remove the variability in yield, particularly the yield spikes as seen in Figure 4.13.

Table 5.11: Assessing the impact on the variability of average RGM outcomes from the two yield calculation methods

	Baseline	May SOI	June SOI	July SOI	August SOI	Sept SOI	Oct SOI	ENSO	100,000 ML
Predicted yield	0.74	0.76	0.74	0.76	0.76	0.77	0.76	0.71	0.74
Mean yield	0.69	0.70	0.69	0.71	0.71	0.72	0.71	0.65	0.69

	Baseline	SOI <-5,>+5	SOI <-7,>+7	SOI <-10,>+10	SOI <-15,>+15	SOI <-20,>+20	100,000 ML
Predicted yield	0.74	0.75	0.74	0.74	0.71	0.72	0.74
Mean yield	0.69	0.70	0.68	0.68	0.66	0.66	0.69

5.4.4. The plant area decision

In the version of the IQQM used in this study an expectation of off-allocation volume for the upcoming season is not included in the planting decision. Consequently in a year where the off-allocation cap is raised (which occurs at the time of planting in October) there is no adjustment to the plant area decision of irrigators. The implications for this study are two fold. First, in a year where the off-allocation cap is set at or towards 100,000 ML it is likely that a proportion of irrigators would plant more area to irrigated crops in the expectation of increased water supply through off-allocation. Alternatively, the action might be to decrease the area planted when off-allocation access is denied. The IQQM does not take this activity into account and thus is a deviation from reality. But in the absence of information on the volume of expectations of irrigators it is a necessary approach.

The impact of this is at least partially offset by the way in which off-allocation is treated in the model. In general a substitution effect occurs when there is a flow in the river and off-allocation availability is announced. When this occurs irrigators (and the model) will opt to fill on-farm storages with off-allocation water instead of ordering water from their licensed allocation (on-allocation). This is because by using off-allocation water when available, the on-allocation water remains available for use at other times when irrigation water supply is scarce. Thus using off-allocation water in preference to on-allocation is also a risk mitigation exercise. Table 5.12 confirms this outcome by showing that the relative proportion of off-allocation diverted versus on-allocation diverted changes as access to off-allocation water is increased. In the baseline scenario, of the total water diverted from the river for irrigation (approximately 5.37 million ML over the 101 year scenario) 24 percent, or 1.17 million ML was made up of off-allocation and 76 percent from on-allocation. This compares to 30 percent (1.72 million ML) and 70 percent (3.85 million ML) for the 100,000 ML scenario where total diversion was about 5.582 million ML. The substitution effect, though relatively small in percentage terms, occurs leading to higher off-allocation use as access to off-allocation water increases.

Table 5.12: The relative proportions of off and on-allocation water use changes in response to increasing caps on off-allocation

Off-allocation access scenarios	Proportion off-allocation	Proportion on-allocation
Baseline	24 percent	76 percent
60,000 ML	27 percent	73 percent
100,000 ML	30 percent	70 percent

A second factor mitigating the impact of the plant area decision (related to this substitution effect) stems from the fact that on-allocation water not used in a water year where the off-allocation cap was set at 100,000 ML is subsequently available (subject to some restrictions) in the following water year. Availability of on-allocation is through carry-over rules (Podger 2006) such that a proportion of the on-allocation in a particular year, if not used, is able to be carried into the next water year so that starting water supply in that year is higher. Where the starting volume of water in subsequent years is higher, larger irrigated plant areas will result.

5.4.5. The implications of using mean regional gross margin as the economic indicator

Reporting of the regional gross margin (RGM) results centered on the changes amongst scenarios of the total gross margin for each water year averaged across the simulation. From an economic perspective, the higher the RGM the more desired is the outcome *ceteris paribus*. RGM however has a number of limitations that, while not changing its status as the most appropriate variable to use in this study, should be noted in considering the results and subsequent conclusions.

In the first instance while change to regional gross margin is a general measure of the profitability of agriculture in a region, it is a short-term measure as it does not take into account the ability of irrigators to adjust to changes in water supply regime (NSW Agriculture 2001). However, an increase or decrease in regional gross margin provides an indication of whether the changes to producer surplus from a change in water management rules are likely to be positive or negative.

Second there is the issue of variability of RGM. On its own RGM does not measure variability of simulation results, which can have a major impact on the acceptance by stakeholders depending on their attitude to risk (where risk is defined as variability of returns). In this study the level of variability is highlighted in the summary tables (Table L.1, Table L.3 and Table L.4, Appendix L), as well as Figure 5.5 and Figure 5.6 for the first scenario and Figure 5.13 and Figure 5.14 for the forecast scenarios. Results are shown to be consistently variable across the scenarios tested. This minimises the impact of variability on the level of acceptance by stakeholders as the SCF scenarios do not markedly increase variability of returns over scenarios where SCF is not used.

A more interesting finding, but one that is unlikely to change the conclusions, is from the results of the 10-year moving average of RGM. The results presented in Table

5.4 and Table 5.5 show that in some 10-year periods a scenario based on the SCF that was not viable over the 100-year timeframe may give better outcomes from an economic perspective. For example, in Table 5.4 it is reported that for the September SOI phase scenario, nine of the nineteen 10-year moving average periods resulted in higher RGM than the baseline scenario for the same periods. This implies that it may be worthwhile to undertake study into why these periods gave more favorable results with a view to improving the application of forecast knowledge. Knowledge of what years not to take a forecast into account may improve overall results as one of the key issues is identifying the 10-year periods where losses are very large such as 1904 to 1913 and 1909 to 1918 in Figure 5.15. Furthering the research in this direction could provide an opportunity to promote the use of SCF information in water resource management decision-making.

Two final related issues to note in regard to the process of assessing regional gross margin over a period of 10 or 100 years is that of catastrophic events. That is this approach does not consider the first risk to individual farms nor second the problem of a single or small span of years with catastrophic results which may cause business failures of individual farms or groups of farms. While it is beyond the scope of this study to address these issues it is acknowledged that adoption of SCF information in water resource management decision-making needs to be evaluated for its impacts at the individual farm level. This serves two purposes, first it provides an indication of whether impacts at that level are acceptable and second it informs questions of distributional impacts.

5.5. Joining the economic and environmental results to assess the Pareto outcomes

In this study, evaluating the efficacy of seasonal climate forecasting on environmental flow management is being undertaken by examining both the economic and environmental outcomes from flow regimes based on SCF information. Presentation of the results thus far has been in the silo form considering economic and the environmental results separately. However, the discussion on Pareto outcomes through the matrix of outcomes presented in Table 5.2 requires that the results from the two perspectives are brought together.

The summary results of both the economic and environmental analyses are reported in Table 5.13 and highlight that of all the scenarios assessed the two SOI value scenarios SOI <-15,>+15 and SOI <-20,>+20 as well as the ENSO scenario are the only ones to represent potential Pareto improvements. As discussed above, these findings are only marginal with the result of improved economic benefits being increases in average regional gross margin over the baseline of 0.7, 0.35 and 0.4 percent respectively.

Table 5.13: Combining economic and environmental results to identify Pareto outcomes

Scenarios	Economic outcome - *Change in regional gross margin*	Environmental outcome - *Change in hydrologic indicators*	Pareto outcome
May SOI	↓ Economic benefit	No change	-
June SOI	↓ Economic benefit	No change	-
July SOI	↓ Economic benefit	No change	-
August SOI	↓ Economic benefit	No change	-
Sept SOI	↓ Economic benefit	No change	-
Oct SOI	↓ Economic benefit	No change	-
ENSO	↑ Economic benefit	No change	Potential Pareto improvement
SOI <-5,>+5	↓ Economic benefit	No change	-
SOI <-7,>+7	↓ Economic benefit	No change	-
SOI <-10,>+10	↓ Economic benefit	No change	-
SOI <-15,>+15	↑ Economic benefit	No change	Potential Pareto improvement
SOI <-20,>+20	↑ Economic benefit	No change	Potential Pareto improvement

5.6. Conclusion

This chapter presented the modelled results of the scenarios identified in Table 4.8 for both the economic and environmental analyses. The results of the first simulation testing the impacts from gradually increasing off-allocation were in line with expectations although proportionally small, such that gradually increasing the availability of water gradually increased economic returns. The results for the environmental indicators was not as clear cut with some hydrologic indicators of environmental outcomes increasing (PFlowDur 0.5 and 0.8) and others decreasing. A key finding for both economic and environmental results was the small change in values as off-allocation access increased greatly from 10 megalitres to 100,000 megalitres for each water year of the scenario period 1894 to 1994. This finding together with the finding that for a considerable number of years when water supply was increased (i.e. made available via an increased cap and higher streamflows permitting increased off-allocation use) it is not used when available, indicates that other factors, e.g. bottlenecks in the system, are at play as well. This suggests that the potential benefit from use of SCF information may be small as the band in RGM and environmental indicators between the smallest and largest off-allocation caps are small or unclear.

Results of the comparison between the SCF and baseline scenarios indicate that only three scenarios of the 12 scenarios tested reported RGM results greater than that of the baseline over the 100-year timeframe. These were the ENSO phase, SOI <-15,>+15 and SOI <-20,>+20 scenarios. Again the results for the environmental analysis were mixed with some hydrological indicators increasing while others decreased. As with the first simulation, the absolute change in hydrological indicator values was very small. Using the Jones, Hillman et al. (2003) none of the scenarios negatively impact on river health as for all the scenarios assessed the hydrological indicators remain above two-thirds of natural flow.

This chapter also reported results from testing the key assumptions used in the economic analysis. This includes the impact on results of the soil moisture depletion rules and methods for using OZCOT yield data. Both assumptions were found to have considerable impact on results but remain used within the EETMM. In the case of SMD the assumption used to ensure yields are not over-estimated, by including a yield penalty when crop water supply does not meet crop water requirements, has a material impact on results. This impact is such that when the yield penalty is removed the results indicate that no SCF scenarios have an RGM greater than the baseline scenario. The OZCOT yield calculation used brings more of the inherent variability in yield outcomes rather than using a smoother average yield which is not reflective of field conditions.

The final section of the chapter brought the two analyses, economic and environmental, into one table to determine the Pareto outcomes indicated by the results. This showed that of the 12 SCF scenarios tested only 3 represent potential Pareto improvements. The next chapter takes this result and explores the implications for the study hypotheses in the trade-off analysis framework.

6. Reviewing the potential value of seasonal climate forecasting

6.1. Introduction

The intention of the CoAG and NWI water policy directions is to encourage or drive the consumptive use of water to a more sustainable level, maximising the economic value of water while bearing in mind the environmental consequences of the use of this water. The reform agenda has an objective to alleviate some of the environmental damage caused in the past and at least ensure damage does not worsen. This policy direction provides the basis for the first aim of this study to:

> Develop the trade-off methodology and modelling framework for examining the impact of using three ENSO based SCF tools in environmental flow management on both the regional economy and the health of the riverine environment of the Border Rivers Catchment.

The inherent trade-off between consumptive use and in-stream use of water is an overarching concern (MDBMC 1995; Harris 2006; Department of the Environment 2008a; NLWRA (National Land & Water Resources Audit) 2001; Schofield et al. 2003; Quiggin 2001). SCF information represents a knowledge advancement that is not currently utilised in water management decision-making to any serious degree. While SCF information shows potential it's use in attaining positive environmental outcomes whilst not producing negative economic impacts has not been tested to date.

The view is that in a large proportion of catchments where high levels of water use occurs, the point has been reached where increasing the level of consumptive use of water will result in unacceptable levels of damage to the health of riverine environment. It is also socially, economically and politically unpalatable for consumptive water use to be heavily reduced to produce benefits to the riverine environment. This is however the result of a range of programs at the state and Australian government level (Department of Environment and Conservation NSW 2006; Social and Economic Reference Panel for the Murray–Darling Basin Commission 2008; Department of the Environment 2008b; Australian Government 2007). These programs represent one set of actions amongst a range of alternative actions that may be undertaken to begin to address the constraints and challenges facing the water management regimes in Australia elucidated by Schofield et al. (2003, p. 8):

"1. Improving water allocation and water-trading arrangements,
2. Assessing the costs and benefits of increasing allocations to the environment,
3. Understanding and managing the impacts of reductions in allocations for consumptive water use,
4. Developing cost-effective ways to enhance environmental flows,
5. Improving administrative arrangements for the management of water allocations".

This research has focused on one potential action that links in with components of challenges 1 and 2. That is positive findings mean the adoption of SCF information into water resource management decision-making may be an aid in partly meeting these challenges. That is the adoption of seasonal climate forecast information into water resource management decision-making may identify periods when the taking of water for consumptive purposes would be expected to have a minimal negative impact or positive impact on environmental outcomes. Positive outcomes may be possible, for instance where the use of SCF results in changes to the timing of access to water, e.g. by identifying upcoming periods of high flow, with the result that less drought like flows occur. This is achieved by examining the economic and environmental trade-offs from changing water management rules and could equate to an improvement in how water is allocated. The assessment of this potential action is the second objective of the study and encompasses the study hypotheses:

- *Hypothesis 1: The use of any of the three seasonal climate forecasting tools based on the El Niño-Southern Oscillation phenomenon to manage off-allocation water access by irrigators will increase the regional economic output of irrigated agriculture and/or produce conditions that will lead to improved riverine environmental health in the case study catchment.*

- *Hypothesis 2: An improvement in forecasting accuracy will increase the value of crop production and environmental flows to levels above those prevailing under the use of the identified forecast tools.*

These hypotheses were tested within a framework that also considers the impact of a number of modeling components on results.

Much of the work undertaken in developing the literature review and methodology was to consider the range of analytical frameworks for testing the hypotheses within the boundaries set by the policy direction in order to develop an appropriate analytical framework. In this context "appropriate" is taken to mean a framework that is able to assess the apparent trade-off between consumptive and in-stream use in order to make a judgement of efficacy of the use of SCF information in line with the policy direction.

The framework developed uses a case study approach comparing outcomes from the use of seasonal forecast technology in water management decision-making with outcomes where no seasonal forecasting information is used. Following the Pareto principle, an assessment of both economic and environmental outcomes was undertaken. Identifying a positive change in at least one of the outcomes, without diminution of the other, leads to a judgement of the efficacy of SCF information. It should be noted however, that the results are specific to both the case study examined and the location. Specificity of location is related to the variable nature, both spatial and temporal, of the forecast "skill", where the relationship between ENSO based forecasts and streamflow is not present in all locations nor in all years in locations where the relationship is strong (Wernstedt et al. 2002).

In the following section the implications of the results for hypothesis one are considered by identifying the trade-offs between consumptive and in-stream use. Then hypothesis two is examined using the concept of the perfect forecast. The chapter concludes with an examination of the results and the potential for SCF information to be adopted into water resource management decision-making.

6.2. Testing and discussing the implications of hypothesis 1

6.2.1. Identifying the general consumptive - in-stream trade-offs

The first part of the results section presented the change in regional gross margin (RGM) and a number of hydrologic indicators separately before combining them to develop a trade-off curve (Figure 5.11). These results showed how each variable responded to incremental changes in the availability of off-allocation water supply from zero to 100,000 ML per year. The trade-off curve introduced in chapter 2 represents a framework that can be used to visually represent threshold value analysis and examine how these results combine to allow an overall decision on efficacy of the information to be made.

Recapping, from a given set of inputs the trade-off curve illustrates the maximum amounts of two outcomes, in this case economic and environmental outcomes that can be produced from a limited quantity of an input (water) given a particular set of technology and resources. In doing this the trade-off curve highlights the trade-off between the two outcomes from changing the relative use of the input water between outputs and is shown graphically in Figure 6.1 theoretically and Figure 5.11 analytically. In Figure 6.1 the Trade-Off Curve$_{AB}$ (TOC$_{AB}$) represents the maximum level of production of the two outcomes – environmental and economic, possible from varying the usage levels of the input – water. The levels of water input corresponding to AB can be loosely thought of as allocation of water from one outcome to another. Behind each point on the curve a certain quantity of water is allocated to each outcome. In order to move from point B to A more water is allocated to the environment leaving less for productive purposes. The outcomes from this change in allocation of resources is measured by calculating the marginal rate of transformation (MRT) which is defined as the tangential slope of the trade-off curve (Doll and Orazem 1984; Parkin 1990). The MRT which increases as we move from left to right, is calculated by the ratio of the change in environmental use benefits (as measured by the hydrologic indicator), Δa^1 to the change in consumptive use benefits (regional gross margin) Δb^1. That is

$$MRT = \frac{\Delta a^1}{\Delta b^1}.$$

It measures the trade-off, which in this example appears to be relatively even, i.e. $\Delta a^1 \approx \Delta b^1$. However this trade-off changes depending on the slope of the curve at any particular point.

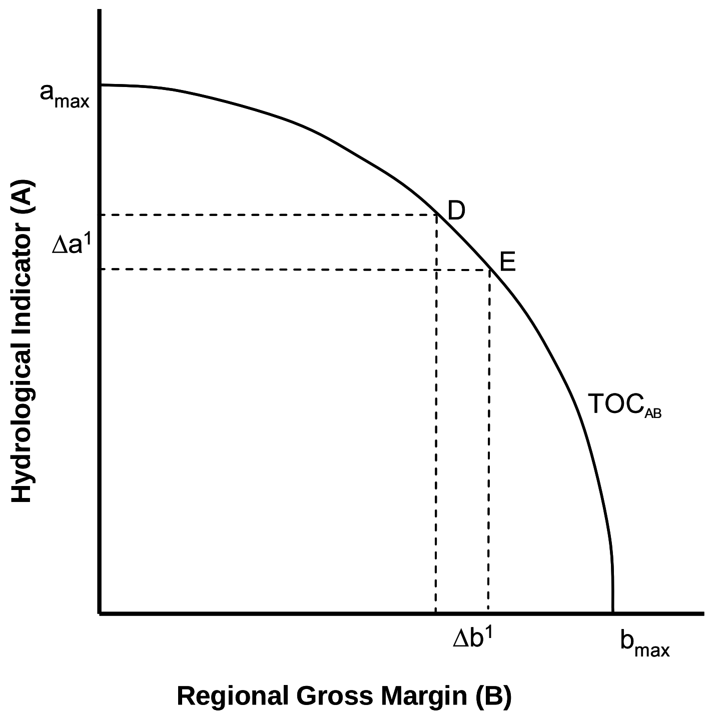


Figure 6.1: Simple curve highlighting trade-off between outcomes.

Figure 6.2 provides an example of the change in MRT along the slope of the curve using the typical convex trade-off curve. A change in allocation that results in an adjustment from B to A produces a trade-off measured by Δa and Δb. In this example Δa < Δb implying that the trade-off has increased for consumptive use benefits as more of it has to be given up in order to obtain a much smaller increase in environmental use benefits than in Figure 6.1. Had the allocation change resulted in a move from B\` to A\` the trade-off would have been reversed as $\Delta a^1 > \Delta b^1$.

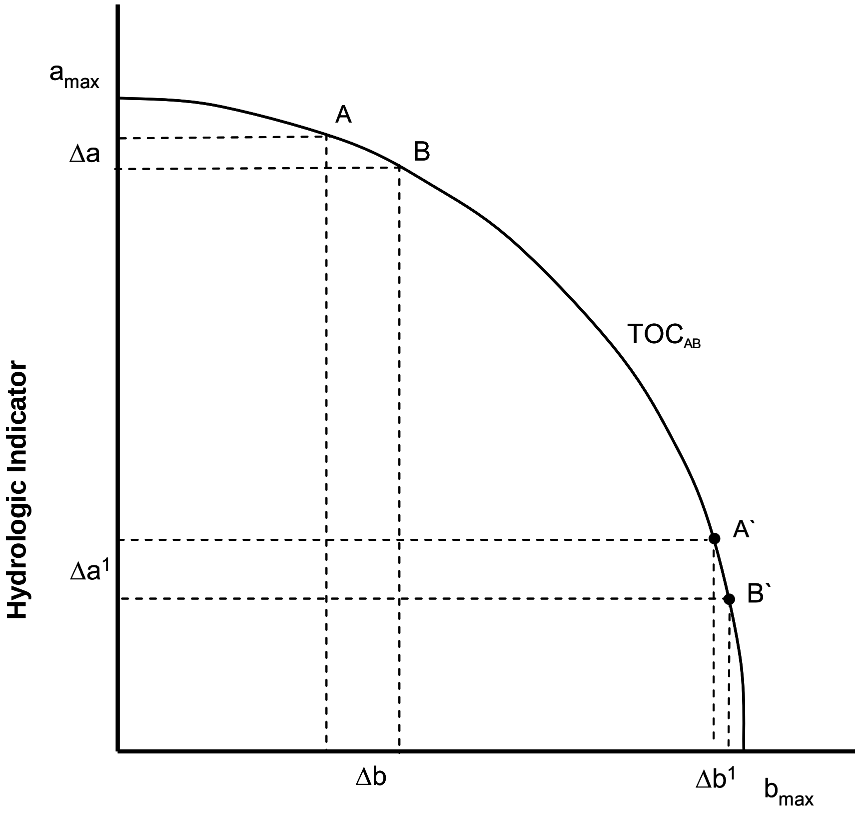


Figure 6.2: Highlighting the change in trade-off as the allocation levels move around the frontier.

In chapter 2 a number of different relationships, complementary, supplementary and joint, related to the characteristics of the PPF curve were introduced (Doll and Orazem 1984). Given that changes to the variables that comprise the trade-off curve in this study remain driven by the physical relationship with changing water supply these relationships are assumed to extend to the trade-off analysis curves developed in this study. The supplementary relationship, where the amount of one outcome can be increased without decreasing the production of another outcome, was identified as one of particular importance as this is in effect what is being sought through the use of SCF information.

The supplementary relationship is one that may or may not exist in scientific terms but in any event may exist in practical terms. For instance, Figure 6.3 demonstrates a standard convex trade-off curve where the point x represents the level of allocation that society deems is the minimum allocation to consumptive use benefits. In other words, at this threshold the level of environmental use benefits identified by a is the maximum level of outcome society is prepared to accept because higher levels may result in consumptive use outcomes that are too low (i.e. to the left of b). The existence of this threshold is supported by the elicitation of environmental performance objectives and hydrologic indicator values in water resource plans and water sharing plans (DNRM 2003; NSW Department of Water and Energy 2007b). Therefore, the line ax while not a true supplementary relationship is one for practical purposes. This notion implies that the area of concern for the trade-off in this study lies to the right of the arbitrary point x along the curve $a_{max}b_{max}$ but perhaps to the left of point y. Point y is the conceptual allocation level where society is unwilling to allocate more water to the production of increased regional gross margin because of the size of the negative effect on hydrologic indicators and therefore by extension riverine health.

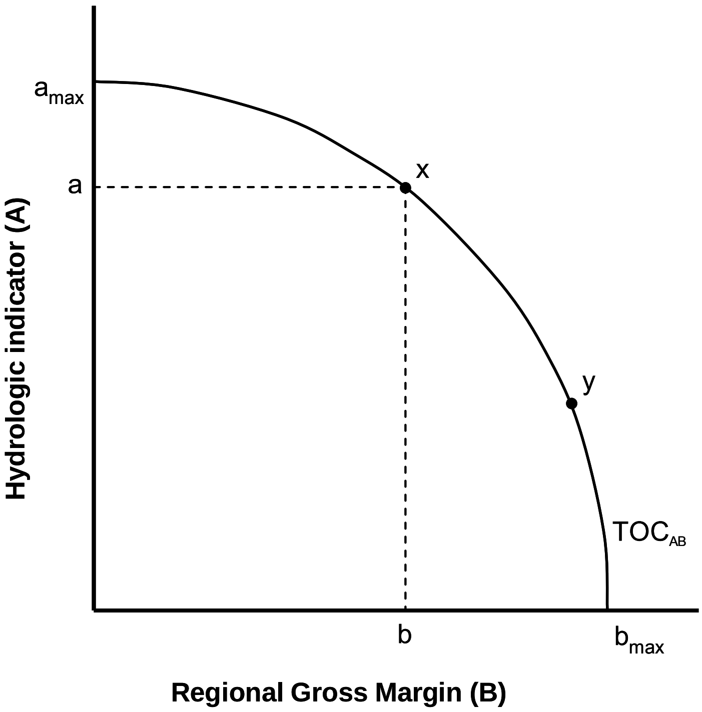


Figure 6.3: Identifying a supplementary relationship below which environmental outcomes are deemed unaccepatble.

To the right of point x lies a large subset of combinations of output and relationships that may display complementary, supplementary or competitive relationships depending on the shape of the frontier between x and b_{max}. The slope of the frontier has strong implications for policy choices as the policy adopted is likely to result in a movement along the curve (or even within the curve), which in turn will determine whether the intent of policy direction is achieved. This concept is demonstrated in Figure 6.4 where the value of the MRT at point x is smaller than point y indicating that the trade-off at y is greater than at x. The implication from this depends on the location of the baseline or starting position. If the starting point is at x then a marginal move to allocate more water to consumptive uses will result in a relatively even trade-off. A small move to the right from x that results in Δa for in-stream use benefits, and Δb for consumptive use benefits, represents an even trade-off as proportionally the gains from one equal the loss to the other, i.e. $\Delta a = \Delta b$. However if the starting point is at y where the curve is much steeper (higher MRT), then the same marginal movement of water to consumptive uses will have a higher trade-off as relatively higher decreases in environmental use benefits result from smaller increases in consumptive use benefits. So where the loss of in-stream use benefits is the same as that for the move from x, $\Delta a = \Delta a^1$, this represents a much larger trade-off as the increase in consumptive use benefits is proportionally small. That is $\Delta a^1 < \Delta b^1$. Thus the location of the starting point and the slope of the trade-off around that point is valuable information.

The concept introduced in Figure 6.4 can be extended to a discussion on threshold value analysis (TVA) where the threshold is the point (when moving from right to left on the trade-off curve) at which the decrease in hydrologic indicator levels exceed the consumptive benefits gained. Therefore, above this threshold the proposed policy change is not acceptable. This concept can be explored using the two-thirds "rule" introduced by Jones et al. (2003): if the key hydrological attributes are above two-thirds of natural then there is a high probability that the river is in a healthy working state. This rule assumes that, moving from left to right, once the key hydrological indicators reach a certain level there is a marked decrease in the environmental health of the riverine ecosystem. If point x (Figure 6.5) represents the point at which the key hydrological indicators are at two-thirds this implies that there is a significant change in the MRT and a subsequent increase in the slope of the curve at this point. The slope beyond point x is much steeper revealing the higher level of trade-off of in-stream benefits for each unit of consumptive benefit.

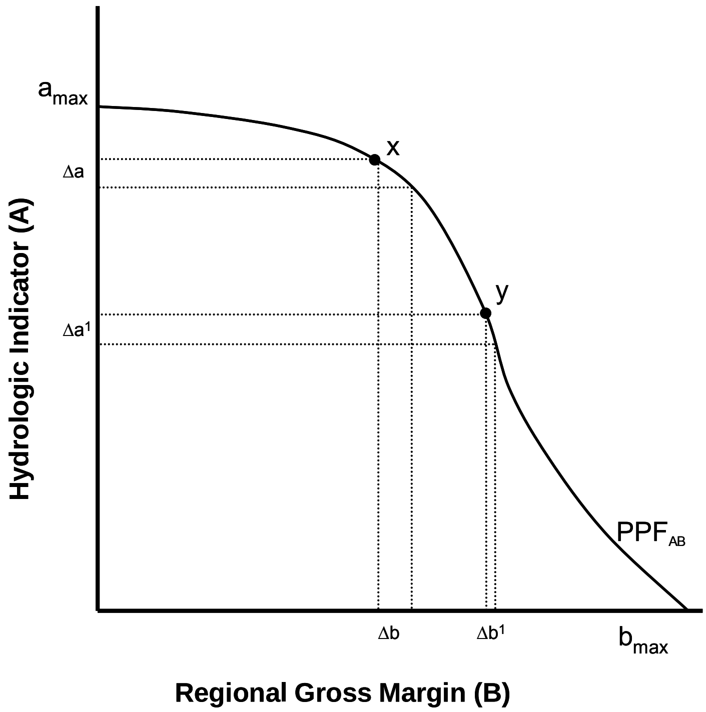


Figure 6.4: Changes in the values of MRS as the slope of the trade-off curve changes.

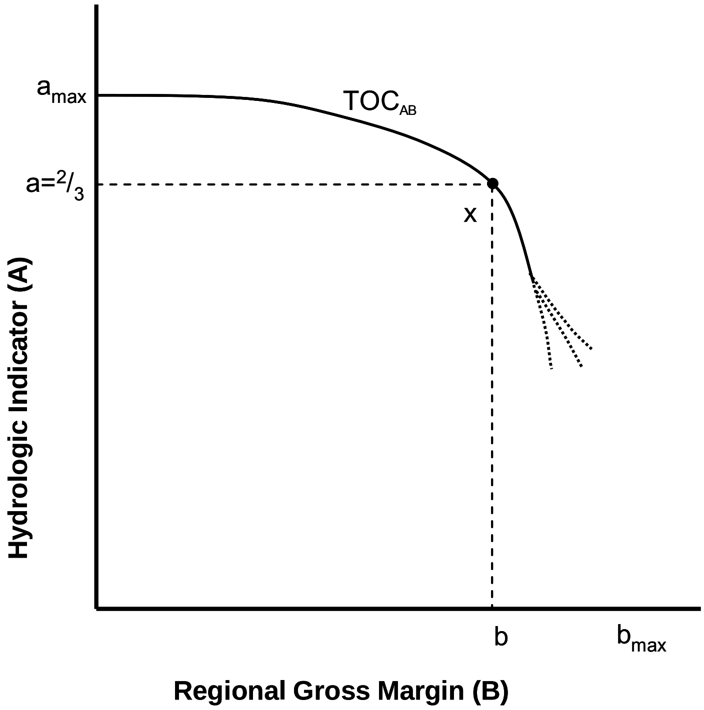


Figure 6.5: Linking the trade-off curve and threshold value analysis to improve decision-making.

In general after point x there can be three broad outcomes (see Figure 6.6). The first is a continuation of the trade-off curve at a similar trajectory to end at point D. This maintains the high level of trade-off of environmental for consumptive benefits. An alternative is shown by curve axC suggesting that at some point after x the continual allocation of water to consumptive uses would have deleterious effects on productive capacity, thereby resulting in a decrease in consumptive outcomes as well as environmental outcomes. This outcome may be plausible over the longer term should the increasing use of water for productive uses lead to increased water or soil quality problems which could impact on productive outcomes. The third alternative, trade-off curve axE, shows an outcome where the trade-off changes again with increasing allocation of water to productive uses. Here continually increasing the allocation of water to productive uses eventually produces an outcome where the incremental negative impacts on the environment become negligible. Past this point the trade-off changes and again leads to higher consumptive benefits. The likelihood of this outcome is related to the ability of the water to be used in productive enterprises. In the BRC where irrigation is currently the largest user of water, the likelihood is related to the ability of irrigators to make use of additional water through activities such as expanding irrigation enterprises.

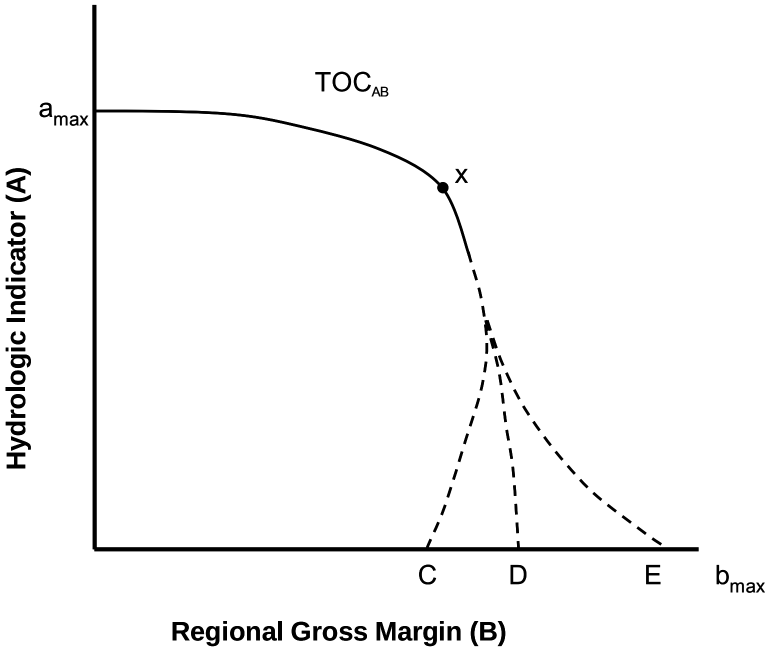


Figure 6.6: Categorising the different shapes of the trade-off curve depending on the relationship between the outcomes using the input – water.

Again the location of the current policy setting on the trade-off curve is of key importance to the outcome of any revision of policy settings. Similarly to the discussion on Figure 6.4 whether the current policy setting is at point x or somewhere around the juncture of the dashed lines in Figure 6.6 will affect the policy decisions. Comparatively a given marginal allocation from in-stream uses to consumptive uses will have a much larger negative trade-off if the starting point is at the juncture of the dashed lines than at x. This is because the slope of the curve, MRT, is higher at the juncture of the dashed lines than at x. A second related significant piece of information in this decision-making framework is the slope of the trade-off curve at the current policy setting. Presumably the policy decisions would be different if it were believed that the frontier was tracking towards point C instead of point D. As alluded to in the above figures, if the MRT at point x is negative, the outcome relationship is competitive and producing more of one outcome will result in less of the other. If the MRT is positive, the relationship is complementary and an increase in one outcome will lead to the production of more of the other outcome. In the event that the slope is either zero (i.e. horizontal) or non-defined (i.e. vertical) the relationship between the two outcomes is supplementary and increasing the production of one outcome will not impact on the level of production of the other output (Doll and Orazem 1984).

6.2.2. Implications from the trade-off framework – simulating increasing off-allocation availability

The first step in considering the study results in the trade-off framework necessitates formulating the results into the conceptual space of a trade-off curve. In practical terms, as outlined in section 4.5, this involves merging the economic results of the simulation where access to off-allocation water was gradually increased (Table L.1 Appendix L) with the results for the environmental indicators reported in Table L.2 (Appendix L). A complication arises due to the fact that there is no single environmental indicator representing the environmental outcome. In the absence of this, subsequent graphs (Figure 6.7 and Figure 6.8) show the trade-off curve relationships from a subset of the indicators (PFlowDur 0.02 and 0.5) where each represents the environmental outcome. The economic outcome is represented by average annual regional gross margin for the simulation.

The economic outcome starts at $27 million because this is the point where off-allocation availability is set at zero and reflects the fact that in the absence of off-allocation water, irrigators can still use on on-allocation water supply, overland flow and rainfall. The maximum economic value of $31 million is achieved when the off-allocation cap is set at 100,000 ML per water year. Therefore, this curve represents the trade-off curve for the outcomes of consumptive use benefits and in-stream use benefits with the input being off-allocation supply[34] in the range of 10 ML/year to 100,000 ML/year.

From the lower left hand graph in Figure 6.7 the curve appears flat indicating that the MRT is zero or close to zero which may lead to the conclusion that a supplementary relationship exists between the two outcomes. The insert enlarges the view to examine the trend and identify any point of inflection where there may be a change in the MRT and hence a sizeable change in the trade-off between outcomes. While noting the scaling effect of the enlargement, it is clear that there is a trade-off between the two outcomes although the MRT would be relatively low in moving from the hydrologic indicator PFlowDur 0.02 to consumptive use i.e. regional gross margin.

[34] Noting that supply in this case does not guarantee availability. This study is set around the use of setting varying levels of a cap on availability. The availability of off-allocation water in any given year is contingent on the flows in the river at any given time and the operational rules in announcing that off-allocation is available.

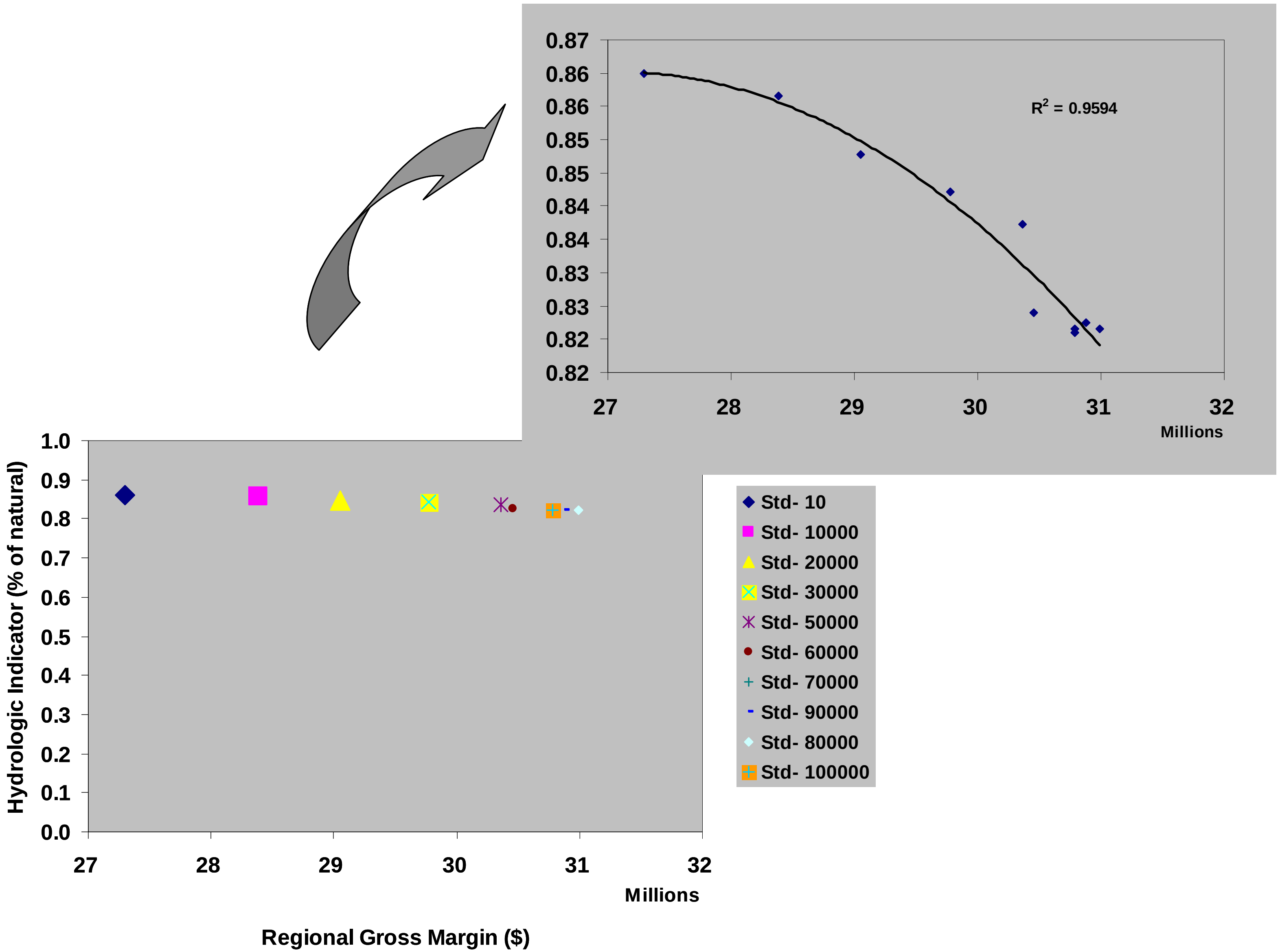


Figure 6.7: Developing the trade-off curve using RGM and PFlowDur 0.02.

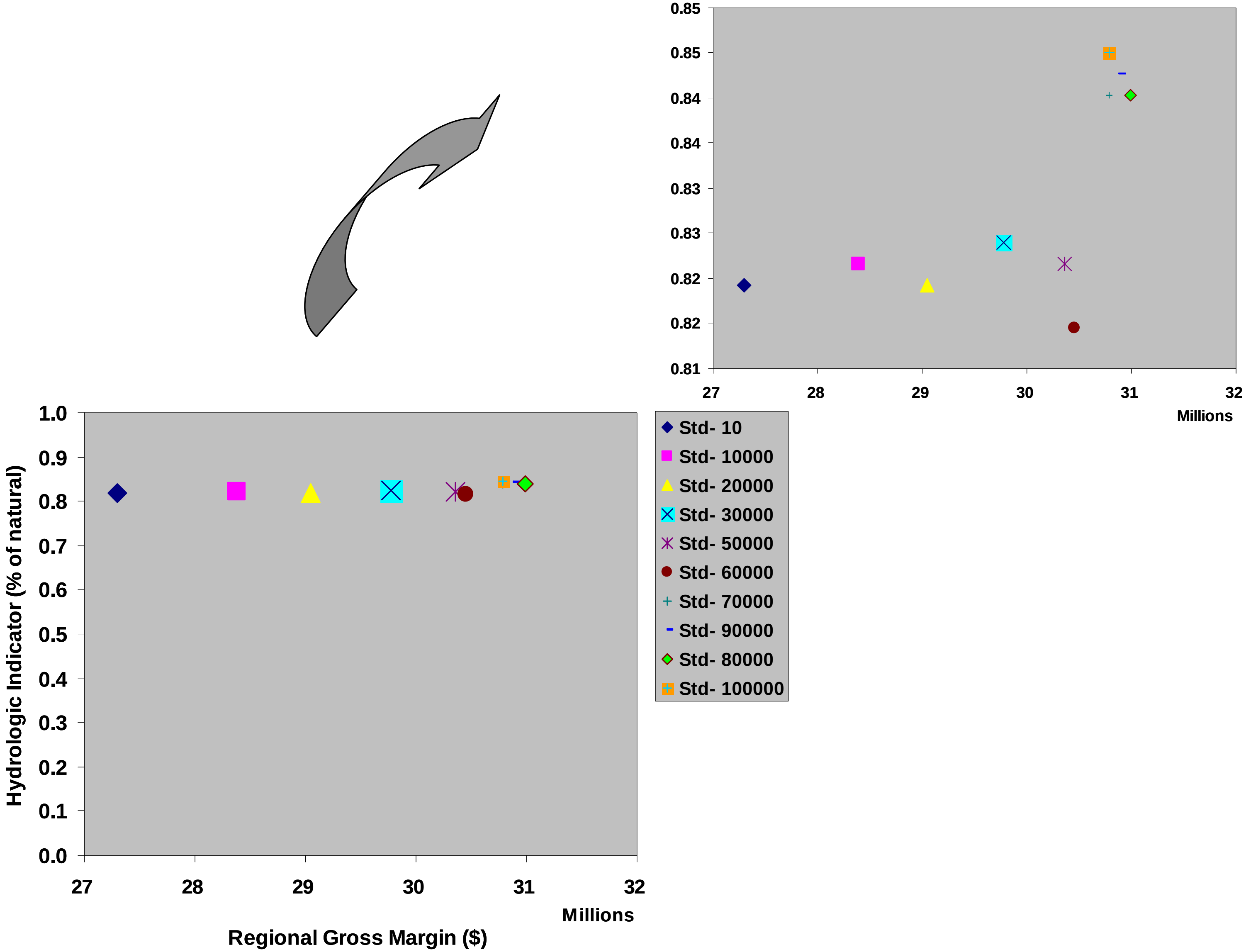


Figure 6.8: Developing the trade-off curve using RGM and PFlowDur 0.5.

The flat response curve which is evident in both Figure 6.7 and Figure 6.8 is likely a result of the hydrologic indicator as both proportion of flow duration percentile indicators record results in the 80 to 90 percentile range. Therefore, despite the increase in off-allocation availability from effectively zero ML/year to 100,000 ML/year, the lower left hand graph appears to be relatively flat.

As previously noted the increased access to off-allocation does not translate to increased off-allocation extraction (see section 5.2.2). This subsequently has an effect on the location of an outcome point on the trade-off curve for a particular level of off-allocation access. If more off-allocation was able to be extracted at a particular access level, *ceteris paribus*, this would lead to increased consumptive use and decreased in-stream environmental outcomes. This effect can be explained by examining the off-allocation use of the scenarios run to develop the trade-off curve.

In the scenario where the off-allocation cap is set at 100,000 ML/year for all years, in 89 of the 101 year simulation (1894 to 1994) the off-allocation water actually diverted was less than that permitted and available. While setting the off-allocation cap to 100,000 ML/year gives a nominal cap of 10,100,000 ML (101 years x 100,000 ML) over the simulation period, this assumes that the full 100,000 ML is available every year. Recall that off-allocation is not a *fait acompli* and access within a water year may be announced by State authorities when dams spill or high flows enter the river system and only after all other user needs are met. For the 100,000 ML scenario the amount of off-allocation that irrigators were able to access over the period 1894 to 1994 was calculated by the IQQM at 4,743,949 ML. The difference between this total of off-allocation diversions permitted and that actually diverted was approximately 19 percent (4,743,949 – 3,842,046 ML). This pattern is also a feature of the baseline (50,000 ML off-allocation cap) scenario although the difference is reduced to 6 percent (2,957,295 – 2,772,662 ML). An example of the comparative permitted volumes and diverted volumes of off-allocation is shown in Table 6.1. This pattern of under-usage of off-allocation water suggests that in the case of the baseline scenario some 184 633 ML of un-used off-allocation water or on average 1 828 ML/year was available for irrigators to use. In the 100,000 ML scenario this climbs to 901,903 ML not accessed or 8,930 ML/year.

Table 6.1: The difference between permitted off-allocation diversions and actual off-allocation diversions (ML) over the water year 1 October-30 September

	Baseline (50,000 ML) scenario		100,000 ML scenario	
	Off-allocation permitted	**Off-allocation diverted**	**Off-allocation permitted**	**Off-allocation diverted**
1894	34,265	33,401	34,763	33,744
1895	9,971	8,901	9,997	8,945
1896	21,604	21,274	17,197	17,197
1897	33,365	29,653	33,475	29,781
1898	28,466	23,413	28,486	23,369
1899	49,167	46,678	49,838	47,392
1900	18,863	18,863	18,862	18,862
1901	0	0	0	0
1902	58,744	58,683	130,764	121,058
1903	69,557	61,253	175,322	79,650
.	.	.	.	.
.	.	.	.	.
1994	.	.	.	.
Total	2,957,295	2,772,662	4,743,949	3,842,046

A consequence of the underutilisation of available water could be that irrigators change their operations to take advantage of these volumes by, for example, building infrastructure to allow more of the available water to be used. Should this occur there would be a movement of the frontier outwards as shown in Figure 2.5 and a subsequent decrease in in-stream benefits, and a corresponding increase in consumptive use benefits. In effect, building further storage capacity results in more off-allocation water being diverted which would subsequently decrease the amount of water able to produce in-stream benefits and we would expect a worsening in the hydrologic indicator. The level of impact on both consumptive and in-stream benefits is contingent on the slope of the frontier at the starting point as illustrated using Figure 6.4 and the volume of off-allocation water diverted that was previously underutilised.

In addition to the flat response curve a comparison of Figure 6.7 and Figure 6.8 highlights a key difficulty encountered in this study. While the slope of the trade-off curve (MRT) for Figure 6.7 is very low it does nonetheless display much of the characteristics expected of a trade-off curve. There is a competitive relationship or at least a supplementary relationship between the two outcomes and it is concave to origin. On the other hand Figure 6.8 does not display either of these characteristics and as shown by the insert exhibits very little obvious relationship or trend. Therefore the change in pattern of water use when access to off-allocation is increased for consumptive uses does not result in more of 0.50 percentile flows occurring. Consequently, the PFlowDur 0.5 indicator does not move closer to one as more water is allocated to in-stream benefits which if it occurred would represent a move to the pattern of this type of flows that occurred in the natural state (see section 5.2.3).

Limiting levels of water use and the effect on the shape of the trade-off curve

This study starts out assuming a typically convex trade-off curve showing a competitive relationship between economic and environmental outcomes. The shape of the curve is determined by the expected relationship between the relative outcomes and it is clear that the results obtained did not suggest a strong competitive relationship. This leads to the conclusion that for a large range of the study scenarios the relationship between economic and environmental outcomes under conditions of increasing off-allocation is supplementary or at least close to it. The results generally show a slight increase in consumptive use benefits as evidenced by the increases in the regional gross margin indicator with little or no change in in-stream use benefits. This finding is closely related to the limited increase in the actual use of water when off-allocation availability is increased (refer to Figure 5.3), thereby resulting in little impact on the hydrologic indicators and therefore environmental health.

One conclusion from this analysis may be that the response of hydrologic indicators to changes in the way water is used is limited. However this is not the case. In preparing to develop the water resource plan for the Border Rivers catchment the Queensland Department of Natural Resources and the NSW Department of Land and Water Conservation (2000) tested a number of scenarios that simulated increasing demand for water and assessed the impact on hydrologic indictors. It should be noted that the version of IQQM used in the Queensland Department of Natural Resources and the NSW Department of Land and Water Conservation (2000) study had a number of differences to that of this research and that the number of irrigation districts and therefore consumptive users, included was larger. The scenarios tested included:

1. historical levels of development, (91/92 , 93/94, 98/99); and,
2. two development projections (PR1, PR2).

The scenarios under point 1 represent the level of development for taking water that existed during the 1991/92, 1993/94, and 1998/99 water years. The development projection PR1 tests an initial short-term incremental increase in development assuming current irrigation practice and no additional environmental flows or diversion limits above the existing licensing limitations. Case PR2 tests increasing levels of development in the long-term via a doubling of the on-farm storages, planted areas, and NSW pump capacities from that used in the 1998/99 case.

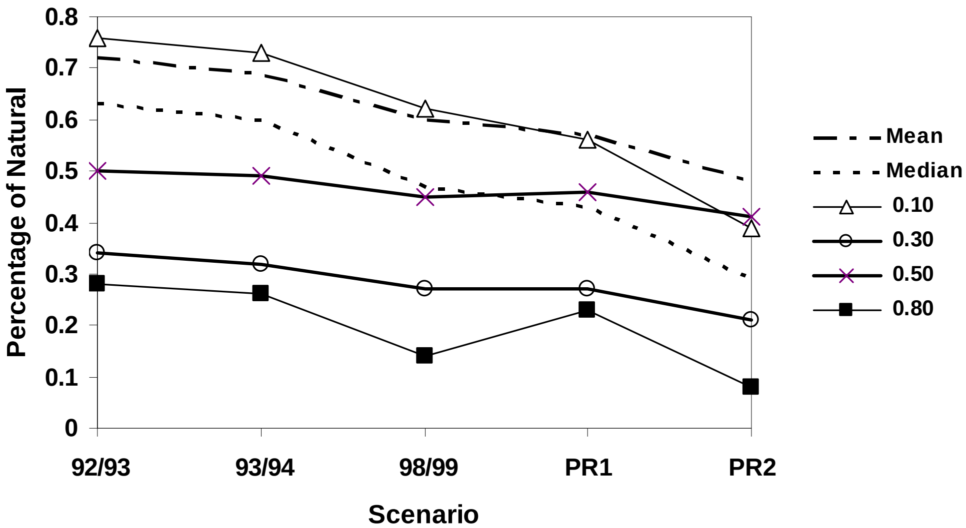
The results of these scenarios reported in Table 6.2 and shown in Figure 6.9 confirm that an increase in water use to levels greater than that assessed in this study will have substantial impacts on hydrologic indicators far in excess of the impacts found in this study. While no economic impact analysis was undertaken as part of the Queensland Department of Natural Resources and the NSW Department of Land and Water Conservation (2000) study to allow a trade-off curve to be developed, these results indicate that should another study consider higher impacts on water use, then a trade-off curve which exhibits a larger response of hydrologic indicators would be the expected result.

Table 6.2: Hydrologic flow statistics for scenarios testing different water use scenarios

Hydrologic indicator	Scenario				
	92/93	**93/94**	**98/99**	**PR1**	**PR2**
Mean AF	0.72	0.69	0.6	0.57	0.48
MAF	0.63	0.6	0.47	0.43	0.29
10 percent	0.76	0.73	0.62	0.56	0.39
30 percent	0.34	0.32	0.27	0.27	0.21
50 percent	0.5	0.49	0.45	0.46	0.41
80 percent	0.28	0.26	0.14	0.23	0.08

(Source: Department of Natural Resources and Department of Land and Water Conservation 2000).

The implication for my study is that despite the differences between the IQQM model used in this research and that for Department of Natural Resources & Department of Land and Water Conservation (2000), the general conclusion that a competitive relationship exists between consumptive and environmental outcomes remains valid. The results in this study indicate a supplementary relationship occurs because of the limited increase in the water use over the baseline in this study. Considering Figure 6.9 which is a graphical representation of Table 6.2. If the results of my study were combined into this graph, it is likely that the results of this research would be located on the first part of the curves close to the y axis. In this part on the curve the response to changing levels of water use is small, hence the curve is flatter and suggests a weak competitive or close to supplementary relationship. As the consumptive use of water becomes larger, the hydrologic indicators begin to show a marked response indicating a competitive relationship.



(Source: Department of Natural Resources and Department of Land and Water Conservation 2000).

Figure 6.9: Change in hydrologic indicators from scenarios run as part of the water reform planning process.

Difficulty in ascribing a single environmental indicator to undertake the trade-off analysis

The implication of the apparently conflicting result between Figure 6.7 and Figure 6.8 is difficult to ascertain. The finding is seemingly at odds with the two-thirds rule of Jones et al. (2003) which strongly implies that a trade-off or competitive relationship exists between the economic and environmental outcomes as relative water use changes. In this study the implications of changing off-allocation availability are minor as the absolute value of the changes to hydrologic indicators are small indicating that there is no negative (or positive) impact on riverine health. However, if the study had tested scenarios that had a larger impact on water use this conflicting result may be overcome.

This study used a range of hydrologic indicators in order to ascertain whether the impacts on the riverine environment from using SCF information in water management rules are positive or negative. These indicators include: mean annual flow ratio (Mean AFR); median annual flow ratio (MAFR); and, the proportion of flow duration percentile (PFlowDur). In order to undertake a practical trade-off analysis there needs to be well specified aims or outcomes. These outcomes can be economic, environmental or even social outcomes (not considered in this study). Within each of these outcomes there should be some rationale for aggregating indicators to single outcomes for comparison. Other studies have used a range of indicators, for example Boscolo and Vincent (2003) used indicators for biodiversity, carbon sequestration and value of timber production as outcomes. Calkin et al. (2002) used the value of timber and the likelihood of species persistence in the trade-off analysis. The single economic indicator, regional gross margin, used in this study was selected to represent economic outcomes but no single hydrologic indicator represented environmental outcomes. This presents a problem particularly when the indicators respond in opposing directions to scenarios. For example the indicator PFlowDur 0.5 did not respond as might be expected to increasing water availability for irrigators. Instead of decreasing over the scenarios as water access to irrigators was increased no trend was observed Figure 6.8.

It is possible to use a number of indictors to represent an environmental outcome provided there is either a method of aggregating to a single score, such as is done with a biodiversity index, or that each indicator represents a different environmental outcome. Such might be the case where one indicator is chosen because it represents impacts on fish populations and another because it represents the health of riverine macro invertebrates. When this method of analysis is undertaken the problem becomes one of optimising against three outcomes rather than assessing the trade-off between the two used in this study.

When considering the likelihood of adoption of the rules considered in this study, the choice of three or more outcomes presents a number of difficulties. In the first instance an optimisation of three or more variables is more difficult, although not necessarily insurmountable. Secondly, it becomes much more difficult to explain the results to, and gain support from, stakeholders affected by the new management rules. This is not advocating that there is an absolute need for a simple environmental indicator (although the simpler the better) it recognises that adoption is heavily reliant on understanding and acceptance of the way outcomes are measured. Complicated and conflicting measures are unlikely to gain the acceptance of stakeholders and lead to a conflict over the science behind the environmental assessments. On the other hand, overly simplistic less robust measures will be easy for the disaffected to challenge. Recent gains in the use of market based instruments in the field of natural resource management often rely on quite complex environmental benefit indicators (EBIs) to guide choice amongst a range of options (Stoneham et al. 2003; O'Connor 2008; Whitton et al. 2004; Parkes et al. 2003). These may provide an avenue for progress particularly if linked to the research being undertaken into assessing riverine health. Where more knowledge on appropriate flow management regimes, and indicators to measure these regimes, are developed their incorporation into an index would be beneficial for the type of analysis undertaken in this study.

Adopting a simplified environmental threshold rule to determine impacts on riverine health

An alternative to the use of specific hydrological indicators to assess the impacts of the various scenarios on riverine health and its associated complications, is the more general rule proposed by Jones et al. (2003) where there is a high probability that the river is in a healthy working state if the key hydrological attributes are above two-thirds of natural. See Table 2.1 which outlines the thresholds of hydrological indicators where the probability of having a healthy river changes.

Using this approach, rather than focusing on the specific value of and directional change of each hydrological indicator, the assessment is carried out on the group of hydrological indicators. Arthington et al. (2006) criticism of this approach (see section 2.3.2) need to be considered when undertaking this simplified approach. In the case of this study an assessment of the results based on the changes to the environmental indicators suggests that there is a high probability of having a healthy river for the scenarios assessed as they all are above two-thirds of natural. Therefore it can be interpreted that there is no negative outcome from increasing off-allocation access to 100,000 ML nor from using the SCF information based rules either for the SOI phase or SOI value scenarios. In this circumstance the value of information based on the intended outcomes rests solely on the economic outcomes.

6.2.3. Using the trade-off curve to assess the efficacy of the SCF scenarios

Despite the difficulties identified above, the seasonal climate forecast scenarios were incorporated into the trade-off framework. This step assists first in examining how the result of each scenario compares with the baseline starting position. Second this comparison is a visual depiction of Table 5.13 which identified the Pareto outcome for each scenario. Finally the trade-off framework provides insight into the trade-off between outcomes that would occur should the SCF rules be adopted into decision-making. The results presented in Figure 6.10 combine the trade-off curve generated in Figure 6.7 with the outcomes of the SCF scenarios (Figure 5.13 - Table L.3 and Table L.5). The figure is generated in a stepwise process where the first step is to estimate the trade-off curve as in Figure 6.10. The frontier is estimated by a fitted curve using the Microsoft Excel trend line function on the results of the simulation (indicated by X in Figure 6.10), where the access to off-allocation was gradually increased from 10 ML/year to 100,000 ML/year identified in Table 4.7. The trend line is quadratic with an R^2 of 0.97 and is an approximation of the trade-off curve. The pink square points are the outcomes for each of the SOI phase scenarios and the round grey point delineates the ENSO scenario. As previously noted, a movement between scenarios results in a trade-off between environmental and economic outcomes.

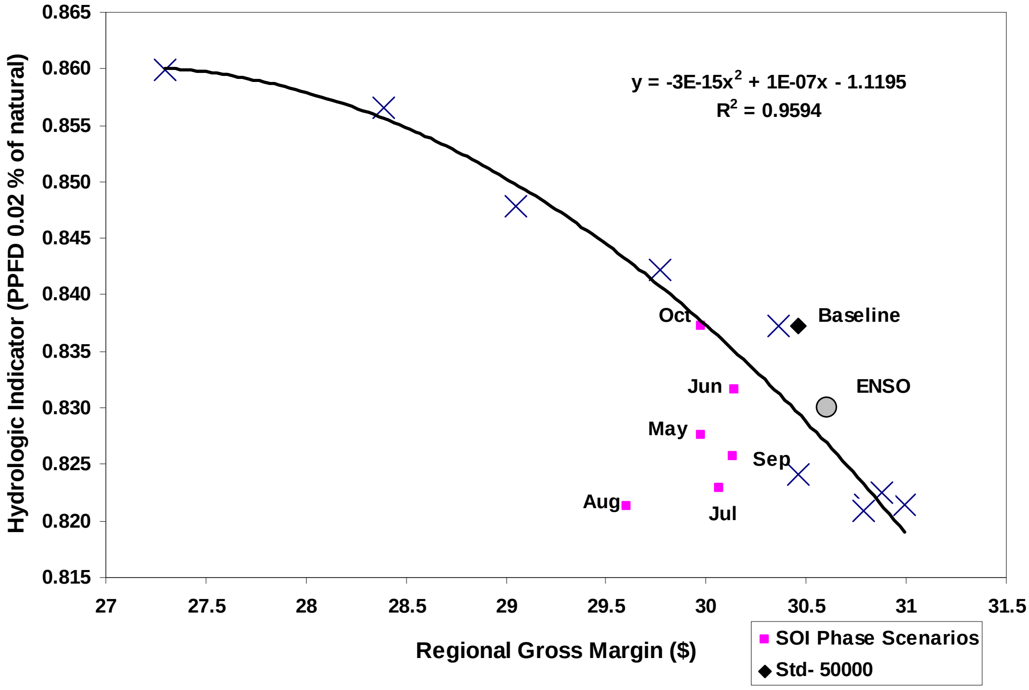


Figure 6.10: Comparing the SOI phase and ENSO scenario outcomes in trade-off curve space.

The graph in Figure 6.11 follows the same process as that of Figure 6.10, however in this case it reports the results for the SOI value scenarios. The solid line is the fitted curve of the results for the scenarios with off-allocation increasing from 10 ML/year to 100,000 ML/year.

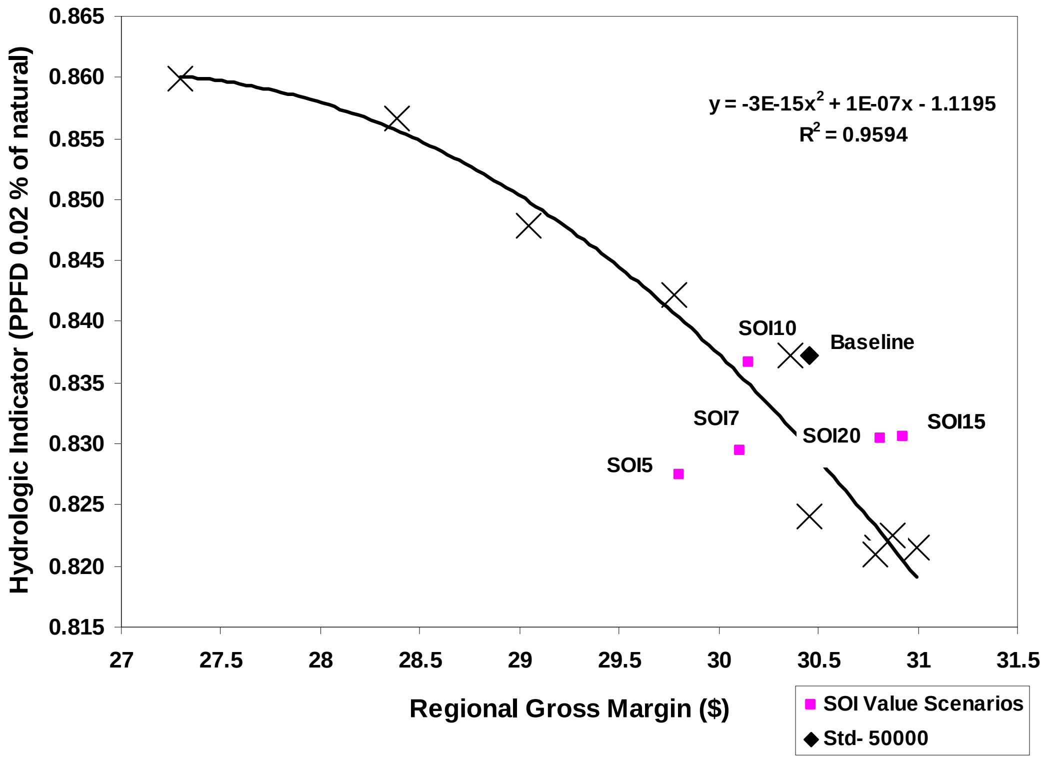


Figure 6.11: Comparing the SOI value scenario outcomes in trade-off curve space.

The first point evident from Figure 6.10 and Figure 6.11 is the variability around the estimated trade-off curve. This is not unexpected as the results reported in chapter 5 did not indicate conclusive linear responses to increases in off-allocation access. The mean and median regional gross margin results reported in Figure 5.2 do exhibit strong linearity but this is tempered by the trend in diminishing marginal returns as access to off-allocation increases reported in Figure 5.4. This trend shows some variability that was earlier ascribed to the existence of bottlenecks in the water supply and use system. The results for environmental indicators reported in Figure 5.7 on the other hand exhibit very little linearity. This trade-off curve pattern is not dissimilar to the pattern reported by Calkin et al. (2005) in developing a production possibility frontier for fire threat reduction and late successional forest. In that case the trade-off curve displayed a similar trend to that in Figure 6.10 and Figure 6.11 with a number of points in the trade-off curve being slightly off the curve. The impact of the weak linear response is evident in the positioning of the baseline or 50,000 ML scenario and raises the question of how this affects the interpretation of the graph.

The approach taken in this study is to consider the relative position of results rather than the absolute position. Therefore if the baseline scenario is considered to be on the estimated trade-off curve then so to must the June SOI phase results in Figure 6.10. Even without this complication the decision on whether or not the SCF scenarios represent a better outcome than the baseline scenario is a complex issue. Figure 6.10 captures the results presented in the previous chapter (Figure 5.14 - Table L.4 and Table L.6, Appendix L) but highlights some interesting points about the relative outcomes. In Figure 6.10 if we assume that the actual trade-off curve in fact runs through the points identified by the October SOI phase, baseline and ENSO scenarios the choice of optimal outcome is not clear cut.

From a theoretical perspective the optimal point equates to the point of maximum social utility which is achieved using the concept of a social indifference curve (SIC) (Boadway and Bruce 1984). The point of maximum social utility is found by combining the production possibility frontier and the social indifference curve showing utility as in Figure 2.11. In terms of Figure 6.10 the socially optimal point would be found by mapping the SIC in relation to the trade-off curve. A hypothetical SIC – trade-off curve (Figure 6.12) identifies that the baseline scenario on SIC μ_2 as the optimal point as it is on the highest SIC. The SCF scenarios October SOI phase and ENSO are on the lower SIC and are therefore considered sub-optimal. Furthermore, as the remaining SOI phase scenarios are inside the trade-off curve, and also below the highest SIC, they are also sub-optimal.

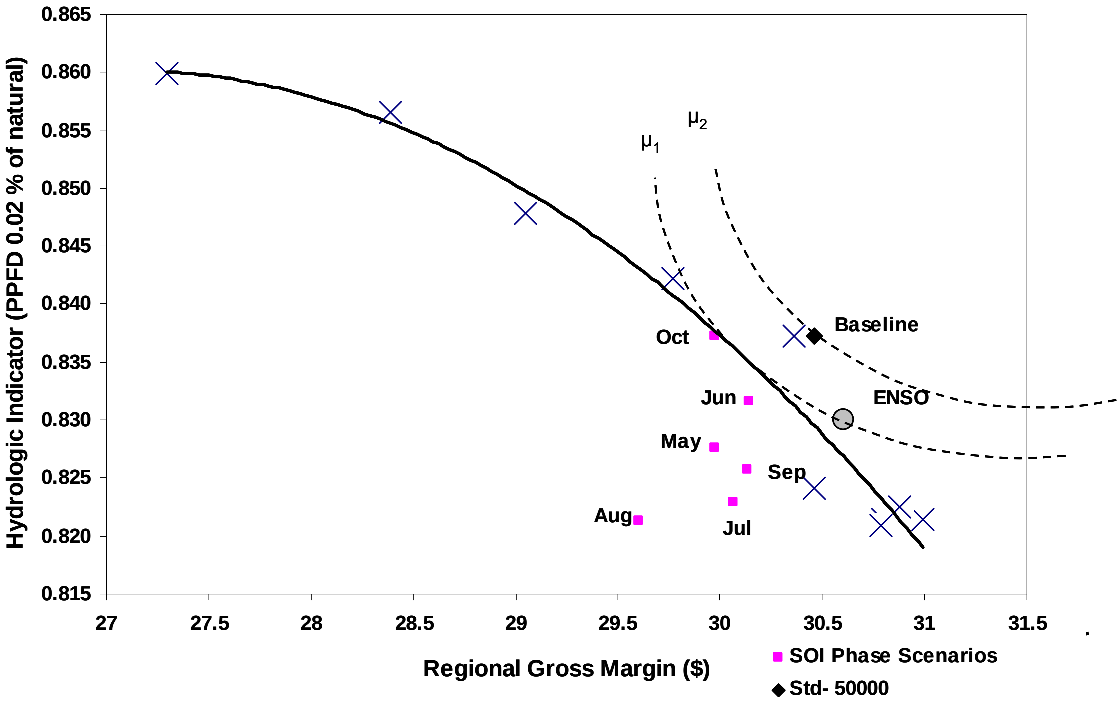


Figure 6.12: Determining the point of maximum social utility by mapping the social indifference curve with the trade-off curve.

This type of analysis though needs to be kept in perspective as we have no knowledge on the placement or shape of the SIC. An alternative SIC could look like that in Figure 6.13 where due to a change in the slope of the SIC, socially there is no difference between the October SOI phase and baseline outcomes.

Dudley et al. (1997) discuss the mechanics of choosing amongst options such as these. They propose that when the PPF (trade-off curve) is developed using extended benefit-cost analysis then the optimal mix of outcomes can be determined by identifying the point on the PPF (trade-off curve) where total outcomes are maximised. In CBA terms this is the point where maximum revenue is attained. As all points on the PPF have constant total costs (see Doll and Orazem (1984)) this point also represents the point of maximum profit. In situations such as this study where the PPF is not developed using full CBA principals, Dudley et al. (1997) argue that this means the trade-off comparison is "flawed" in that it is not comparing like outcomes. While coming from a different perspective Dudley et al. (1997) have rationalised that the problem is one of TVA analysis and is more akin to cost effectiveness. In this case the trade-off comparison is in terms of regional gross margin measured in dollars and an environmental indicator measured in non-dollar terms. Furthermore, they argue that it is therefore not possible to identify the revenue or outcome maximising point and that final choice of which scenario is optimal is made by the decision-maker such as a politician.

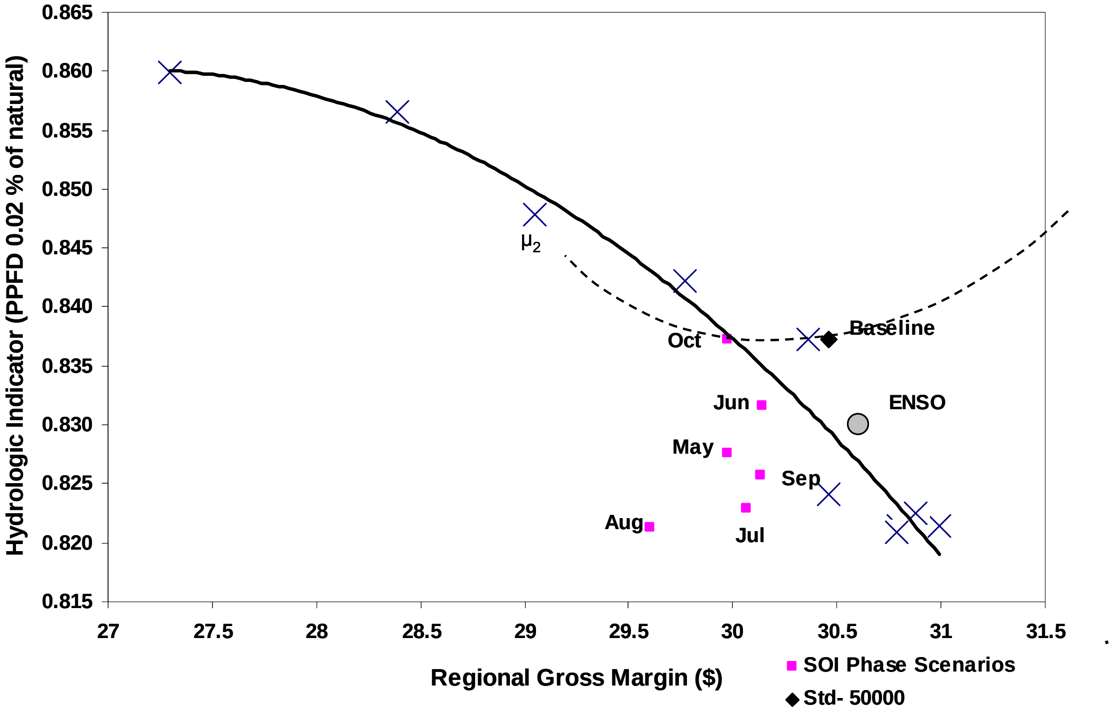


Figure 6.13: An alternate view determining the point of maximum social utility by mapping the SIC with the trade-off curve where the SIC is characterised differently.

While the choice amongst those scenarios with outcomes falling on the trade-off curve, assumed to be October and June SOI phase, ENSO and baseline scenarios, is not clear cut, the choice between these scenarios and the remaining scenarios is potentially more straight forward. Standard theory of production possibility frontiers holds that points on the boundary signify Pareto efficient states where resources are efficiently allocated between uses. Alternatively, any change in resource allocation would not result in making one person better-off without making another worse-off. However, translating to this study where the trade-off curve is analogous to the PPF, points inside the boundary such as May, July, August and September SOI phases scenarios, suggest inefficient resource use and are therefore not Pareto optimal. Moving from any of these points towards the boundary in a north-easterly direction results in either win-win or win-no-change outcomes and represents a Pareto improvement. As the May, July, August and September SOI phases scenario results are within the boundary they are identified as being inferior to the results of the SCF scenarios June SOI phase, October SOI phase and ENSO.

In the case of Figure 6.11 the SOI <-10,>+10, SOI <-15,>+15 and SOI <-20,>+20 could be assumed to be on the trade-off curve and therefore Pareto optimal, whereas the SOI <-5,>+5 and potentially SOI <-7,>+7 scenarios appear to be within the boundary and are therefore inferior. The conclusion from the graphical analysis is that there is a set of SCF scenarios that could be assumed to be on or close to the trade-off curve and are therefore in the efficient set. This finding is at least partly contrasting to that of Table 5.13 which identified only the ENSO, SOI <-15,>+15 and SOI <-20,>+20 scenarios as being in the efficient set.

While acknowledging the points of Dudley et al. (1997) regarding the point of maximum revenue/outcome, it is also possible to determine which scenarios are the most preferable from society's point of view. One way is to use extended CBA to ascribe monetary values to the environmental outcomes thereby permitting the point of maximum revenue/profit to be attained. However as has been mentioned earlier, this involves the use of estimating non-market values, an analytical method that has its problems in terms of acceptance for use in making policy decisions (McMahon et al. 2000; Lockwood and De Lacy 1992; Young 1996).

An alternative that may be more practical is to consider the intention of the water policy reform. In particular the intention of ARMCANZ (1995) identified the goal of water reform as achieving the "highest and best value of the limited resource for community benefit whilst ensuring that use of the resource is ecologically sustainable" (ARMCANZ 1995, p. i). More specifically, principle 5 of the *National Principles for the Provision of Water for Ecosystems* states that "further allocation of water for any use should only be on the basis that natural ecological processes and biodiversity are sustained (i.e. ecological values are sustained)" (ANZECC 1996, p. i). In effect these statements can be interpreted as meaning that the maximum economic value should be obtained from using the water resource provided the riverine environment is maintained in a healthy and sustainable state.

Referring back to the trade-off curve curves of Figure 6.3 this can be interpreted as the point x which represents the level of allocation that society deems is the minimum allocation to consumptive use benefits. In Figure 6.5 point x represents the place at which the environmental – economic trade-off changes dramatically i.e. on the lower side of x the relative trade-off is larger and proportionally larger losses in the level of hydrologic indicator have to be forgone to achieve small economic gains. Reviewing the study results in Figure 6.7 and Figure 6.8 identifies that the trade off for off-allocation water in this study is very flat and no point of inflection is found. This is largely due to the absence of any discernable tipping point in environmental outcomes. That is, no significant decrease in environmental indicators was calculated across the scenarios tested, implying that there are no negative impacts on riverine environmental health. However, as pointed out when discussing Figure 6.9 if more of the total water supply had been allocated to consumptive uses, then we would expect to see a point of inflection appear at some level of water use.

The location of the point of inflection is an important factor for the (threshold) trade-off analysis and the identification of the scenarios which may be acceptable and those which may not. The adoption of the two-thirds rule of Jones et al. (2003) provides what is potentially a useful measure for making this decision. Figure 6.14 graphically illustrates the decision in relation to the two-thirds rule and the point of trade-off change. Here the point where the environmental flow indicator reaches two-thirds of natural is identified by x and is a point of inflection where from an environmental perspective the environmental health of the river system begins to decline markedly.

Between the points (a=2/3, x) and (a=1/2, y) the probability of having a healthy working river drops to moderate. Below the next point of inflection (a=1/2, y) the probability of a healthy working river decreases again to low. Outcomes to the right and below the point (a=2/3, x) are deemed by society to be unacceptable as they would be contrary to the goal of ARMCANZ and principle 5 of the *National Principles for the Provision of Water for Ecosystems*. In effect this analysis simplifies the trade-off for a decision-maker if the outcomes are close the point of inflection.

The results shown in Figure 6.10 and Figure 6.11 are also illustrated in Figure 6.14 and can be interpreted as showing that the SCF scenarios ENSO, SOI <-15,>+15 and SOI <-20,>+20 are to the right of the baseline but to the left of x. Therefore these scenarios point to some level of efficacy in terms of value of information as they result in higher economic benefits without degrading the environmental benefits to the point at which environmental health is compromised. These findings would tend towards supporting hypothesis 1 showing the use of the seasonal forecasting tools to manage off-allocation water access by irrigators may improve the regional gross product of irrigated agriculture without major negative impacts on environmental health. The small scale of the findings however, mean that from a policy implementation perspective transaction costs of implementation are likely to prevent the use of SCF information being implemented in water management.

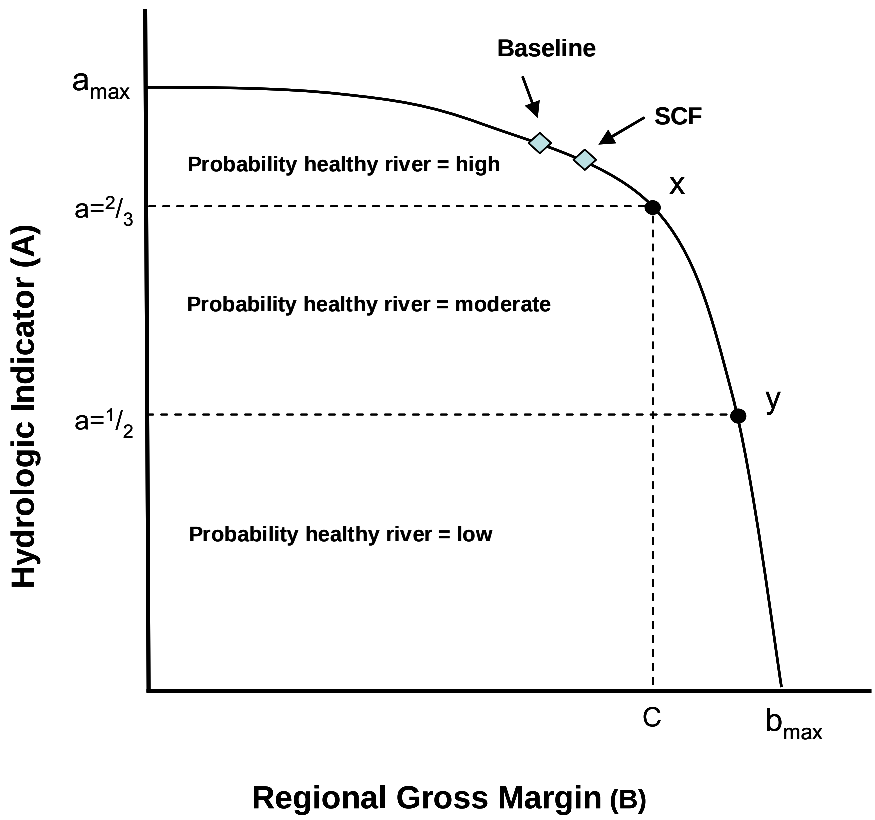


Figure 6.14: Depicting the trade-off analysis in relation to the two-thirds rule.

6.3. Reviewing the value of forecast information

The process of determining the value of forecast information assumes that the use of a perfect forecast will produce the best outcome at least in terms of economic outcomes. As noted in the methodology section the value of information can be described as:

$$\Omega < \chi < \partial \qquad (1).$$

Where ∂ refers to the perfect forecast, χ refers to the seasonal climate forecasting scenario(s) and Ω represents the without forecast scenario. The economic value of forecast (VoF) is calculated by:

$$\text{VoF} = \chi - \Omega \qquad (4).$$

The results reported thus far have in most cases provided a negative economic VoF with the exceptions being the SCF scenarios ENSO, SOI <-15,>+15 and SOI <-20,>+20. For instance, comparing the baseline scenario RGM result of \$30.49 million and that for the SOI <-20,>+20 scenario of \$30.81 million gives a VoF of \$355,858. However as alluded to in (1) there may be considerable upside to the VoF depending on the economic return when using a perfect forecast.

In the methodology it was pointed out that a perfect forecast was unable to be modelled, but that one way to overcome this is to adopt a proxy perfect forecast scenario where no restrictions were put on water use. In effect this scenario permits off-allocation water to be taken by irrigators whenever it is available without restriction (up to a total of 100,000 ML per year). Thus (1) became:

$$\Omega < \chi < \partial' \qquad (6).$$

The extension to this is that the value of a perfect forecast (PFV) is:

$$\text{PFV} = \partial' - \Omega \qquad (7).$$

From Table L.1 (Appendix L) the result for the 100,000 ML scenario (proxy perfect forecast) is RGM \$30.98 million and the PFV is \$524,005. A comparison with the VoF calculated using (4) shows that some 68 percent of the PFV is explained by the VoF. Therefore, while the conclusion reached in the previous section was that the benefit may be very minor and possibly too small to justify incorporation of the SCF information into water resource management decision-making, nonetheless the SCF scenarios explained between 28 and 89 percent of the value of perfect forecast information. The calculation of these values is summarised in Table 6.3.

Table 6.3: Summarising the value of forecast information for those scenarios where it is positive

	Baseline	SOI <-15,>+15	SOI <-20,>+20	ENSO	100,000 ML	PVF
Variable	Ω	χ	χ	χ	∂'	
Equation number		(4)	(4)	(4)		(7)
RGM	30,458,094	30,921,933	30,813,952	30,603,491	30,982,099	
Value of forecast	na	463,839	355,858	145,396	na	524,005
Percent of the PFV is explained by the VoF		89 percent	68 percent	28 percent		

The other side of the trade-off, the environmental outcome, is more difficult to consider in the value of perfect information framework. The proxy perfect forecast was developed essentially with regard to maximising economic outcomes. It is assumed that if the volume of water available for irrigation is known with certainty then irrigators will be able to make improved decisions and maximise economic outcomes. The environmental outcomes from this were unknown in advance but logically might be expected to be unfavorable. The application of a perfect forecast to production decisions to maximise economic outcomes should lead to a decrease in environmental health. The result reported in Figure 5.7 and Table L.2 indicate that while some of the hydrologic indicators do decrease by one to two percentage points there is no steep decrease in environmental health from the perfect forecast scenario. This result is not unexpected given the water use results reported in Figure 5.3 showing mean water use under the proxy perfect forecast scenario is only 1.3 percent larger than the baseline scenario (112,784 and 111,267 ML respectively)

The conclusion therefore in relation to the second hypothesis is that an increase in forecast accuracy would improve regional gross margin without negatively impacting upon the environmental health of the riverine environment. Again however, the overall size of the value of the perfect forecast remains small in comparison to the regional gross margin attained under the baseline scenario. A comparison of the PFV at $535,498 with the RGM attained for the baseline scenario of $30.49 million highlights that the improvement is only 1.8 percent. The small size of both the gains from the use of both the proxy perfect forecast and the SCF scenarios leads to the issue of adoptability and what barriers there may be to adoption and subsequent implications for the results of this research.

6.4. Adoption of SCF information into decision-making

Long and McMahon (1996) reported that water resource managers in Australia use SCF information in a qualitative background manner implying a zero value of information (VOI). In simple terms, at the very least before SCF information will be adopted in decision-making processes, stakeholders need to be convinced that its use will meet the goals of the water reform process. That is assist in achieving the "highest and best value of the limited resource for community benefit whilst ensuring that use of the resource is ecologically sustainable" ARMCANZ (1995, p. i). The results of a subset of scenarios, while being marginal, do identify some potential for adoption.

The simple approach of a positive VOI dictating adoptability however, has been found to be unsound. Barriers to the adoption of SCF in water resources management in the USA have received considerable recent attention (Callahan et al. 1999; Changnon et al. 2003; Lee 1999; Pagano et al. 2001; Pulwarty et al. 1997; Pulwarty et al. 2001; Wernstedt et al. 2002). These studies have also provided recommendations to overcome barriers to adoption which are directly relevant to this study and are likely to be equally relevant in the Australian water industry. The congruency of issues affecting the use of SCF information in water resources management and similarities in water management aims between the two countries suggests that strategies proposed for overcoming these barriers in the USA are likely to be useful in the Australian context. The four commonly reported barriers/recommendations relate to:

1. technical e.g. accuracy, relevance, understandability, timing;
2. lack of (successful) demonstrated use;
3. poor communication between forecasters and potential forecast users and lack of forecast interpretation resources; and,
4. a need to address institutional barriers in the water management organisations in particular institutional decision-making inflexibility.

These barriers are all applicable to the findings of this study and their adoptability to some degree but are discussed in the context of the two barriers with the strongest link to the study.

6.4.1. Considering the impact of accuracy on study findings

Consistent with findings in the USA, a major reason for the finding by Long and McMahon (1996) is likely to be the perception by water industry stakeholders that SCF is too inaccurate to be used confidently. Chiew, Zhou and Panat (2003) found that the level of SCF accuracy was insufficient for use in a low risk urban water management setting.

Accuracy of forecast is also a misunderstood concept. In excess of eighty percent of respondents in structured interviews conducted by Pulwarty and Redmond (1997) to examine the knowledge and degree of use of SCF stipulated a need for the forecast to be right at least 75 percent of the time in order to be useful. Changnon and Vonhamme (2003) reported a similar finding, but also found that no-one interviewed could identify the skill range of SCF. The term “inaccurate”, from a user perspective is usually applied in relation to the success (or lack of) of a forecast in predicting an outcome in terms of rainfall or perhaps streamflow volumes or events (Hartmann et al. 2002; Ritchie et al. 2004). What may stop managers from incorporating a climate forecast in decision-making is not its relative accuracy, but the implications arising from the outcomes (Ritchie et al. 2004).

Different organisations have a range of outcomes they are seeking. Businesses such as irrigation farms are expected to have maximising profit as a key driver, whereas government agencies, including those managing water resources generally have broader goals including environmental outcomes. The use of probabilistic information in decision-making by both of these types of institutions is widespread. At its core the use of probabilistic information comes with a risk to outcomes. This necessitates that institutions either explicitly or implicitly take a risk position whereby they may be prepared to take more or less risk for the possibility of achieving higher or more certain outcomes. Institutions such as irrigation farms bear the outcome of their own risk decisions, however for government water management agencies the outcomes are borne by third parties such as irrigators. Therefore the decision-making procedures in these institutions are usually geared to minimising risk. In the context of this study, raising the off-allocation cap when the seasonal climate forecast indicates a wet year is a decision based on a probabilistic forecast. Should irrigators make decisions to expand the area planted to irrigated crops and the year is much drier than expected then the irrigators will bear the costs of crop failures. They will then perceive the forecast as inaccurate and consequently would not be in favour of adopting SCF information into water management decision-making. This issue is congruent with that reported by Pulwarty and Redmond (Pulwarty et al. 1997) who found that institutions with profit motives were able to embrace risk while those without, such as public institutions, avoided risk based on climate information.

Not only is forecast accuracy difficult to define and given different names (Mjelde 2002), its meaning is contingent on the user. To a climatologist, the accuracy of a forecast may refer to "the average degree of correspondence between individual forecasts and observations in the verification data sample" (Katz and Murphy 1997, p. 31). However, the potential user of forecast information, for example a water allocation manager, may have an entirely different meaning for this term. Barrett (1998) argued that potential users of SCF information make decisions on the expected impact of climate variability on variables directly related to their operations. For example, a forecast of expected El Niño conditions for the upcoming cropping season has limited value without the ability to link this to potential effects on rainfall, streamflow, crop yields or prices, etc. Therefore, accuracy from a user's perspective is related to how well a decision that is made using SCF information meets users' expectations, i.e. how often desired outcomes are achieved. As Barrett (1998) pointed out, ENSO forecast information is only indirectly related to these outcomes. To ascertain reliably the value of SCF information, a methodology that addresses the issue of accuracy and its implications should be developed and adopted. This stance is supported by Hartmann et al. (2002) and Mjelde et al. (1993) who proposed that the usefulness of forecast information is a function of the context or intended use of the forecast system from the perspective of the user. It is only a short step to then conclude that accuracy is best measured by the ability of decision-makers to use forecast information to improve their achievement of outcome goals.

The results of this study allow some inferences to be made about forecast accuracy at the outcome level over the simulation period as opposed to a single year within the simulation period. Due to the lack of a defined relationship between expected water supply (including off-allocation) and the plant area decision, a systematic analysis of results was not undertaken to measure in each year whether the decisions made to increase or decrease access to off-allocation water supplies was correct. If these had been directly related, i.e. if upon announcement of the off-allocation cap in any year adjustment was able to be made to the planting decisions in that year, this type of analysis would have been justified. However, the variable access to off-allocation used in this study acts through making more water available in very wet years allowing off-allocation water to be substituted for on-allocation water subsequently leading to less crop stress in dry periods and making more water available in following years. Therefore the impact is spread over a number of years and as such a judgement on individual year accuracy cannot be made.

The link by Ritchie et al. (2004) to outcomes is one way to make inferences on forecast accuracy. This is via the results of the PFV = ∂' - Ω (7) at \$535,498; the VoF = χ - Ω (4) at \$353,740, and the finding of effectively no impact on environmental outcomes. In this case the value of the forecast is approximately 66 percent of the value of the perfect forecast. This suggests there is a level of accuracy, or effectiveness of meeting the goal of maximising RGM, because the result indicates returns above the baseline (without forecast scenario). However the level of accuracy in percentage terms is unable to be quantified.

A second factor that inhibits the calculation of accuracy levels is the fact that off-allocation access is determined based on other demands at a point in time, that is may be granted to water-licence holders' water when dams spill or high flows enter the river system and only when all other user needs (including environmental) have been met. Furthermore, despite off-allocation access being granted in any given year the ability for irrigators to actually take off-allocation water is constrained by factors such as pump size, on-farm storage size and the proportion of the on-farm storage available for water storage. In other words, if the on-farm storage is already full the off-allocation water will not be able to be captured.

The inference from these factors is that while continuing to develop forecasting techniques to improve the forecast accuracy in terms of the ability to forecast amounts of rainfall or streamflow are an important aspect, this should not be done in isolation of considering what outcomes the end-users of the information are seeking. Developing a highly accurate forecasting system without considering outcomes from its use is likely to limits its usefulness.

6.4.2. Demonstrating the potential of climate forecasts in case studies

Taking the accuracy issue into account, an obvious important step to improving adoption of SCF information would appear to be to prove its value in case studies. However, the proposition that demonstrating successful use of SCF in case study situations would lead to broad scale adoption of the information is not supported by evidence.

First, Pulwarty and Redmond (1997) found that studies examining the economic value of forecast information would make no difference in their use of this information for 25 percent of people interviewed. Secondly, many of the technical aspects of forecast, such as skill, vary spatially and temporally. Forecast "skill"[35] at one location may differ between seasons being perhaps stronger at forecasting summer streamflows than winter flows. Therefore for a particular location a seasonal forecast may be valid and useful for summer cropping decisions however the skill may be such that for winter the forecast offers no more valid information than the climate record. Additionally this "skill" varies between adjacent locations (Wernstedt

[35] Where skill is defined in line with Katz and Murphy (1997, p. 31) as " the accuracy of the forecasts of interest relative to the accuracy of forecasts produced by a naïve forecasting system such as climatology or persistence"

et al. 2002). Therefore, a successful case study at one location is not necessarily a pointer to a more widespread use, particularly where the information is used in specific decision-making. Nor is an unsuccessful case study necessarily a barrier to potential use in other locations.

Third, the communication barrier caused by a lack of understanding between forecasters and water managers in terms of forecast skill and manager need, also impacts on case study situations. Not only is there evidence to suggest that forecasters do not understand the needs or operating environments of water managers, but also users do not comprehend the potential and limitations of seasonal forecasts (Callahan et al. 1999). This has the potential to lead to non-adoption of SCF information when outcomes are not as expected.

Lastly, case studies that involve incorporation of forecast information into current management regimes may be constrained because current river management regimes, including off-allocation access rules, were devised prior to the potential for forecast information being recognised. Consequently, the opportunities for forecast information may be limited in current decision-making procedures (Hamlet et al. 2002). Potentially exacerbating this situation is the fact that current management regimes were also developed for objectives tied to human consumptive or safety goals, rather than today, where environmental goals are also important (Callahan et al. 1999).

The use of case studies to demonstrate successful application of SCF information in achieving desired outcomes remains a key component of improving the adoption of this information into decision-making despite these issues. The cautionary note is that successful application in a case study is unlikely to have, on its own, a major impact on adoption. Other factors such as those already raised, and institutional decision-making inflexibility, and communication between forecasters and potential forecast users, will need to be addressed as well.

Ray (2004) considered this issue from a different perspective and while finding similarly limited use of climate information in reservoir decision-making for consistently similar reasons to those identified above went what may be considered a step further by suggesting that climate information (including forecast) providers should spend more effort on understanding how climate variability interacts with the outcomes water resource managers are trying to achieve and then structuring information development to these needs. Ray proposed a user-assessment approach with the following steps to analyse the context of climate related problems:

i) *"Identify critical problems which are sensitive to climate*

ii) *Identify decision makers and their key stakeholders*

iii) *Assess how climate variability interacts with their critical problems; a decision calendar may help organize recurring decisions*

iv) *Identify user groups who may be willing partners in testing and prototyping this new technology.*

v) *Assess the products they currently use, the requirements for their mental and operational models, in order to determine how to make existing product lines useable"* (Ray 2004, p. 292)

An approach that follows these steps would be more likely to overcome the barriers identified as impeding adoption of seasonal climate information (Ray 2004).

The second aim of this research was to evaluate the efficacy of seasonal climate forecasting on environmental flow management in the Border Rivers catchment and was principally aimed at this second barrier to adoption and in doing so gaining an insight into the practicalities surrounding aspects of the first barrier. Not withstanding the lack of significance in results, the issues raised above do not lessen the importance of demonstrating forecast potential, rather they highlight that demonstrating in a singular context alone will be insufficient (although still useful) to increasing forecast use or providing pointers to where forecast information may be useful. In this case the forecast signal illustrated in Figure 2.18 has not translated into useful information in the context of the decision assessed. The strongest indications of the cause of this is shown in Figure 5.3 and Figure 5.7. The response of firstly the area planted and the water applied to a ten-fold increase in off-allocation availability is limited. Secondly, the insensivity of hydrologic indicators to increases in off-allocation access increases. The ten-fold increase in off-allocation availability resulting in a five percent increase in average annual planted area, and a six percent increase in irrigation water applied to the crop, is suggestive of other aspects of the water management system impinging on the usefulness of the forecast information. This is reiterated by the hydrologic indicators where changes are all less than 5 percent. As previously explained, these aspects include bottlenecks caused by a fixed area available for planting, fixed on-farm storage size and the ability to use other sources of water such as rainfall and on-allocation water to substitute for off-allocation.

An important principle in undertaking this research was to develop an appropriate methodology that will facilitate rapid evaluation of the use of forecast information in decision-making to allow a number of scenarios to be tested in an adaptive framework. The analytical framework developed, while having a number of shortcomings, does allow this rapid analysis using variables of interest to decision-makers which is then a precursor to undertaking a more detailed assessment of options that appear favourable. Just as importantly the analytical framework could be expeditiously employed in different locations to test the relationship between the seasonal climate forecasting tools and the economic and environmental outcomes. Consequently, an alternative decision or location which may have a different outcome could be assessed using this framework.

6.5. Conclusion

Examining the effect of increasing access to off-allocation water supply in a trade-off framework highlights the changing relationship between economic and environmental outcomes. In this chapter trade-off curves were developed for the economic outcome using regional gross margin and environmental outcomes using two hydrologic indicators, PFlowDur 0.02 and 0.5. For the trade-off curve RGM and PFlowDur 0.5 no competitive relationship was identified. Conversely, the trade-off curve for RGM and PFlowDur 0.02 showed some sign of a weak competitive relationship when viewed at a fine scale. These weak results are due to the lack of impact on the environmental indicators from changing access to off-allocation water supply as reported in Figure 5.7. An alternative analysis using the rule proposed by Jones et al. (2003), where if the key hydrological attributes are above two-thirds of natural then there is a high probability that the river is in a healthy working state, suggested that the economic – environmental outcome relationship is supplementary for the scenarios tested.

The economic and environmental results of the seasonal climate forecasting scenarios were overlayed on the RGM and PFlowDur 0.02 trade-off curve to test the hypothesis that the use of current seasonal forecasting tools to manage off-allocation water access by irrigators will either improve the regional gross margin of irrigated agriculture or produce conditions that will lead to improved riverine environmental health. For the majority of scenarios tested the hypothesis was not supported. However for three SCF scenarios ENSO, SOI <-15,>+15 and SOI <-20,>+20 results showed an increase in RGM without a negative impact on the environmental indicator. It is noted that the scale of the improvement is minor which highlights a level of uncertainty with the significance of the positive results. Furthermore, the conclusions are also based on the premise that no major ecological thresholds exist prior to the two-thirds of natural state as indicated by Jones et al. (2003).

The second hypothesis, assessed whether an improvement in forecasting accuracy will improve the regional gross margin and / or hydrologic indicator levels representing riverine environmental health to levels above that prevailing under the use of the forecast tools examined. Results showed that increasing the accuracy would increase RGM without negative impacts on the environmental outcomes. Again however the value of a proxy perfect forecast was relatively small indicating insufficient significance for the hypothesis to be supported.

The final part of this chapter considered the implications from the results in relation to the likelihood of SCF information being adopted into water resource management decision-making. The issue of accuracy is reported as being particularly important when considering adoptability of forecast information, although a number of researchers have found that accuracy is a misunderstood term. The finding of a positive value of forecast suggests that there is a level of accuracy where accuracy is measured in terms of outcomes that are expected to matter to the user. Apart from the relatively small value of the forecast, a number of other issues are identified in the literature that affect the adoptability of SCF information. These issues provide both blockages for this information being used by water resource management decision-makers and identify steps to be taken in future studies to improve adoptability.

7. Conclusions

7.1. Reviewing the study

The purpose of this study was to develop and test a methodology for evaluating the efficacy of a number of defined and commonly used seasonal climate forecasting tools when used in environmental flow management in the Border Rivers catchment. [36]The evaluation was undertaken following the ARMCANZ (1995, p. i) goal of achieving the "highest and best value of the limited resource for community benefit whilst ensuring that use of the resource is ecologically sustainable". The rationale for this study is that Australia's riverine environment is becoming increasingly degraded and in addition to environmental consequences, this degradation is likely to lead to a loss of agricultural productivity with subsequent social and economic adjustment costs. The National Land and Water Resources Audit reported that the degradation issues are widespread with 26 percent of Australia's river basins near or over sustainable usage levels (NLWRA cited in Schofield et al. (2003)).

Schofield et al. (2003) categorised the water reform process as one where irreversible environmental degradation is a key driver and the balancing of the outcomes of generally conflicting environmental and human consumptive use in order to retain the use of the water resources for future generations as the basic policy problem. ARMCANZ (1995) through its specification of the goal of water reform, i.e. "highest-valued use is defined as including economic returns from consumptive uses and the value to society from environmental and other non-consumptive water use", further defined the policy problem.

In considering the water reform problem, Schofield et al. (2003, p. 8) identified five broad socio-economic challenges of which two are related to this study:

"1. Improving water allocation and water-trading arrangements
2. Assessing the costs and benefits of increasing allocations to the environment".

These challenges form the basis of the research problem by identifying opportunities for the use of new information in water management decision-making. In this research seasonal climate forecasting (SCF) provides information on the hydrologic impact of climate variability which may assist in water management decision-making and may have potential to assist in meeting challenges 1 and 2. The ability to use this knowledge to forecast both rainfall and streamflow is becoming increasingly recognised (Abawi et al. 2001; Stone et al. 1992; Chiew et al. 2000). Interviews with 31 forecasters and water managers in the United States from twenty-eight organisations with wide ranging roles and responsibilities (including hydropower, environmental management, flood control and irrigation management) by Callahan et

[36] Commonly used in the sense that they are reported frequently in the media e.g. weekly on the ABC news, in newspapers such as the Queensland Country Life and on websites such as the Longpaddock, http://www.longpaddock.qld.gov.au/SeasonalClimateOutlook/OutlookMessage

al. (1999) concluded that while potential forecast utility was high, the information was rarely used for operational decision-making. Forecast information was used as background information. Similarly, the adoption of this knowledge into water resources management in Australia remains limited to use as qualitative background information (Callahan et al. 1999; Long and McMahon 1996).

In seeking to address the problem of whether SCF information may assist in achieving the policy goals identified by ARMCANZ two study aims were developed. Nested under these aims were two study hypotheses:

Aim 1: Develop an appropriate methodology and modelling framework for examining the impact of using SCF in environmental flow management on both the regional economy and the health of the riverine environment of the Border Rivers catchment (BRC).

Aim 2: Evaluate the efficacy of seasonal climate forecasting on environmental flow management in the BRC.

- *Hypothesis 1: The use of any of the three seasonal climate forecasting tools based on the El Niño-Southern Oscillation phenomenon to manage off-allocation water access by irrigators will increase the regional economic output of irrigated agriculture and/or produce conditions that will lead to improved riverine environmental health in the case study catchment.*

- *Hypothesis 2: An improvement in forecasting accuracy will increase the value of crop production and environmental flows to levels above those prevailing under the use of the identified forecast tools.*

In carrying out this research the method of examination was through a case study approach in the Border Rivers catchment straddling the Queensland - New South Wales border. The methodology involved undertaking a desktop modelling exercise to simulate the economic and environmental outcomes from making decisions on access to off-allocation water using SCF information. Specifically, the relationship between the SOI and streamflows was used to develop rules whereby access to off-allocation water supply for irrigators is restricted in years when the seasonal climate forecast indicates that streamflows are likely to be reduced and increased when the climate forecast indicates that streamflows are likely to be increased. That is, seasonal climate forecast information was incorporated into water resource management decision-making to identify periods when the taking of water for consumptive purposes would be expected to have a minimal impact or positive impact on environmental outcomes. This action would equate to an improvement in how water is allocated.

To improve adoption of the information into decision-making processes, an assessment of the costs and benefits of the application of alternative water management arrangements was seen as critical to the ability of governments to gain acceptance of, and implement, new arrangements. It was necessary therefore to develop a methodology to identify the scale and direction of trade-offs involved in policy alternatives while facilitating expeditious evaluation of the use of forecast information in decision-making. The trade-off analysis was undertaken because there is an increasing requirement for resource managers to meet multiple and often conflicting economic and ecological goals as identified by ARMCANZ. The information on the trade-offs from the analysis can then inform policy decision-making about these conflicting goals.

The analysis of results was carried out using the trade-off framework and the Pareto principle to study the trade-off between economic and environmental outcomes. The trade-off analysis adopted a threshold value approach where the economic outcomes were assessed as the impacts on regional gross margin for irrigated cotton in the BRC. Environmental outcomes were assessed through the use of hydrologic indicators which identify the change from the natural regime as a proxy for environmental outcomes. Together these outcomes were used to identify a threshold, which the environmental benefits (as measured by hydrologic indicators) should exceed for the policy change, in this case the use of SCF information, to be deemed acceptable.

In relation to hypothesis 1 the research supported the hypothesis finding that use of SCF information did lead to an improvement of regional gross margin without major impacts on environmental outcomes. The research also supported the second hypothesis finding that increased forecast accuracy led to an improvement in regional gross margin without major impacts on environmental outcomes. Despite these findings it was also acknowledged that in both cases regional gross margin improvements were small which has ramifications for the adoptability of SCF information. The process of adopting the SCF information into policy and decision-making will involve a number of transaction costs such as policy development from the water manager’s viewpoint and changing irrigation management practices for irrigators. For adoption to occur water managers and users would need to attain benefits that increase their profit above that attained in the baseline scenario and the transaction costs involved. Therefore it is likely that these findings would not overcome the barriers to adoption and the SCF information would not be implemented into water management operations.

7.2. Methodological implications

In undertaking the study a number of implications of the methodology, the modelling components and the assumptions made in regard to these, were identified. The trade-off analysis using the production possibility concept and two outcomes is an appropriate approach to develop the modelling. The research problem could be considered in an optimisation framework that addresses more than two outcomes, for example economic, environmental and social using a production manifold instead of a two dimensional frontier. Alternatively, more indicators for the two outcomes could have been chosen, for instance: change in regional gross margin and change in employment derived from input-output analysis as economic indicators; and mean annual flow and an indicator for high flow events, as environmental health indicators. However, this would have added unnecessary complexity.

The RGM does have limitations. It is an imprecise short-term measure (NSW Agriculture 2001) that does not take into account the ability of irrigators to adjust to changes in water supply regime. Over time irrigators will respond to changes in the water supply regime by making changes to their property layout, machinery, or their cropping types and cycle. The type and extent of this autonomous change is expected to be largely dependent on the circumstances of individual irrigators and no attempt to model this was undertaken in this research. Therefore the research is only assessing the short term impacts of changes to water management rules assuming consistent responses. The implications of this for the study conclusions are not significant because the impacts of the changed rules on RGM are relatively minor. It can be assumed that these minor impacts will not trigger significant adjustment by irrigators. If on the other hand the impact on RGM had been much greater, then the potential adjustment response by irrigators would need to be considered more closely.

The economic analysis does not explicitly consider the economic effects of the changes to water management rules on the remainder of the regional economy. In effect this research adopts an uncomplicated approach where an increase in RGM is expected to have positive flow-on effects to the rest of the regional economy, or alternatively, that a decrease in RGM will have negative flow-on effects. Again the scale of the results is important with this assumption. Should the impact on RGM be significant, and therefore the adjustment required by the irrigators also be significant, then this may have measurable flow-on impacts to the remainder of the regional economy.

One qualification should the impacts be large and positive, is that a potential expansion in irrigated output might divert resources, for example labour, from other sectors of the regional economy to the irrigated sector resulting in negative impacts on the source sectors. In the event that this occurs, an assessment of the extent of the flow-on impacts would be required to determine if the positive impacts on the irrigation sector are outweighed by the negative impacts on other sectors. Over time it might be expected that resources would transition into the regional economy from the wider economy mitigating the longer term negative impacts. It is acknowledged that this may be at the cost of other national sectors, but would be only of a very minor nature given the scale of the irrigation sector in the Border Rivers catchment compared to the national economy. The input-output analysis by CARE (2006) provides an example of how part of this assessment on the regional economy could be undertaken. The inability for input-output analysis to be used to determine impacts on social welfare as identified by Marsden Jacob Associates (2006) and Bennett (2000) remain valid, and would also need to be considered. For this study however the impacts on RGM are small in proportion to the baseline position and impacts on the regional economy are consequently expected to be minor.

The link between the natural flow paradigm and environmental flow statistics and environmental health is complex and in using the environmental flow statistics the study takes an approach (in line with the thrust of the methods used in the water resource plan) that allows a desktop analysis. As there is some uncertainty about the link between ecological condition and river flows it is important to bear in mind that the indicators used in this research represent proxies for ecological outcomes, not certainty that the outcomes have been achieved if they are met (Jones et al. 2003). This approach recognises that there is a need to make decisions on improving water management in the present and also acknowledges that over time estimating environmental impacts of changes to water management regimes will improve as more research is undertaken. The adoption of the two-thirds rule proposed by Jones et al. (2003) permits a simplified ranking of outcomes which is easily incorporated into the trade-off analysis.

It was reported in the results that tracking soil moisture depletion (SMD) and adjusting yields when the soil moisture level exceeded a threshold value that indicates the crop wilting point has a material impact on the results. Excluding the SMD rule did not change the pattern results for the SOI Phase and SOI value scenarios which did not exhibit positive values of information (VOI). Conversely, it did change the pattern of results for the ENSO and SOI <-15,>+15 and SOI <-20,>+20 scenarios. The results for these scenarios changed to the extent that the positive VOI exhibited in the analyses was eliminated, thereby eliminating the VOI for all of the scenarios based on SCF information.

This finding has potentially serious implications for studies using not only the IQQM but also other studies that discount the impact of timing of events on crop yields and therefore economic results. While the penalty levels used in this study lack empirical support and is somewhat of a blunt instrument, the approach tends to reflect reality. It is possible however that the approach adopted in this study overestimates the yield penalty and it is acknowledged that further research to fine tune the impacts is warranted. Approaches that embed the crop modelling in a more detailed manner than the version of IQQM used in this study within the modelling framework are likely to be in the best position to overcome the issue.

In adopting the case study approach it needs to be recognised that the results are restricted to the geographical region due to the characteristics of the regional response of rainfall and streamflow to the ENSO phenomenon. Studies by Chiew et al. (2000), Chiew et al. (1998), Chiew et al. (1994) and McBride and Nicholls (1983) have shown that a positive relationship exists between streamflow variability and the ENSO phenomenon. It has also been established that the strength of this relationship varies spatially and temporally. The implications for the findings of this research are that they are not necessarily transferable to similar types of situations in different locations. However the methodology could be adopted and test the same or different forecasting tools in other locations or on different parameters other than off-allocation.

7.3. Future research

Five opportunities for improving future research were identified during this study. Two opportunities surround the experiment itself, two relate to the production model and one relates to the environmental model.

The first opportunity relates to the proportion of the irrigation water supply impacted upon by this experiment. In chapter 5 it was reported that the volume of water used per water year on average increased by 6,858 ML or 6.5 percent when the potential access to off-allocation water increased greatly from 10 megalitres to 100,000 megalitres for each water year of the scenario period 1894 to 1994. Reasons for this included a substitution effect with on-allocation water supply and the existence of a number of bottlenecks in the system. These bottlenecks mask usefulness of the forecast of off-allocation water. One way to overcome this is to minimise the ability of the substitution effect by adopting water management rules where all irrigation water supply is subject to the forecast. In practice this is likely to have considerable effects on the outcomes for irrigators and may also lead to extensive capital adjustment by irrigators who may respond by increasing the level of on-farm storage. It would however give a more complete picture of the impact of SCF accuracy on the economic and environmental outcomes. An aim of this revision would also be to identify the point on the trade-off curve where the slope changes and the trade-off changes dramatically (i.e. identify point x on Figure 6.5) which would improve the ability to consider trade-off impacts.

The second factor that should be incorporated into future studies of this type is the impact of climate change. The use of a 100-year modelling timeframe where streamflow is determined by the rainfall records over the 100-year timeframe implicitly assumes that the streamflows in future years will be of a similar pattern and volume to the last 100 years. However, to the extent that climate change has already occurred this assumption is weakened. Regardless of this, the New South Wales Government (2007) and CSIRO (2007) research report into the impact of climate change in the Border Rivers catchment concluded that runoff is more likely to decrease than increase in the catchment. The scale of the changes reported in chapter 1 is dependant on the scenario being considered but the best estimate (median) climate 2030 scenario indicates a 9 percent reduction in annual runoff with water availability reduced by 10 percent and subsequently a reduction in end-of-system flows by 12 percent and total diversions by 2 percent. The impact of climate change on the irrigation industry in the Border Rivers is at present unclear as is the link between climate change and the ability to forecast streamflow on a seasonal basis. Nonetheless further research in this area should account for potential changes to the long term patterns and volumes of streamflow.

The first opportunity for improvement identified for the production model is the ability to integrate production and economic parameters into the hydrologic model. Linking the irrigation rules of the IQQM to yield calculation of a crop model would minimise the error associated with soil moisture depletion and yield calculation leading to more robust results. The use of hydrologic models in a policy planning framework requires the ability to estimate economic impacts because best practice in policy-making requires options to be ranked according to their net economic benefits (Office of Regulation Review 1998).

The second issue that might be studied as part of future research from an economic perspective is the relationship between seasonal climate forecasting and the outcomes when examined over shorter subsets of years. Table 5.4 and Table 5.5 identify that over the 100-year timeframe there are a number of 10-year assessment periods when the SCF based rule gives higher RGM outcomes than the baseline scenario. Conversely, there are a larger number of times when the SCF based rule outcomes are significantly less than the baseline. This result may represent an opportunity whereby if it were able to be identified what periods to use the SCF information and when not to, then overall outcomes may be improved.

The final opportunity identified for improving study outcomes for future research relates to the potential for better identification and measurement of environmental health outcomes and the ability to represent this in a single indicator. A key difficulty in undertaking the study was the use of a number of hydrologic indicators for determining impacts of scenarios on environmental health. Specifically, the varying results of the indicators where one suggests slight decline in environmental health whilst another is contradictory. The inability to aggregate these indicators into a single measure, and to identify thresholds where significant changes in environmental health may occur, limits the usefulness of the trade-off analysis. It is possible that the lack of consistent results for the hydrologic indicators is due to the small absolute variation in the levels of water used. The trend shown in Figure 6.9 where all the indicators demonstrating decreasing environmental health is supportive of this conclusion. Therefore research to aggregate various indicators of environmental health into a single indicator that facilitates identification of possible thresholds would make the trade-off analysis more useful. The use of the two-thirds rule of Jones et al. (2003) is one method of overcoming this weakness.

7.4. Contributions to the body of knowledge

This research will lead to the discounting of the use of SCF information in setting access to off-allocation water in the BRC within the current management framework for the time being. The results of this study show that while there is potentially an increase in economic returns with little/no impact on environmental health, the change in economic return is unlikely to be sufficiently significant to provide confidence and warrant a change at this stage.

The results suggest the need for a more robust but useful method of measuring environmental benefits as part of the trade-off analysis. While the study did not show strong impacts (negative or positive) on environmental indicators it became clear in implementing the methodology that it is difficult to use a range of indicators, that can show conflicting results, as the measure of environmental outcomes.

A key contribution of this research is the furtherance of a multi-disciplinary approach to managing natural resources. The approach used was to distil the detailed analysis that can be undertaken by the economic and environmental disciplines and incorporate them into a single straightforward analysis. An aim of this approach is to take on board the findings of other researchers into why SCF information is not adopted by potential users and develop a method to overcome some of the identified barriers to adoption.

The use of the trade-off curve and Pareto principle to examine the changes of economic and environmental variables simultaneously and identify the relationship between the two shows useful potential from two angles. The first relates to the actual use of the trade-off analysis and the Pareto principle in water resources management where stakeholder participation and acceptance of changes to water management regimes is becoming more important. Examples of this approach in water resource management was not located in the literature indicating that is not commonly used. The trade-off analysis shown is a useful means of summarising complex results in an uncomplicated and visually appealing manner that is able to be understood by the non-scientific community. Consequently, the use of a trade-off analysis may be a useful tool to assist in obtaining stakeholders' support for issues whereby the basic problem involves a trade-off in outcomes.

The second angle stems from the method being a means to integrate information from different scientific disciplines, but also for analysing proposed policy and management changes. In particular the attempt to assess the change based on the slope of the curve is potentially useful information to water policy-makers and is highlighted for further development. This approach allows the integration of ecological thresholds into the analysis and its development would add descriptive power to policy making particularly where more often there are legislative requirements to balance a range of often competing outcomes when managing natural resources.

7.5. Concluding comments

The aims of this research were two fold. Firstly, to the development of a methodology and modelling framework for examining the impact of using SCF information on economic and environmental outcomes was undertaken. Secondly, the efficacy of seasonal climate forecasting on environmental flow management in the Border Rivers catchment was tested.

The results indicate a greater relative shift in economic outcomes than environmental indicators in response to variations in access to off-allocation water supplies. In comparison to the ten-fold increase in water supply in some scenarios, the economic response remains relatively small. The results shown in Figure 6.10 and Figure 6.11 when considered in relation to Figure 6.14 can be interpreted as showing that the SCF scenarios: ENSO, SOI <-15,>+15 and SOI <-20,>+20 are to the right of the baseline, but to the left of the point at which the relationship between the economic and environmental outcomes change such that environmental outcomes worsen more rapidly as economic outcomes increase. These results point to some level of efficacy in terms of value of information as they result in higher economic benefits without degrading the environmental benefits to the point at which environmental health is compromised. Therefore the findings would tend towards supporting hypothesis 1, showing the use of current seasonal forecasting tools to manage off-allocation water access by irrigators did improve the regional gross margin of irrigated agriculture without negative impacts on environmental health. However, while there are no hard and fast rules about the size of impact needed to ensure adoption into water resource management decision-making, a 1.8 percent improvement in RGM is unlikely to be sufficient.

Related to this is the low value of perfect information and the large proportion of it explained by the scenarios with a positive VOI. Again this is due to the low increase in water use (1.3 percent over baseline). Similarly to the first hypothesis, it was found that an increase in forecast accuracy would improve regional gross margin without negatively impacting upon the environmental health of the riverine environment. The overall size of the value of the perfect forecast nonetheless remains small in comparison to the regional gross margin attained under the baseline scenario. A comparison of the PFV at $524,005 with the RGM attained for the baseline scenario of $30.46 million highlights that the improvement is only 1.7 percent.

It may be concluded that a very weak competitive or potentially a supplementary relationship exists between the economic and environmental outcomes (Figure 6.7) at least over the range of outcomes studied. However, this relationship was not shown consistently across the hydrologic indicators used when comparing Figure 6.7 and Figure 6.8. The difficulties in this component of the study are strongly related to the ability for irrigators to substitute on-allocation water for off-allocation water when access to it is restricted and the existence of bottlenecks limiting the actual take of off-allocation water.

It is clear that the environmental results indicate that while the aim might be simple i.e. improved environmental health, implementing a method in a multi-disciplinary study analysing the two outcome types is complex and it may be that some aspect are improved whilst others are made worse.

An important outcome implied by the first aim of the study is to develop an impact assessment methodology that would facilitate the adoption of SCF information into the decision-making procedures for managing water resources. Considering the four commonly identified barriers to adoption, this study did not provide sufficiently robust results that are likely to lead to adoption. In particular, the following barriers are substantial:

1. technical e.g. accuracy, relevance, understandability, timing; and,
2. lack of (successful) demonstrated use.

Accuracy, a commonly misunderstood term, may be more a function of demonstrated use than a technical assessment of skill of the seasonal climate forecast system. This is due to the importance that decision-makers place on the outcomes of decisions made using SCF information. In a systems modelling environment such as this study, particularly when the variable of interest is not causally linked with explicit decisions in the year of the forecast being made, an assessment of accuracy is difficult. When assessed in terms of the overall outcome in relation to the value of a perfect forecast, thereby demonstrating successful use, the low value of information implies little is to be gained from use of the SCF information in the context of this case study.

8. References

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9. Appendices

Appendix A Characteristics of expansionary and mature water economies

Item	Expansionary phase	Mature phase
Long run supply of impounded water	Elastic	Inelastic
Demand for delivered water	Low, but growing: elastic at low prices, inelastic at high prices	High, and growing: elastic at low prices, inelastic at high prices
Physical condition of impoundment and delivery system	Most is fairly new and in good condition	A substantial proportion is aging and in need of expensive repair and renovation
Competition for water among agricultural, industrial and urban uses and in-stream flow maintenance	Minimal	Intense
Externality etc. problems	Minimal	Pressing: rising water tables, salinisation, saline return flows, groundwater salinisation, water pollution etc.
Social costs of subsidising increased water use	Fairly low	High and rising

(Source: Quiggin 2001)

Appendix B National Principles for the Provision of Water for Ecosystems

GOAL

The goal for providing water for the environment is to sustain and where necessary restore ecological processes and biodiversity of water dependent ecosystems.

PRINCIPLE 1 River regulation and/or consumptive use should be recognised as potentially impacting on ecological values.

PRINCIPLE 2 Provision of water for ecosystems should be on the basis of the best scientific information available on the water regimes necessary to sustain the ecological values of water dependent ecosystems.

PRINCIPLE 3 Environmental water provisions should be legally recognised.

PRINCIPLE 4 In systems where there are existing users, provision of water for ecosystems should go as far as possible to meet the water regime necessary to sustain the ecological values of aquatic ecosystems whilst recognising the existing rights of other water users.

PRINCIPLE 5 Where environmental water requirements cannot be met due to existing uses, action (including reallocation) should be taken to meet environmental needs.

PRINCIPLE 6 Further allocation of water for any use should only be on the basis that natural ecological processes and biodiversity are sustained (i.e. ecological values are sustained).

PRINCIPLE 7 Accountabilities in all aspects of management of environmental water provisions should be transparent and clearly defined.

PRINCIPLE 8 Environmental water provisions should be responsive to monitoring and improvements in understanding of environmental water requirements.

PRINCIPLE 9 All water uses should he managed in a manner which recognises ecological values.

PRINCIPLE 10 Appropriate demand management and water pricing strategies should be used to assist in sustaining ecological values of water resources.

PRINCIPLE 11 Strategic and applied research to improve understanding of environmental water requirements is essential.

PRINCIPLE 12 All relevant environmental, social and economic stakeholders will be involved in water allocation planning and decision-making on environmental water provisions.

(Source: ANZECC 1996)

Appendix C Listing and definition of Key Flow Statistics

Table C.1. Attributes, threats, environmental flow requirements and hydrological indicators.

System Level Attribute	Key Threats	Environmental Flow Requirements (EFR's)	Hydrological Indicator	Code
Flow Volume	Reduced flow volume	Increase flow volume in the river channel and across the floodplain	Median annual flow (GL/year)	MAF
			Total volume of flow > channel capacity (GL)	FGC
			Spell Analysis* - Average time above significant floodplain inundation threshold (months/year)	ATS
Flow Distribution	High summer flows	Reduce summer flows in the Upper Murray	Median Summer flow (Nov-Marsh) flow (GL/m)	MSF
	Loss of flood flow sequence (small to medium floods)	Ensure flood flows are followed by a flow of similar magnitude at an interval promoted towards natural	Spell analysis* -Median event interval (commence to flow)	FIC
			Spell analysis* - Median event interval (significant floodplain inundation)	FIS
Flow Variability	Reduced flow range	Increase range of flows on a	SRA Seasonal Amplitude index	SAM

System Level Attribute	Key Threats	Environmental Flow Requirements (EFR's)	Hydrological Indicator	Code
		seasonal basis		
	Constant flows	Avoid unnaturally prolonged periods of constant river height	75th percentile of daily change in river level (cm/d), November to February	SDC
	Unnatural rates of change in river height	The rate of change of the rising and falling limbs of the hydrograph should remain within the natural range	Not subject to modelling using the Monthly Simulation Model (MSM)- no indicator to assess	-
Connectivity	Barriers to in-channel fish movement	Enhance opportunities for weir drown-out	Weir drown out (percent years) Lock 1 drowned out, September – Marsh Weir 32 drowned out, August - November	WDO
	Reduced floodplain inundation	Promote towards natural the frequency and duration of floodplain inundation	Spell Analysis* - Median event duration (commence to flow)	MDC
			Spell Analysis* - Frequency of events above commence to flow threshold	FRC
			Spell Analysis* - Frequency of events above significant floodplain	FRS

System Level Attribute	Key Threats	Environmental Flow Requirements (EFR's)	Hydrological Indicator	Code
			inundation threshold	
			Spell Analysis* - Median event duration (significant floodplain inundation)	MDS
Flow Related Water Quality	Cold water releases from large dams	Ensure downstream water temperature in within natural seasonal range and changes at close to natural rates	Downstream Temperature. Not subject to modelling using the Murray Simulation Models – no indicator to assess	-
	Reduced in-stream productivity due to high summer turbidity	More natural proportion of Darling River discharge to the Murray during November-Marsh	percent Darling water of total at Lock 10 (Average: November – February)	PDA
	Unnatural salinisation	Maximise river flows for salt dilution purposes, within the natural range	Salinity (average level in EC at Morgan)	ECM
	Increased frequency of toxic cyanobacterial blooms	Reduce weir pool residence times to less than 10 days	percent years Lock 3 < 4,000 ML/d November-April (Moderate security threshold)	ABM

(Source: Jones et al. 2003)

Table C.2. Key flow statistics chosen for WAMPs chosen by Technical Advisory Panels (TAPs) for the Condamine Balonne, Fitzroy, Logan and Barron WRPs.

Flow statistic	**Condamine - Balonne**	**Fitsroy**	**Logan**	**Barron**
General				
Median annual flow (percent of natural)	√	√		
Mean annual flow (percent of natural)		√		√
Annual proportion of flow deviation	√	√		
Variability				
Proportion of natural monthly flow variability	√			
Flow regime class (Flow seasonality)			√	√
Inter-annual variability			√	
Ratio of 10 percent exceedance flow to 90 percent exceedance flow for each month of the year			√	
In-channel and low flow				
Frequency of half bank-full flows (percent of natural)		√		
Frequency of bank-full flows (percent of natural)		√		
80 percent daily exceedance flow for each month (percent of natural)			√	√
50 percent daily exceedance flow for each month (percent of natural)			√	
5 percent of daily exceedance flow for each month (percent of natural)			√	
Proportion of natural 'low flow' event frequency	√		√	√
Proportion of 'no flow' event frequency	√			
Flow duration percentile for zero flow			√	√
High flows and floodplain				
Proportion of natural 'high flow' event frequency	√			
1.5 year average return interval (ARI) (percent of natural)		√	√	√
5 year ARI (percent of natural)			√	√
10 year ARI (percent of natural)				√
20 year ARI (percent of natural)			√	√
Volume of the first post-winter flow event (percent of natural)		√		
Frequency of floodplain inundation (percent of natural)		√		
Proportion of natural storage level duration for the Narran Lake system	√			
Estuarine productivity		√	√	
Direct infrastructure effects				
Proportion of river inundated by dams and weirs	√	√		
Frequency of weir drown-outs		√		

(Source: Whittington 2000)

Table C.3. Border Rivers Water Resource Plan Performance Indicators

The following definitions of key flow statistics were sourced from the Border Rivers Water Resource Plan (DNRM 2003)

Flow indicator	**Definition**
End of system flow	the volume of water from the plan area that crosses the border from the State into New South Wales at node A and to the west of node A in the simulation period
Low flow	the total number of days in the simulation period in which the daily flow is not more than half the pre-development median daily flow
Summer flow	the average number of summer flow days in the simulation period
Beneficial flooding flow	the median of the wet season 90-day flows for the years in the simulation period
1 in 2 year flood	the daily flow that has a 50 percent probability of being reached at least once a year

Appendix D Specifications for the OZCOT model

The OZCOT model was used to simulate cotton yield in each season (1891-1995) for a range of water supply options per hectare. The meteorological data used in this model were those from Goondiwindi Post Office. The model was run for Goondiwindi by Dr. Mike Bange of Plant Industry, CSIRO, Narrabri (Personal communication). The major assumptions made in this model are:

- Location: Goondiwindi (Latitude -28.00, longitude 149.75)
- Crop: Cotton
    - Variety S189
    - Row spacing 1.0 m
    - 12 plants per m row
    - Sowing depth 5.0 cm
    - Solid plant
    - Soil water holding capacity: 300mm (Starting soil water: 100mm)
- Sowing date: 1st October
- OZCOT decides when to irrigate.
    - Pre-irrigate 12 days before sowing.
    - Crop irrigated at 90 mm deficit
    - Allocation: 0 to 12ML/ha, 0.1 ML per ha increment
    - Efficiency 100 percent
    - Timing of first irrigation: 14 days after 1st square
    - Timing of last irrigation: 20 percent bolls open
- Soil nitrogen 100 kg/ha
- Fertilizer N 200 kg/ha applied on Julian day 250.
- Soil water reset on Julian day 261.

An example of the yield data for a water supply option of 0 to 5 ML/ha with an increment of 0.5ML/ha over the period 1895-1915 is shown in Table D.1. The simulated water availability from IQQM and cotton yield data from OZCOT was linked to carry out an economic analysis (chapter 6) to determine the value of forecasting water supplies.

Table D.1 Simulated cotton yield data for different water supply options

Year	Water (ML/ha)																								
	0	**0.5**	**1**	**1.5**	**2**	**2.5**	**3**	**3.5**	**4**	**4.5**	**5**	**5.5**	**6**	**6.5**	**7**	**7.5**	**8**	**8.5**	**9**	**9.5**	**10**	**10.5**	**11**	**11.5**	**12**
	Yield (bales/ha)																								
1895	2.0	4.4	3.9	4.2	4.2	8.6	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8
1896	3.4	3.2	3.5	3.4	3.4	4.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2
1897	3.8	3.6	3.9	3.6	6.5	6.5	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8	9.8
1898	3.9	3.6	3.8	3.8	4.9	7.3	7.3	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6	10.6
1899	0.1	0.1	3.3	3.1	3.1	5.7	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3
1900	0.0	0.0	3.5	3.5	6.2	6.2	8.0	8.0	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1	10.1
1901	0.4	0.0	2.9	2.7	2.7	5.3	6.9	6.9	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1
1902	0.9	1.0	2.6	2.9	2.9	5.1	5.1	7.7	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2	9.2
1903	2.0	2.0	3.1	3.0	3.0	5.3	7.2	7.2	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1
1904	2.1	5.1	4.1	4.9	4.9	6.8	6.8	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7	11.7
1905	1.6	2.6	4.3	4.8	4.8	8.0	8.0	7.2	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5	9.5
1906	2.6	3.3	3.1	3.4	3.4	6.2	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0
1907	7.2	7.1	7.3	7.0	7.0	9.3	9.3	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1	11.1
1908	1.6	3.7	3.0	3.5	6.6	6.6	11.1	11.1	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9
1909	1.2	1.1	2.2	2.2	4.4	4.4	6.8	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9
1910	6.6	3.3	6.5	6.5	6.6	6.6	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0
1911	10.6	10.9	10.4	10.4	10.4	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9	4.9
1912	0.3	0.7	3.4	3.3	3.3	5.8	5.8	8.0	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9	9.9
1913	0.6	2.4	2.4	3.0	5.8	5.8	8.5	8.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5	11.5
1914	0.6	0.9	3.4	4.0	4.0	5.7	5.7	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4	7.4
1915	3.6	3.7	3.7	3.7	4.9	4.9	6.7	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3

(Source Hearn 1994) (The model was run by Dr. Mike Bange of Plant Industry, CSIRO, Narrabri)

Appendix E Calculating Yield Using the Water Use Efficiency Concept

ADVICE

GRAINS RESEARCH

Research Information for Farm Advisers

From the Grains Research and Development Corporation (GRDC)

Research Update

Water Use Efficiency - a tool for bench marking production

July 1998

GRDC

Grains Research & Development Corporation

Rain - to - Grain

Water use efficiency - The yardstick to yields

Water use efficiency (WUE) is a robust concept, useful for yield targeting, bench marking paddock and farm performance and comparing management options. Comparing WUE data within groups of farmers can assist understanding of farming systems and help identify areas for potential improvement.

Water supply is a common limiting factor in cereal production. It is this common limitation that allows WUE to work over a wide range of conditions. While the timing of rainfall can be all-important, WUE can explain as much as 60% of yield variation. This fact makes WUE a quick and simple calculation to check how well we are converting rain to grain. WUE also provides a guide to future crop expectations.

WUE, a framework for considering yield expectations and planning inputs

WUE (kg/ha/mm) = Crop yield (kg/ha) / Water supply (mm)

WUE relies on estimating the amount of water available to the crop. This is the sum of water stored during the fallow, plus growing season rainfall, less an allowance for in-crop water loss.

Variance in WUE between crops, years, fields or farms can serve as a flag to prompt the questions "Why the difference?" and "Can we influence or allow for it?".

Water supply

Water supply (mm) = Stored water (mm) + Growing season rainfall (mm) - Water loss factor (mm)

The likely water supply for a crop can be estimated as: stored water plus growing season rain, which can be measured or estimated. Rainfall records provide a source of historical water supply data for past crops.

The water loss factor in the water supply equation, is an allowance for in-crop evaporation and runoff, water left after harvest, and water used in building the crop before grain production. A first estimate of the water loss factor is 100 mm, although with experience this can be adjusted.

Storing soil water in a fallow

Fallow efficiency varies with the pattern of rainfall received, soil type, slope, soil water profile, infiltration rate, fallow management, temperature, evaporation, stubble cover and tillage. In the northern cereal belt,around 40% of rain comes in falls of less than 15 mm. These small falls tend to evaporate without making a significant contribution to stored soil water. This limits how much fallow efficiency can be improved.

When calculating water supply, only a percentage of fallow rainfall is thus included. 20% is typical. Seasonal conditions and management skill will raise or lower this figure.

A fallow efficiency of 25% is common where high levels of stubble are maintained. Drier hotter areas typically have poorer fallow efficiencies than cooler wetter areas.

Runoff on sloping country is reduced by maintaining stubble and reduced or zero tillage. Typical runoff on bare fallow is around 10% of rainfall. This can be reduced to around 5% by maintaining stubble. Tighter crop sequences will increase fallow efficiency and open cracks will allow capture of storm rainfall.

Effective management of fallow weeds is also critical to fallow efficiency.

Yield potential

Yield potential (kg/ha) =
Water supply (mm) x WUE (kg/ha mm)

WUE is greatest when the fallow captures and stores rain efficiently and the crop is able to efficiently convert stored water into yield.

Fallow efficiency is improved by timliness of weed management, maintainence of cracks early in the fallow, stubble retention and reduced cultivation Converting water to yield requires good agronomic management.

What is extra stored water worth?

Stored water is worth money as crop yield, but how is its value estimated? The following example shows an increase in gross margin of 20% resulting from a 5% improvement in fallow efficiency.

Fallow water storage increases from 20% to 25% of fallow rainfall

Base treatment: WUE of 9 kg/ha/mm, fallow rain = 440 mm, growing season rain = 240 mm, variable costs = $132/ha and wheat price = $120/t.

Gross margin @ 20% fallow efficiency ={((440 x **0.20** + 240) -100) x 9 x $120/1000} - $132 = $114/ha.

With improved fallow management fallow efficiency can increase to 25%. This is typical for stubble mulch or no-till fallows;

Gross margin @ 25% fallow efficiency = {((440 x **0.25** + 240) -100) x 9 x $120/1000} - $132 = $138/ha

This marginal analysis indicates that improving water storage by only 5% (22 mm) increased the gross margin by 21% or $24/ha.

Improving WUE and profit

WUE can be improved by timely planting, matching nutrition to yield potential and sound rotational and varietal management. Increasing WUE increases crop yield and profit per unit of water. As a 15% increase in yield will approximately double profit, the potential rewards from increasing WUE are large. In the following example the WUE is increased from 9 kg/ha/mm as used above, to 12 kg/ha/mm. Base treatments and costs are as above. At a WUE of 12 kg/ha/mm, gross margin increases to $228/ha, an increase of 65% over a WUE of 9.

Gross margin @ 25% fallow efficiency and WUE of 12 = {((440 x **0.25** + 240) -100) x 12 x $120/1000} - $132 = $228/ha

Ideas for grower groups

Use work sheets to calculate WUE's and their impact on profit. Discuss individual fields and why there is variance in WUE and potential ways to lift water storage or the WUE. Benchmark fields, crops or farms by graphing data from real life situations. Use work sheets to compare and evaluate management options and estimate financial impact.

Table 1: Typical water use efficiencies (WUE) for a range of crops. (Source QDPI)

Crop	Range	Water use efficiency (kg/ha/mm) Better managed crops
Wheat	5-20	10-15
Barley	5-20	10-15
Chickpea	3-10	8-10
Sorghum	5-20	10-15
Sunflower	2-8	7
Mungbean	2-6	
Cotton		around 0.0125 bales/ha/mm

For further information and copies of work sheets to use with growers, contact: Dr David Freebairn QDNR / APSRU Toowoomba Ph: 0746 881 200 or Jan Edwards Ag NSW Tamworth Ph: 0267 631 100

NATURAL RESOURCES Queensland Government

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NSW Agriculture

Example of a work sheet used with growers

A group of growers can compare their farms or paddocks, using WUE to remove some of the difference due to rainfall between seasons and properties. It is also useful to plot data graphically.

Paddock/ property name	Fallow rainfall (mm)	Fallow water storage (mm) (A x 0.2)*	Growing season rainfall (mm)	Water supply (mm) less water loss factor of 100mm B + C - 100	Yield (kg/ha)	WUE (kg/ha/mm)
	A	B	C	D	E	E / D
Example	250	250 * 0.2 = 50	175	225 - 100 = 125	1500	1500/125 = 12

** Fallow efficiency of 20% or 0.2*

Appendix F Production model data

Surface Irrigated Cotton

Northern Zone
Summer 2002-2003

Gross Margin Budget

Income		
Lint -	6.75 bales/ha at $480.00 /bale (at gin)	3,240.00
Seed -	2.43 tonnes/ha at $230.00 /tonne (at gin)	558.90
	A. Total Income $/ha	**$3,798.90**
Variable Costs		
	Cultivation	199.05
	Sowing	59.42
	Crop insurance	55.00
	Fertiliser and application	107.73
	Herbicide and application	197.55
	Insecticide and application	613.33
	Irrigation	62.84
	Contract harvesting	439.85
	Cartage to gin	66.51
	Ginning charges	405.00
	ACF and research levy	28.69
	Other	45.00
	B. Total Variable Costs $/ha	**$2,279.97**
	C. Gross Margin (A-B) $/ha	**$1,518.93**

Source: NSW Agriculture (2003b)

Surface Irrigated Sorghum

Northern Zone
Summer 2003-2004

Gross Margin Budget

Income		
	8.00 tonnes/ha at $140.00 /tonne (on farm)	1,120.00
	A. Total Income $/ha	**$1,120.00**
Variable Costs		
	Cultivation	60.22
	Sowing	78.84
	Fertiliser	192.92
	Herbicide	67.44
	Insecticide	30.69
	Irrigation	144.09
	Harvest	120.09
	Levies	23.37
	B. Total Variable Costs $/ha	**$717.66**
	C. Gross Margin (A-B) $/ha	**$402.34**

Source: NSW Agriculture (2003c)

Flood Irrigated Wheat

Northern Zone
Winter 2003

Gross Margin Budget

Income		
	5.50 tonnes/ha at $175.00 /tonne (on farm)	962.50
	A. Total Income $/ha	**$962.50**
Variable Costs		
	Cultivation	14.62
	Sowing	64.67
	Fertiliser	178.53
	Herbicide	36.98
	Insecticide	0.00
	Irrigation	47.97
	Contract Harvesting	85.22
	Levies	9.77
	Insurance	19.73
	B. Total Variable Costs $/ha	**$457.49**
	C. Gross Margin (A-B) $/ha	**$505.01**

Source: NSW Agriculture (2003a)

Appendix G Monthly SOI phase values used in this study

Year	May	June	July	August	September	October
1894	NZ	NZ	NZ	NZ	CP	CP
1895	NZ	NZ	NZ	NZ	NZ	NZ
1896	RF	CN	CN	CN	CN	CN
1897	CN	RR	NZ	NZ	NZ	NZ
1898	RF	NZ	RR	NZ	NZ	NZ
1899	RF	NZ	CN	NZ	RR	RR
1900	CN	RR	CP	CP	RF	CN
1901	NZ	RR	CP	RR	RF	CN
1902	CP	CP	NZ	RF	RF	CN
1903	CP	RF	RR	NZ	RR	CP
1904	CP	RF	NZ	RR	NZ	NZ
1905	CN	CN	CN	CN	NZ	NZ
1906	RR	RF	RR	RR	CP	CP
1907	NZ	NZ	NZ	NZ	NZ	NZ
1908	NZ	NZ	NZ	RR	RR	CP
1909	RR	RR	CP	CP	CP	NZ
1910	NZ	RR	CP	CP	CP	CP
1911	RF	CN	CN	CN	CN	CN
1912	CN	CN	RR	NZ	NZ	NZ
1913	NZ	NZ	NZ	NZ	NZ	CN
1914	CN	CN	CN	CN	CN	CN
1915	RR	NZ	NZ	CP	NZ	RF
1916	RR	CP	RR	CP	CP	CP
1917	CP	CP	CP	CP	CP	CP
1918	CP	RF	RF	CN	NZ	NZ
1919	NZ	NZ	CN	CN	NZ	NZ
1920	NZ	RR	CP	CP	CP	RF
1921	CP	CP	CP	RF	RR	CP
1922	NZ	RR	NZ	NZ	RR	CP
1923	CP	NZ	RF	CN	CN	CN
1924	RR	CP	CP	CP	CP	CP
1925	RF	NZ	RF	CN	CN	RF
1926	NZ	NZ	RR	NZ	RR	NZ
1927	CP	NZ	NZ	NZ	NZ	NZ
1928	RF	NZ	RR	RR	CP	CP
1929	RF	RR	NZ	NZ	NZ	RR
1930	RR	RF	NZ	NZ	NZ	RR
1931	CP	CP	CP	NZ	NZ	NZ
1932	NZ	NZ	NZ	NZ	RF	CN
1933	NZ	RF	RR	NZ	NZ	NZ
1934	RF	RR	CP	RF	CN	RR
1935	RF	NZ	NZ	NZ	RR	CP
1936	CP	NZ	RR	RF	RR	NZ
1937	NZ	NZ	RF	RR	NZ	NZ
1938	RR	CP	CP	CP	CP	CP
1939	NZ	NZ	RR	NZ	RF	CN
1940	CN	CN	CN	CN	CN	CN
1941	CN	RF	CN	CN	CN	RF
1942	RR	CP	NZ	RR	CP	CP
1943	CP	RF	RR	RR	CP	CP

Year	May	June	July	August	September	October
1944	NZ	NZ	NZ	RR	NZ	RF
1945	RR	RR	CP	RR	CP	CP
1946	CN	CN	CN	CN	RF	CN
1947	RF	RR	RR	CP	CP	RF
1948	NZ	RF	RR	NZ	NZ	RR
1949	NZ	RF	RR	NZ	RR	NZ
1950	CP	RR	CP	CP	CP	RR
1951	CN	RR	RF	CN	RF	CN
1952	RR	CP	CP	NZ	NZ	RR
1953	RF	RR	NZ	RF	CN	RR
1954	NZ	NZ	RR	RR	CP	NZ
1955	RR	CP	CP	CP	CP	CP
1956	CP	CP	CP	CP	CP	RR
1957	RF	RR	NZ	RF	CN	RR
1958	RF	RR	NZ	CP	RF	NZ
1959	NZ	RF	NZ	NZ	NZ	NZ
1960	CP	NZ	RR	CP	CP	NZ
1961	CP	NZ	NZ	NZ	NZ	NZ
1962	RR	CP	NZ	RR	CP	CP
1963	CP	RF	RR	NZ	NZ	RF
1964	RR	CP	CP	RR	CP	CP
1965	RR	RF	RF	CN	CN	CN
1966	NZ	RR	NZ	RR	NZ	NZ
1967	NZ	RR	NZ	RR	CP	NZ
1968	RR	CP	CP	NZ	NZ	NZ
1969	CN	NZ	NZ	NZ	RF	CN
1970	RR	RR	RF	RR	RR	CP
1971	CP	CP	NZ	RR	CP	CP
1972	RF	CN	CN	CN	CN	CN
1973	RR	RR	CP	CP	CP	CP
1974	CP	CP	RR	CP	CP	CP
1975	CP	CP	CP	CP	CP	CP
1976	NZ	NZ	RF	CN	CN	RR
1977	CN	CN	CN	CN	CN	CN
1978	RR	CP	NZ	NZ	NZ	NZ
1979	RR	NZ	RR	RF	RR	NZ
1980	RR	NZ	NZ	NZ	NZ	NZ
1981	RR	CP	CP	CP	CP	RF
1982	NZ	RF	CN	CN	CN	CN
1983	RR	RF	NZ	RR	RR	CP
1984	NZ	RF	RR	NZ	NZ	NZ
1985	CP	RF	RR	RR	NZ	NZ
1986	NZ	RR	CP	RF	NZ	RR
1987	CN	CN	CN	CN	CN	CN
1988	RR	RF	RR	CP	CP	CP
1989	CP	CP	CP	RF	RR	CP
1990	RR	RF	NZ	NZ	CN	NZ
1991	CN	NZ	NZ	RF	RF	CN
1992	RR	RF	CN	NZ	NZ	RF
1993	CN	CN	CN	CN	CN	CN
1994	CN	CN	CN	CN	CN	CN

Appendix H Average SOI value for the period August to September

Year	Average SOI: Aug to Sep	SOI <-5,>+5	SOI <-7,>+7	SOI <-10,>+10	SOI <-15,>+15	SOI <-20,>+20
1894	6.7	3	2	2	2	2
1895	-3.6	2	2	2	2	2
1896	-5.1	1	2	2	2	2
1897	-20.7	1	1	1	1	1
1898	0.5	2	2	2	2	2
1899	2.6	2	2	2	2	2
1900	-5.8	1	2	2	2	2
1901	-4.4	2	2	2	2	2
1902	-3.1	2	2	2	2	2
1903	-13.3	1	1	1	2	2
1904	4.4	2	2	2	2	2
1905	0.5	2	2	2	2	2
1906	-7.3	1	1	2	2	2
1907	16.9	3	3	3	3	2
1908	-4	2	2	2	2	2
1909	11.5	3	3	3	2	2
1910	5.3	3	2	2	2	2
1911	12.5	3	3	3	2	2
1912	-10.4	1	1	1	2	2
1913	-5.8	1	2	2	2	2
1914	-8.5	1	1	2	2	2
1915	-14.8	1	1	1	2	2
1916	7.3	3	3	2	2	2
1917	10.3	3	3	3	2	2
1918	32.2	3	3	3	3	3
1919	-6.3	1	2	2	2	2
1920	-6.3	1	2	2	2	2
1921	5.2	3	2	2	2	2
1922	-0.9	2	2	2	2	2
1923	1.9	2	2	2	2	2
1924	-16.6	1	1	1	1	2
1925	9.2	3	3	2	2	2
1926	-8.6	1	1	2	2	2
1927	-3.1	2	2	2	2	2
1928	-2.7	2	2	2	2	2
1929	8.9	3	3	2	2	2
1930	-0.1	2	2	2	2	2
1931	-4.4	2	2	2	2	2
1932	2.6	2	2	2	2	2
1933	-7.8	1	1	2	2	2
1934	0.7	2	2	2	2	2
1935	-14.4	1	1	1	2	2
1936	4.2	2	2	2	2	2
1937	-3.1	2	2	2	2	2
1938	2	2	2	2	2	2
1939	10.2	3	3	3	2	2
1940	-4.9	2	2	2	2	2
1941	-19	1	1	1	1	2
1942	-13.6	1	1	1	2	2
1943	6.3	3	2	2	2	2
1944	6.7	3	2	2	2	2

Year	Average SOI: Aug to Sep	SOI <-5,>+5	SOI <-7,>+7	SOI <-10,>+10	SOI <-15,>+15	SOI <-20,>+20
1945	2.9	2	2	2	2	2
1946	10.2	3	3	3	2	2
1947	-10.2	1	1	1	2	2
1948	9.4	3	3	2	2	2
1949	-6	1	2	2	2	2
1950	-1.2	2	2	2	2	2
1951	9.6	3	3	2	2	2
1952	-3.7	2	2	2	2	2
1953	-3.5	2	2	2	2	2
1954	-15.1	1	1	1	1	2
1955	7.4	3	3	2	2	2
1956	14.5	3	3	3	2	2
1957	5.6	3	2	2	2	2
1958	-10	1	1	2	2	2
1959	2.2	2	2	2	2	2
1960	-2.4	2	2	2	2	2
1961	6.7	3	2	2	2	2
1962	0.4	2	2	2	2	2
1963	4.8	2	2	2	2	2
1964	-3.8	2	2	2	2	2
1965	14.2	3	3	3	2	2
1966	-12.8	1	1	1	2	2
1967	0.9	2	2	2	2	2
1968	5.5	3	2	2	2	2
1969	-1.3	2	2	2	2	2
1970	-7.5	1	1	2	2	2
1971	8.4	3	3	2	2	2
1972	15.4	3	3	3	3	2
1973	-11.8	1	1	1	2	2
1974	12.9	3	3	3	2	2
1975	9.4	3	3	2	2	2
1976	21.6	3	3	3	3	3
1977	-12.5	1	1	1	2	2
1978	-10.7	1	1	1	2	2
1979	1.1	2	2	2	2	2
1980	-1.8	2	2	2	2	2
1981	-1.9	2	2	2	2	2
1982	6.7	3	2	2	2	2
1983	-22.5	1	1	1	1	1
1984	5	2	2	2	2	2
1985	2.3	2	2	2	2	2
1986	4.3	2	2	2	2	2
1987	-6.4	1	2	2	2	2
1988	-12.6	1	1	1	2	2
1989	17.5	3	3	3	3	2
1990	-0.3	2	2	2	2	2
1991	-6.3	1	2	2	2	2
1992	-12.1	1	1	1	2	2
1993	1.1	2	2	2	2	2
1994	-10.8	1	1	1	2	2

Source: Australian Rainman.

Appendix I Years partitioned into ENSO types

	ENSO Year Types
1894	La Nina
1895	La Nina
1896	El Nino
1897	La Nina
1898	Other
1899	El Nino
1900	La Nina
1901	La Nina
1902	El Nino
1903	Other
1904	La Nina
1905	El Nino
1906	Other
1907	La Nina
1908	Other
1909	La Nina
1910	La Nina
1911	El Nino
1912	El Nino
1913	El Nino
1914	El Nino
1915	El Nino
1916	Other
1917	La Nina
1918	El Nino
1919	La Nina

	ENSO Year Types
1920	Other
1921	La Nina
1922	La Nina
1923	El Nino
1924	Other
1925	El Nino
1926	El Nino
1927	Other
1928	La Nina
1929	La Nina
1930	El Nino
1931	Other
1932	El Nino
1933	La Nina
1934	La Nina
1935	La Nina
1936	La Nina
1937	La Nina
1938	Other
1939	El Nino
1940	El Nino
1941	El Nino
1942	La Nina
1943	La Nina
1944	La Nina
1945	La Nina

	ENSO Year Types
1946	El Nino
1947	La Nina
1948	La Nina
1949	La Nina
1950	Other
1951	El Nino
1952	La Nina
1953	El Nino
1954	La Nina
1955	Other
1956	Other
1957	El Nino
1958	La Nina
1959	La Nina
1960	La Nina
1961	La Nina
1962	La Nina
1963	La Nina
1964	Other
1965	El Nino
1966	La Nina
1967	La Nina
1968	La Nina
1969	El Nino
1970	La Nina
1971	Other

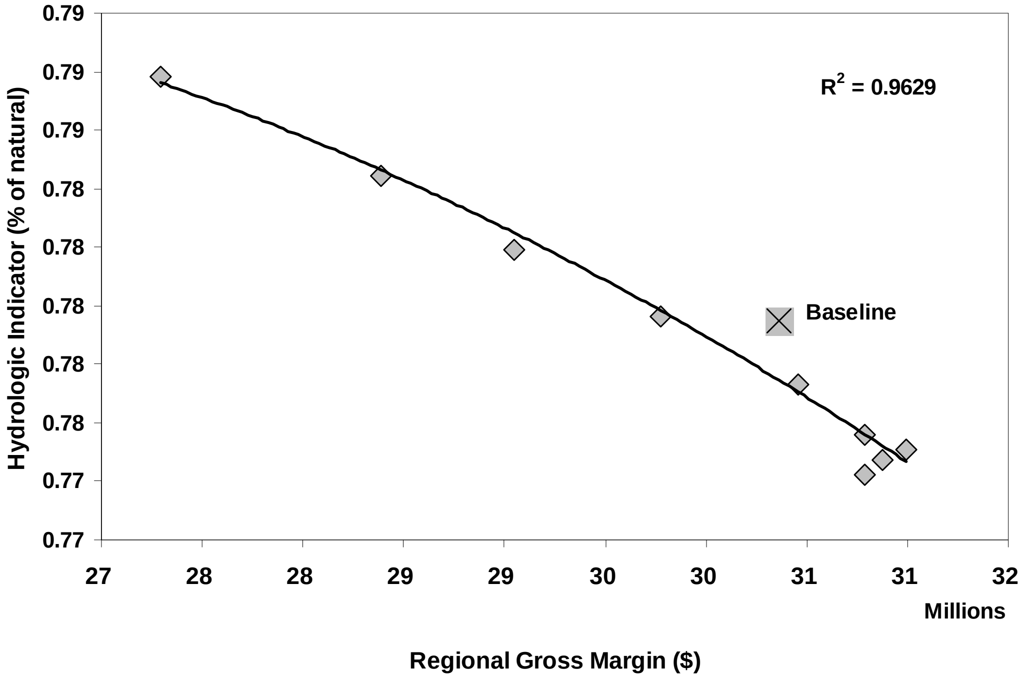
	ENSO Year Types
1972	El Nino
1973	Other
1974	Other
1975	Other
1976	La Nina
1977	El Nino
1978	La Nina
1979	La Nina
1980	La Nina
1981	La Nina
1982	El Nino
1983	La Nina
1984	La Nina
1985	La Nina
1986	La Nina
1987	El Nino
1988	La Nina
1989	Other
1990	La Nina
1991	El Nino
1992	El Nino
1993	El Nino
1994	El Nino

Appendix J Relationship between streamflow and off-allocation water supply for October to February in the Border Rivers Catchment, Australia

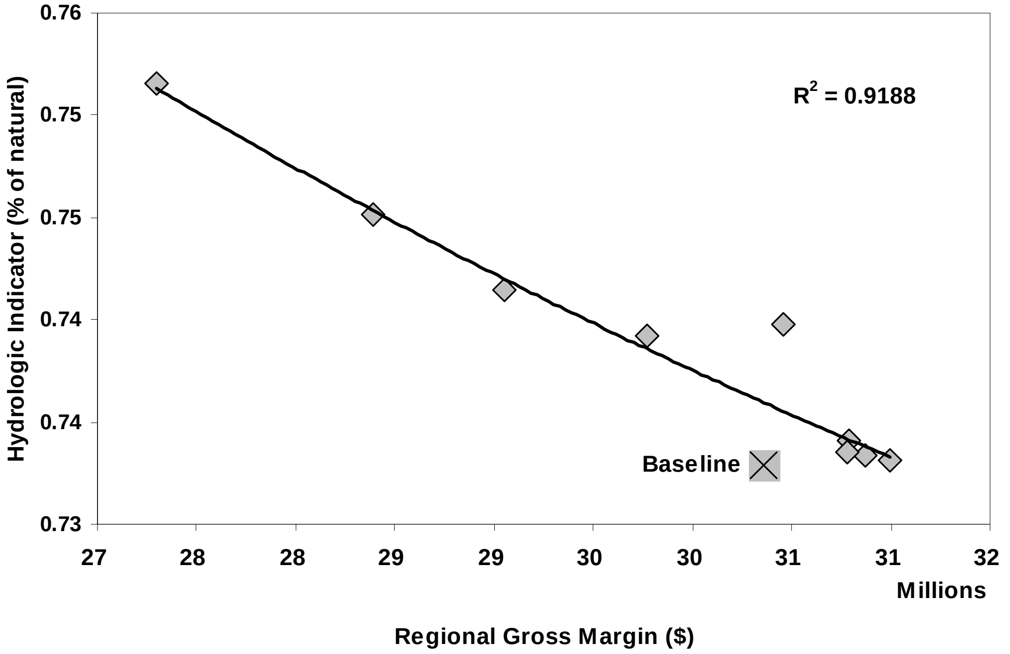
Year	Off allocation flow	Natural flow (Oct – Feb)	Year	Off allocation flow	Natural flow (Oct – Feb)	Year	Off allocation flow	Natural flow (Oct – Feb)
1891	228,246	124,528	1922	105,937	59,670	1953	110,404	73,196
1892	1,207,770	712,411	1923	185,536	78,397	1954	880,914	355,945
1893	290,007	132,722	1924	539,390	245,770	1955	1,916,159	723,631
1894	269,282	131,888	1925	90,294	69,876	1956	169,806	74,995
1895	222,706	107,463	1926	376,035	158,819	1957	24,521	40,803
1896	178,982	95,617	1927	171,157	76,288	1958	382,668	170,685
1897	620,925	288,127	1928	88,282	68,552	1959	700,872	340,204
1898	504,361	232,145	1929	230,976	103,485	1960	49,807	45,576
1899	170,547	93,988	1930	169,095	70,236	1961	786,083	306,803
1900	33,601	39,157	1931	346,203	147,457	1962	130,857	76,390
1901	67,780	35,387	1932	289,274	129,288	1963	180,617	120,122
1902	155,823	50,600	1933	592,784	284,176	1964	173,252	106,151
1903	635,656	374,181	1934	386,035	183,468	1965	376,579	168,517
1904	158,013	92,580	1935	102,813	67,464	1966	418,959	209,369
1905	283,875	125,980	1936	259,301	126,831	1967	153,708	67,073
1906	211,612	107,440	1937	121,280	53,935	1968	161,131	97,463
1907	122,288	73,601	1938	142,633	64,525	1969	297,702	119,974
1908	36,632	10,016	1939	104,136	40,661	1970	1,843,093	869,688
1909	345,148	143,750	1940	175,376	75,009	1971	211,959	114,728
1910	273,345	129,595	1941	93,156	26,170	1972	639,860	319,375
1911	32,627	45,330	1942	567,322	210,757	1973	327,504	184,989
1912	51,604	31,927	1943	485,435	204,678	1974	309,652	137,500
1913	87,205	43,045	1944	52,837	44,476	1975	1,576,659	871,773
1914	100,977	41,657	1945	233,215	113,880	1976	228,288	128,852
1915	44,557	9,877	1946	156,713	81,032	1977	236,002	101,514
1916	956,066	432,473	1947	434,920	191,920	1978	451,643	239,836
1917	905,505	408,899	1948	142,908	87,804	1979	218,257	119,936
1918	26,542	44,027	1949	641,834	292,435	1980	94,276	45,959
1919	32,605	21,760	1950	1,467,990	786,232	1981	95,611	44,804
1920	119,686	69,700	1951	27,607	33,687	1982	115,374	57,469
1921	588,345	323,598	1952	633,100	311,014	1983	840,847	481,464

Year	Off allocation flow	Natural flow (Oct – Feb)
1984	140,701	83,737
1985	211,702	101,421
1986	232,542	111,343
1987	465,821	180,033
1988	76,430	40,052
1989	375,436	180,570
1990	356,025	148,161
1991	400,532	174,037
1992	121,412	54,064
1993	79,887	61,803
1994	92,539	37,883
1995	1,512,569	572,109
1996	361,170	176,962

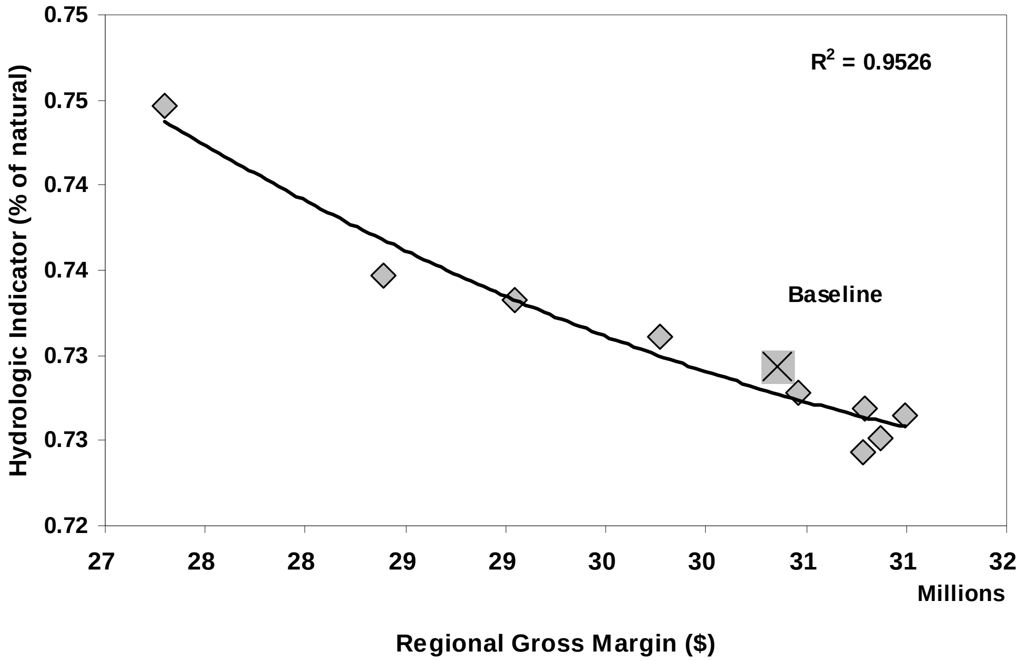
Appendix K Reporting the trade-off curves for the range of hydrologic indicators



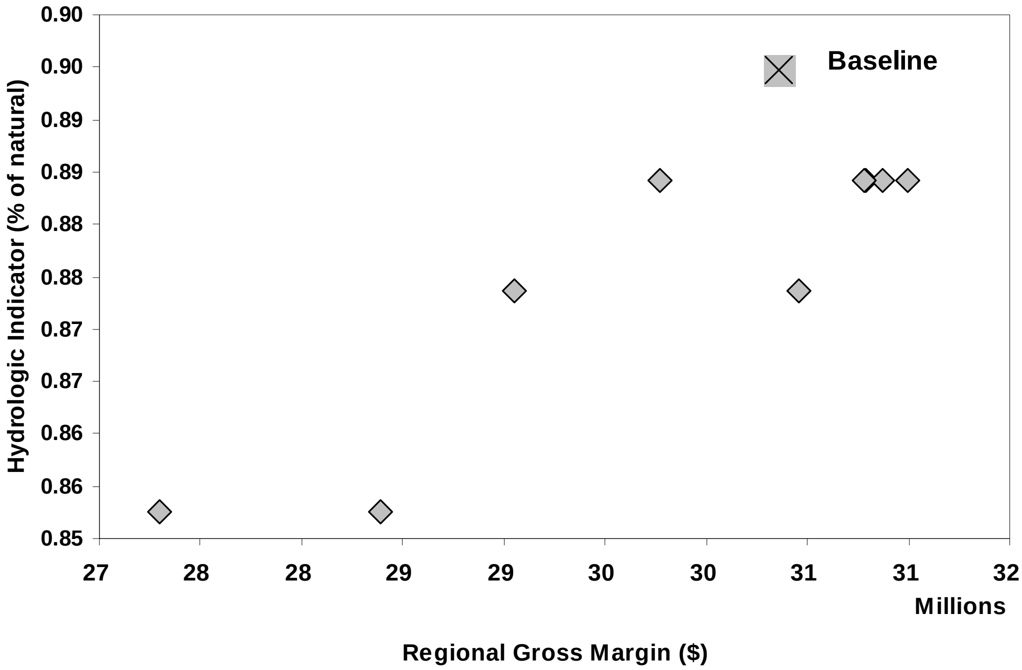
Production possibility frontier using RGM and Mean AF showing a trade-off in outcomes exists



Production possibility frontier using RGM and MAF showing a trade-off in outcomes exists



Production possibility frontier using RGM and PFlowDur 0.1 showing no trade-off in outcomes



Production possibility frontier using RGM and PFlowDur 0.8 showing no trade-off in outcomes

Appendix L Simulation summary results

Table L.1. Economic results of Simulation 1 testing the outcome of gradually increasing access to off-allocation

	Scenarios – off allocation access										
	10 ML	10,000 ML	20,000 ML	30,000 ML	40,000 ML	50,000 ML	60,000 ML	70,000 ML	80,000 ML	90,000 ML	100,000 ML
Economic results											
RGM ($m)	27.30	28.39	29.05	29.77	30.36	30.46	30.79	30.99	30.88	30.78	30.98
Stdev RGM ($m)	23.42	22.53	22.99	22.66	22.50	22.50	22.16	22.63	22.87	22.53	22.89

RGM: Regional Gross Margin in dollars
Stdev RGM: Standard deviation of RGM

Table L.2. Summary of the results for the environmental indicators for Simulation 0

		Scenarios – off allocation access										
		10 ML	10,000 ML	20,000 ML	3,000 ML	40,000 ML	50,000 ML	60,000 ML	70,000 ML	80,000 ML	90,000 ML	100,000 ML
Hydrologic indicators												
Mean		0.79	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.77	0.77
Median		0.75	0.75	0.74	0.74	0.73	0.73	0.74	0.73	0.73	0.73	0.73
PFlow Dur	0.02	0.86	0.86	0.85	0.84	0.84	0.83	0.82	0.82	0.82	0.82	0.82
	0.10	0.74	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.72
	0.50	0.82	0.82	0.83	0.83	0.84	0.84	0.84	0.84	0.84	0.84	0.85
	0.80	0.85	0.85	0.87	0.88	0.89	0.88	0.87	0.88	0.88	0.88	0.88

PFlowDur: Proportion of Flow Duration Percentile

Table L.3. Economic results of simulation testing the outcome of regimes based on the SOI phases.

	Scenarios – off allocation access											
Economic results	**10 ML**	**30,000 ML**	**40,000 ML**	**50,000 ML**	**May SOI**	**June SOI**	**July SOI**	**August SOI**	**Sept SOI**	**Oct SOI**	**ENSO**	**100,000 ML**
RGM ($m)	27.30	29.77	30.36	30.46	29.97	30.14	30.06	29.60	30.13	29.97	30.60	30.98
Stdev RGM ($m)	23.42	22.66	22.50	22.50	22.66	22.32	22.79	22.46	23.07	22.71	21.71	22.87

Table L.4. Economic results of simulation testing the outcome of regimes based on values of the SOI.

	Scenarios – off allocation access									
Economic results	**10 ML**	**30,000 ML**	**40,000 ML**	**50,000 ML**	**SOI <-5,>+5**	**SOI <-7,>+7**	**SOI <-10,>+10**	**SOI <-15,>+15**	**SOI <-20,>+20**	**100,000 ML**
RGM ($m)	27.30	29.77	30.36	30.46	29.80	30.10	30.15	30.92	30.81	30.98
Stdev RGM ($m)	23.42	22.66	22.50	22.50	22.38	22.28	22.35	22.02	22.04	22.87

RGM: Regional Gross Margin in dollars
Stdev RGM: Standard deviation of RGM

Table L.5. Environmental results of Simulation 1 testing the outcome of regimes based on the SOI phases.

		Scenarios – off allocation access								
		Baseline	**May SOI**	**June SOI**	**July SOI**	**August SOI**	**Sept SOI**	**Oct SOI**	**ENSO**	**100000 ML**
Environmental results										
Mean		0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.77
Median		0.73	0.74	0.74	0.73	0.74	0.74	0.73	0.74	0.73
PFlowDur	0.02	0.83	0.83	0.83	0.82	0.82	0.83	0.84	0.83	0.82
	0.10	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.72
	0.50	0.84	0.84	0.83	0.84	0.84	0.84	0.84	0.84	0.85
	0.80	0.88	0.86	0.86	0.88	0.86	0.88	0.87	0.87	0.88

RGM: Regional Gross Margin in dollars
Stdev RGM: Standard deviation of RGM
PFlowDur: Proportion of Percentile Flow Duration

Table L.6. Environmental results of Simulation 1 testing the outcome of regimes based on the SOI values.

		Scenarios – off allocation access						
		Baseline	**SOI <-5,>+5**	**SOI <-7,>+7**	**SOI <-10,>+10**	**SOI <-15,>+15**	**SOI <-20,>+20**	**Proxy Perfect**
Environmental results								
Mean		0.78	0.78	0.78	0.78	0.78	0.78	0.77
Median		0.73	0.74	0.74	0.74	0.74	0.74	0.73
PFlow Dur	0.02	0.84	0.83	0.83	0.84	0.83	0.83	0.82
	0.10	0.73	0.73	0.73	0.73	0.73	0.73	0.72
	0.50	0.84	0.84	0.84	0.84	0.84	0.84	0.85
	0.80	0.89	0.87	0.86	0.86	0.88	0.88	0.88