AN APPROACH TO EFFICIENTLY MANAGING ENVIRONMENTAL WATER ALLOCATIONS

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ABSTRACT

Recent policy changes in Australia have led to the creation of an Environmental Entitlement that can be actively managed. Existing environmental flow recommendations tend to specify a fixed flow regime, with target minimum flow volumes for each element of the flow regime. There is now a volume of water in storage specifically for environmental purposes, and thus environmental flow recommendations need to be structured to allow operational decisions to be made in an adaptive manner. It is likely that the volume of water available to an environmental manager will not allow all elements of the flow regime to be provided in every year. With a limited resource, environmental managers need a transparent approach to decide which elements of the flow regime to provide and which to sacrifice in a given year. This thesis uses the Goulburn River, Australia, as a case study to look at the decision making process. An optimisation model is constructed to determine the timing of environmental water releases to minimise environmental risks. In order to optimise releases, an understanding of the marginal benefit of water to the environment is required. Environmental response curves, relating flow to environmental outcome, are constructed for each significant flow element. These response curves change with time depending on climate and antecedent environmental stress. The optimisation model makes trade-off decisions between the various environmental flow elements to determine the optimal monthly release pattern. This approach provides important insights about the types of information that are needed for efficient management of environmental water and the sensitivity of decisions to the accuracy of this information.
DECLARATION

This is to certify that

(i) the thesis comprises only my original work towards the PhD,
(ii) due acknowledgement has been made in the text to all other material used,
(iii) the thesis is less than 100,000 words in length, exclusive of tables, maps, bibliographies and appendices.
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LIST OF ACRONYMS

ARI - average recurrence interval

BE – Bulk Entitlement

COAG – Council of Australian Governments

EWA - California’s Environmental Water Account

EWM – Environmental Water Manager

EWR – Environmental Water Reserve

GMW – Goulburn Murray Water

GSM - Goulburn Simulation Model

IVTs - inter-valley transfers

MDB - Murray Darling Basin

MFAT - Murray Flow Assessment Tool

MROM - Murrumbidgee River Options Model

MSBc - marginal social benefit of consumptive water use

MSBe - marginal social benefit of environmental water

NWC – National Water Commission

NWI - National Water Initiative

REALM – Resource Allocation Model
LIST OF SYMBOLS AND INDICES

\[ \in \] over the set of values \{\ldots\}

\[ \forall \] for all listed parameters

\[ \Sigma \] sum

\( a \) time in years, \( a \in \{1 \ldots 5\} \)

\( d \) environmental reach, \( d \in \{1 \ldots 2\} \)

(representing Reach 1 downstream of Lake Eildon and Reach 2 downstream of Goulburn Weir)

\( h \) initial storage level, \( h \in \{1 \ldots 3\} \)

(representing low, medium and high storage levels)

\( i \) breakpoint number (used as a model input to solve piecewise linear problem)

\( j \) defines sequence of year-type and storage level

\( o \) initial risk scenario used as model input to define when environmental flow was last provided for each flow component

\( p \) climate or year type, \( p \in \{1 \ldots 3\} \)

(representing a dry, average and wet year)

\( q \) environmental flow element, \( q \in \{1\ldots9\} \)

(representing the flow elements shown in Figure 4-9 and Figure 4-17)

\( r \) risk factor, \( r \in \{1\ldots3\} \)

(representing 0 years to 2 years since flow component was last provided)

\( s \) annual environmental allocation (in GL)

\( t \) time in months, \( t \in \{1 \ldots 12\} \)
1 Introduction

1.1. Problem Statement

Historically, governments and central water authorities controlled the allocation of water resources in Australia. However, in many catchments, existing water resources are now fully allocated and the economic and environmental costs of augmenting supply have increased sharply. As a consequence, the approach to water management in Australia has changed significantly in recent years. Two key changes were the move from government allocation of water to one in which a water market now plays a significant role and the recognition of the environment as a legitimate water user.

Government reforms have concentrated on the efficient allocation of water, treating it as an economic good using market instruments to reallocate a scarce resource among competing users. Water markets have the potential to improve overall economic efficiency (and net social gain) of the system, however, in reality there are a number of potential market failures that may impact negatively on the overall value derived from water use (Etchells, 2004, pp. 43 - 4). Environmental flows are a market failure because consumers of water are not obliged to include the cost to the environment in the price of water. That is, market price is not reflecting an external effect of water extraction. In the current water market, there is no mechanism for compensating those affected by a decline in water available for the environment.

Policies protecting instream environment have historically used regulation, or ‘command and control’, applying licence specifications and trading rules to maintain environmental water requirements (National Competition Council, 2004). A recent policy initiative in providing water for the environment was to grant it with an entitlement whose legal status was the same as water allocated for consumptive users (in Victoria this is referred to as the Environmental Water Reserve (EWR), other Australian states use different terminology). In contrast to the regulatory approach, an environmental water entitlement provides a parcel of water that can be actively and adaptively managed to achieve better
environmental outcomes. It also allows an environmental manager to enter the water market on behalf of the environment.

However, this newly gained flexibility requires clear decision processes about how and when to best use available environmental water. Appropriate decisions will require priorities for allocating environmental water. Most likely there will be insufficient water to meet all environmental demands in every year. Therefore tradeoffs between different competing environmental demands will be required. There is increasing community scrutiny of how these decisions are made and what environmental outcomes they achieve. This, combined with the pressures of drought, has forced policy makers into developing procedures for clear and accountable allocation of environmental water.

Various methods have been used to prioritise or assess environmental flows. Many of these were developed to provide environmental flows for the previous ‘command and control’ approach. While they may be useful for long term planning purposes they are not suitable for operational decisions allowing for changes over time in environmental status, climate, water availability and consumptive use decisions. As well, these methods tend to be subjective and poorly documented. Thus, an approach allowing transparent and consistent assessment of tradeoffs between different environmental flow requirements is required. This research addresses this issue.

1.2. Research Aim

The aim of this study is to develop a conceptual framework for prioritising environmental water releases to maximise environmental amenity in a flexible and transparent manner. As the volume of water allocated to the environment is fixed, how much should be released, and when, to achieve the best environmental outcome? Policy makers are still grappling with the questions of i) how water should be divided between consumptive users and the environment and ii) the volume of the environmental allocation. However, once an environmental allocation is set, how can this water best be used? The latter question is the main research question of this thesis.
A point of clarification is required at the outset of this thesis. It is often assumed when discussing environmental water allocations that the question being addressed is how much water should be allocated to the environment as opposed to other consumptive users. This is not the question of the research described in this thesis. The premise of a market is that each individual acts in their own interest. If the environment is to participate in the market, environmental managers must first understand the environment’s demand for water. This requires an understanding of how each additional megalitre of water allocated to the environment would be used, when it would be released and which flow components it would target. Here lies the focus of this research. While this research does not determine how much the government should allocate to protect the environment, it is an important first step because it begins to provide the tools to quantitatively assess the benefits gained by purchasing water. Without these tools the environment can not enter the market nor can governments make rational decisions about the volume of the environmental water allocation. Once a parcel of water has been allocated to the environment, decisions will be required about the best use of this water to achieve environmental outcomes. While there may be discomfort in making “trade-off” decisions between environmental outcomes, this is the reality of our current allocation system and the almost unlimited competing demands on the use of a limited water supply. The aim of this study is to develop a conceptual framework for prioritising environmental water releases in a flexible and transparent manner.

This will be done addressing the following three research questions:

1. How can environmental water requirements be quantified to indicate the marginal value of water to the environment?
2. What factors influence the decision to release environmental water in a given month?
3. Does an optimisation approach provide the flexibility and transparency to make release decisions?

Environmental flow recommendations are difficult to validate. Gippel states:

_The number of Australian studies that report environmental flow recommendations greatly outweighs the number of studies that report_
scientific evaluation of the results of implementation of these recommendations. In fact, very few such studies can be found in the Australian or international literature. It is impossible to say, on the basis of previous experience and knowledge, how far the environmental flow regime can depart from the natural flow regime in order to maintain or improve biological diversity...The best way to test hypotheses regarding environmental flows is to trial full-scale implementation (2001, pp. 84-5).

As such, the framework and model developed can not easily be validated against real data on environmental outcomes. Rather, they are based on the best information available and will require continual monitoring and adaption as new data becomes available. This is a difficulty encountered in all studies relating to environmental water requirements.

Because flows in unregulated systems cannot be actively managed, and the comparatively thin nature of the water market, the research described in this thesis will focus on environmental release decisions in regulated systems.

1.3. Thesis Structure

An outline of the thesis structure is shown in Figure 1-1. The following chapter (Chapter 2) introduces the fundamental concepts of water markets and environmental flow policy, focussing on, although not limited to, the Australian context. Policy to protect river ecosystems has changed rapidly in Australia. Through discussion of these policies and the science of environmental flows, the successes and limitations of existing approaches to allocating environmental water are highlighted.

Chapter 3 outlines the research questions and method in detail. The Goulburn River Basin is introduced as a case study for this research project. The system operation of the Goulburn River, major storages and demands are described briefly, but in sufficient detail to inform later modelling decisions.

Chapter 4 discusses the limitations that current environmental flow recommendations place on decision making for environmental water allocations. The importance of knowing the marginal value of water to the environment is discussed. The chapter
outlines an approach to developing an environmental demand curve and applies this to the study basin.

Chapters 5 and 6 describe optimisation models that use the environmental demand curves to determine storage release decisions that achieve optimal environmental outcomes. In Chapter 5 a linear optimisation model is developed to determine monthly release patterns over a single year. The optimisation process is complex: beginning with a single year model allows substantial investigation before adding the complexity of a multi-year analysis. In Chapter 6 this model is expanded to produce a multi-year model, showing how environmental release decisions would be made over a number of years.

The model outcomes are discussed further in the context of the research questions in Chapter 7. The approach developed, while based on a study basin, is still conceptual in form. The modelling is used to understand which model inputs require more detailed analyses, and the limitations of the current model are discussed. This highlights a number of areas where further work could be undertaken. The research is drawn together in Chapter 8, where conclusions are drawn specifically to address the research aims.

This research demonstrates how an optimisation approach can be used to improve the transparency of decisions about environmental water use. The novelty of this approach is the use of environmental response curves that provide information on the marginal value of water to different elements of the environmental flow regime. An understanding of the marginal value of water allows clear trade-off decisions on how to make best use of environmental water. The case study demonstrates the array of questions and management decisions that can be assessed using an optimisation technique, and highlights the variables that play a key role in the decision making.

Two terms used throughout the thesis require definition:

- Environmental water describes the water rights of the environment irrespective of the form of the water rights.
- An Environmental Water Manager (EWM) is the designated entity responsible for the management and use of the environment’s water rights.
A list of acronyms and a list of symbols are provided at the start of the thesis. A glossary of terms is provided before an appendix detailing the Goulburn Simulation Model.

Figure 1-1: Outline of thesis structure
2 Background

2.1. Introduction

Australia’s water policy has changed significantly in the past few years, bought about by years of low rainfall and increasing water demands. Two key changes have been the introduction of a water market and the recognition of the environment as a legitimate water user. This chapter is an overview of the issues surrounding the provision of environmental water in the current water framework in Australia.

The chapter begins with a discussion of both international and Australian water policy for the provision of instream flows. Because the water market is now a central part of water allocation in Australia, discussion of any element of water resource management requires some understanding of how the water market works. The chapter goes on to introduce the water market and its relevance to the provision of environmental water. It highlights the difficulties in defining clear objectives and requirements for environmental water, and how these contribute to the difficulty in establishing the value of environmental water. The options for increasing the existing environmental water reserve are discussed, along with management arrangements for environmental water. This chapter aims to set the context of environmental water in Australia and the issues currently on the agenda.

2.2. Water policy for instream flows

2.2.1. International Context

“Sustainability” is now a well known concept, discussed in every aspect of policy and development. From the 1970’s, the sustainability concept gained momentum through numerous international declarations and conventions (for example Ramsar convention for wetland protection (1971), World Commission on Environment and Development (1987) and the Convention on Biological Diversity (1992)). The approach to water management has followed a similar trend, with social and environmental aspects gaining recognition in water projects and planning (for example, the World Commission on Dams (1998 – 2000)) (Zaman, 2005).
Many individual countries have also developed to promote sustainability in water resource management. The concept of providing water for the environment (or environmental flows) originated in the western USA where reduced streamflow was seen to impact on game-fish reserves, and especially salmon populations (Moore, 2004). Gleick (2006) provides a summary of flow conservation projects and legislative approaches to protect instream habitat in different countries. Only a brief overview is provided here. There are at least 70 nations where environmental flow initiatives are being undertaken (Moore, 2004). These projects include:

- restrictions on withdrawals (both surface water and ground water);
- redesign of dams; and
- purchase or withdrawal of consumptive water rights (Gleick et al., 2006).

However, throughout most of the world, environmental flows policy and legislation is in its infancy (Tharme, 2003). Countries where environmental flow policy is more developed tend to have a higher public awareness, available expertise and pilot projects to demonstrate the benefits of environmental flows (Moore, 2004). Japan and Israel both amended their water acts (in 1997 and 2004 respectively) to reflect the benefit of environmental flows. Switzerland, moving a step further, amended its Water Protection Act in 1991 to mandate minimum environmental flows (Gleick et al., 2006). USA (particularly in the Western States), Chile and Mexico all operate using the prior appropriation doctrine, where priority to water is based on the date at which the water licence was created. While the environment is now recognised as a legitimate water user, its water right is lower in seniority to other licences due to it’s relatively immaturity. Water management is usually a state issue in the USA, however federal laws have provided some protection to instream health indirectly through the Endangered Species Act and the Clean Water Act (Gleick et al., 2006).

South Africa, with its National Water Law (1998) has perhaps the strongest legislated protection for environmental flows. The act creates non negotiable water reserves to provide (i) basic human needs and (ii) ecological function and the long term sustainability of aquatic ecosystems (Gleick et al., 2006). Under this legislation, provision of environmental requirements is part of supplying an ongoing water resource into the
future, rather than as a competitor for that resource. The environmental needs (and basic human needs) are met before any commercial consumptive use occurs.

Australia has undergone major water reforms since 1994, and the environment is now seen as a legitimate water user. The concept of environmental flows entered the public domain in Australia as a response to hydropower developments in the 1970s (Moore, 2004). The remainder of this chapter discusses Australian water policy in some length, particularly as it relates to environmental flows. Australia originally took a regulatory approach to providing environmental flows, not dissimilar to the countries discussed above. However, as will become apparent in the next section, in recent years Australia has moved towards purchasing entitlements for the environment. The market is used as a mechanism to increase environmental water entitlements. The distinguishing element from policy in other countries is that an environmental manager will now have a parcel of water in storage that can be actively managed. With this policy approach comes a set of specific issues relating to the management of these entitlements. The Australian policy is thus discussed in some details as it sets the constraints and philosophy of environmental water management.

2.2.2. Australian Context

Policies protecting instream environments historically used ‘command and control’ or regulation, applying licence specifications and trading rules to maintain environmental water requirements (National Competition Council, 2004, p. 16). Australian water policy was reformed in 1994 when the COAG Water Reform Framework recognised the need to improve and co-ordinate water management. As part of this reform, jurisdictions were obliged to recognise the environment as a legitimate user of water through formally allocating water to the environment and introduce water trading.

In 1997 a cap was imposed on water diversion in the Murray Darling Basin (MDB) because of concerns for its health. The cap was defined for each state as the volume of water allocated under 1993-94 levels of development. Caps have also been imposed in other Australia basins, however the MDB cap is the most widely discussed because of its size and the issues surrounding coordination between states. These policies also led
water trading within the MDB to be embraced. In Australia, water trading may take the form of a) transfer of a water entitlement (a permanent transfer), b) a nominated volume of water associated with an entitlement (an annual allocation or temporary transfer), and c) less commonly, leasing and the use of forward water contracts and options.

The 2004 National Water Initiative (NWI) was an attempt “to complement and extend the reform agenda...intended by COAG” (Intergovernmental Agreement on a National Water Initiative, 2004). In particular, the NWI led to a clearer definition of water entitlements — as perpetual rights providing guaranteed access to a specified share of available water resources, separated from land and regulated by a water resource plan. Each year a seasonal volumetric allocation of water is assigned to a water entitlement, based on the share of the resource pool specified on the water entitlement and the water supply available in that season. Another NWI objective was to return completely all over allocated systems to environmentally-sustainable extractions levels. Consequently, some to all of the water savings realised by government funded (private and public) infrastructure projects have been secured for the environment. The Commonwealth Water Act (2007)\(^1\) introduced further change by allowing water for the environment to be purchased directly from the market (and consumptive users) and is thus a shift away from these previous ‘command and control’ policies.

2.3. How water markets work

Historically, water management in Australia has been through a centralised, control and command framework, where government regulations and central authorities define how water resources can be used and by whom. As the water economy enters its mature phase and water approaches full allocation in many catchments, the economic and environmental costs of augmenting supply have increased sharply. As a result, the

\(^1\) The Commonwealth Water Act refers certain powers to the Commonwealth government, from individual state governments in the Murray Darling Basin. This is aimed at providing a central management structure of the MDB.
Council of Australian Governments’ (COAG)\(^2\) reforms recognized the need for a more efficient allocation system, treating water as an economic good and using pricing mechanisms and market instruments to reallocate scarce resources among competing users, promoting higher valued and more efficient water use (Bjornlund, 2000).

A water market for permanent and temporary trade of irrigation water is well developed in Victoria. The COAG agreement made this possible through the separation of water rights from land rights. There have been a number of comprehensive reviews of water market development in Australia (including Tisdell et al., 2002; Young et al., 2000; Zaman, 2005), so only a brief description of the theory of water markets is included here.

Australia is not alone in using a market approach to water allocation. Since the 1960s, there has been a proliferation of literature and ideas espousing the benefit the market as an instrument to allocate water. The first water markets developed in the western United States of America in the 1980s, followed by some areas of Latin America (Zaman, 2005).

Basic descriptions of markets are found in any microeconomic textbook (Saliba & Bush, 1987 provides a good introduction in terms of water markets). A summary is included here to provide background for later discussions. The basic premise behind a water market is that individuals will act to maximise their own individual gain, and as such, a trade will only occur if it is of benefit to both the seller and the buyer. In a water market, this will facilitate the movement of water from low profit farming to higher profit farming. The market will thus facilitate an overall gain in efficiency and use of a scarce resource.

*The motivating force behind market development is mutual perception by potential buyers and sellers that economic gains may be captured by transferring water to a location, season or purpose of use in which it generated higher net returns than under the existing use patterns (Saliba & Bush, 1987, p. 4).*

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\(^2\) COAG is the peak intergovernmental forum in Australia (comprising the Prime Minister, State Premiers, Territory Chief Ministers and the President of the Australian Local Government Association)
Figure 2-1 represents the fundamental concepts of a water market between irrigators. The total benefit curve is the total value that two irrigators place on various quantities of water (Figure 2-1a). The marginal value is calculated as the derivative (or the gradient \( dTV/dQ \)) of the total value; in other words, the marginal value is the value to farm profits of each additional quantity of water. While marginal value of water will vary according to its use, the downwards slope of the marginal value curve shows that as more water becomes available, the value of each additional unit of water decreases (diminishing marginal value productivity).

Figure 2-1b shows two marginal value curves, one for Irrigator A and one for Irrigator B. The market is in equilibrium at the point where the curves intersect (point C), because resources are allocated so that neither irrigator can be made “better off” without making the other farmer “worse off”. The assumption here is that irrigators will act rationally to maximise their profits. If the initial allocation of water is at point \( Q' \), then irrigator B will buy water until \( Q' \) returns to \( Q^* \) because the return (or marginal value) on water for irrigator B is higher than the return for irrigator A at \( Q' \) (point B is higher than point A in Figure 2-1b). If the market moves from \( Q' \) to \( Q^* \), the net social gain is the area ABC (as shown in Figure 2-1c). In other words, the total gain is the difference in the integral of the marginal values.

Figure 2-1: Overview of basic market concepts.

While water markets have the potential to improve overall economic efficiency (and net social gain) of the system, in reality there are potential market failures that can impact
negatively on the overall value derived from water use (Etchells, 2004). Saliba and Bush (1987) summarise five potential water market failures as follows:

1. *External effects of market activities* – this occurs when water transfers impact on a third party who was not involved in the decision process or transfer negotiations (e.g. salinity, increasing water tables).

2. *Public goods characteristics of water resources* – some uses of water provide benefits to more than one person (non rival) and it is difficult to limit the benefits to those who pay for them (non excludable). It is therefore difficult to efficiently allocate water to uses that have public good characteristics (e.g. environmental services).

3. *Imperfect Competition* – when individual buyers or sellers can influence market prices, a market is characterised as imperfectly competitive. Prices will no longer reflect the marginal value of water in alternative uses.

4. *Risk, Uncertainty and Imperfect Information* – a competitive market becomes efficient when accurate information about supply is available, is equally available to all participants in the market and that the risk of investment is well understood.

5. *Equity and conflict resolution* – external intervention (for examples subsidies) may interfere in the price system and inaccurately represent opportunity cost. In almost all cases, tax and social security systems are more efficient than water subsidies in achieving equity.

In the context of environmental flows, a market failure occurs both due to external effects of market activities and due to the public good characteristics of environmental water. Water consumers are not obliged to include the cost to the environment in the price of water. That is, market price is not reflecting an external effect of water extraction. In the current water market, there is no mechanism available for compensating those affected by decreased environmental flows. The problem is exacerbated by the public good characteristics of environmental water, making it difficult to ascertain the true total benefit of providing water to the environment and thus, also making it difficult to find the optimal allocation between consumptive users and environment. The difficulty arises in determining the marginal value of water to the environment. The total benefit curve is based on the value that each member of the public, each user of the river or stakeholder
places on providing water for the environment. There are obvious difficulties in isolating the beneficiaries of environmental flows, and then in determining the benefit curve of every relevant person.

2.4. Water and environmental amenity

Historically, governments and central water authorities defined the use of water resources in Australia. However, in many catchments, existing water resources are now fully allocated (and in many cases over-allocated) (Department of Sustainability and Environment, 2004, pp. 43 - 4) and the economic and environmental costs of augmenting supply have increased sharply. As a result, Australian governments are using market instruments to reallocate the scarce resource among competing users and promote higher valued water use. Thus, Australia has gone down the market path, treating water as an economic good.

The initial issue of water entitlements, reflecting historical allocations, were distributed largely on a first-come-first-served basis, and were usually purchased at the operating costs of the publicly funded capital infrastructure for the storage and distribution of water. One consequence of this distribution of entitlements is that, for many rivers, current demand for water exceeds available supply. Apart from the obvious constraints this places on consumptive users, it has also adversely affected the environmental condition of our Rivers.

Although economic efficiency (and net social gain) is potentially improved by water markets, the public good characteristics of water for environmental flows add complexity. Water markets among irrigators (and other consumptive users) are assumed to involve the trading of rival, excludable, private goods (although even this is debatable when return flows are considered). This means the private benefit to an individual is equal to the social benefit. However, the amenity and recreation value of rivers and environmental water are non rival and non excludable goods (i.e. public goods\(^3\)). As public goods are

\(^3\) Recall that public good characteristics refer to water providing benefits to more than one person (non rival) and being difficult to limit the benefits to those who pay (non excludable).
non rival and non excludable, there is a tendency for individuals to free ride, leading markets to under provide the public good. In this situation, a case can be made for government intervention to ensure the adequate provision of public goods such as water for the environment.

Water is also a common resource, which in the absence of exclusive, enforceable rights to water, would be over-exploited by water users. This outcome is commonly referred to as the “tragedy of the commons”. It can be avoided by levying a charge on water use, or alternatively, creating exclusive and enforceable rights to the available (capped) resource and using the market to make the opportunity cost of water use transparent.

For economic efficiency, the ideal allocation of water between users maximizes marginal social benefit. We aim for marginal social benefit of consumptive water use (MSBc) to equal market price, and approximately equal the marginal social benefit of water allocated to the environment (MSBe). The price of water, and by implication MSBe, will be available from the water market. However, estimation of MSBe requires further work. The difficulties in determining the economic benefit of providing water to the environment are discussed below. An alternative approach is to recognize that all citizens benefit from environmental amenities, which therefore have public good properties. These are best funded from general government revenues, competing with public expenditures on other “public goods” (such as defence, law and order, national parks, education, etc.). Requests for more public funds to purchase additional water property rights for the environment would be part of the regular annual budget allocation process (backed up by social benefit cost studies that try to equate MSBc and MSBe).

2.4.1. Objectives of providing environmental water

The current allocation of water to the environment is estimated using available knowledge of aquatic ecosystem requirements. The objectives of this environmental allocation are often stated in terms of “providing a healthy working river” without clear
definitions or measures of these objectives. Without clear links between environmental flow provision and objectives, and how these objectives are valued, it is difficult to assess the economic benefit of providing environmental water. When considering the reallocation of water to the environment, these issues makes comparison to the value of water for consumptive users complicated.

There are many broad statements defining objectives for environmental health. The National Water Initiative seeks to “ensure ecosystem health by implanting regimes to protect environmental assets at a whole-of-basin, aquifer or catchment scale” (National Competition Council, 2004). This policy does not provide targets for ecosystem health; it simply suggests that the best available science should be used to determine the flow required to ensure ecosystem health. Nor does the policy suggest what level of protection is appropriate or what constitutes ecosystem health.

ANZECC National Water Quality Management Strategy (2000) and the National River Health Initiative (Department of Environment and Heritage 2002) define ecological health as:

The ability of an ecosystem to support and maintain key ecological processes and organisms so that their species compositions, diversity and functional organisations are as comparable as possible to those occurring in natural habitats within a region.

The recent NCC report (National Competition Council, 2004) notes that while this definition is useful, it does not include assessment of the trade-offs with other human use of water.

The living Murray defined the term “Healthy working River” as a river that is managed to provide a sustainable compromise between the condition of the river and the level of human use” (National Competition Council, 2004).

However, once again there is no indication as to how “sustainable” is defined. It is likely that our objectives for river health and our reliance on the river for other uses will vary

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4 The Living Murray project defined a “healthy working river” as 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. Further details can be found in: (Jones et al., 2002)
with time. The relative value the community places on river health will also vary with time and inform our objectives.

One way around this is to have a hierarchy of objectives, with broad concepts or visions for the river filtering through to measurable indicators of those broader objectives. The recent Victorian environmental flows monitoring and assessment project (Chee et al., 2006) used this sort of approach to link measurable outcomes to the broader environmental flow objectives.

2.4.2. Environmental water requirements

While there is progress in quantifying environmental flow requirements, it is still a new science. Techniques for determining environmental water requirements range widely (Arthington & Zalucki, 1998), with the data and resources available in a catchment determining the method and accuracy. Tharme (2003) provides a clear overview of the progress in environmental flow assessments and the range of developed approaches. Environmental flow methods can be grouped into four categories: hydrological rules, hydraulic rating methods, habitat simulation methods and holistic methods (Arthington et al., 2006). Management objective, available expertise, resources (time and money) and the existing legislative structure will all play a role in determining the most appropriate method for a given catchment (Acreman & Dunbar, 2004).

Many environmental flow methods are based on the natural flow paradigm (Arthington & Zalucki, 1998), which considers that the entire flow regime, including natural variability, to be important in maintaining the health of aquatic ecosystems. However, each component of the natural flow regime provides different ecological triggers, important in maintaining the integrity of a river system. These flow components can be described by their “magnitude, frequency, duration, timing, and rate of change” (Poff et al., 1997). It is neither necessary, nor perhaps desirable, that habitat be maximized every year as this would create very artificial conditions for the biota of interest.

Long-term studies of naturally variable systems show that some species do best in wet years, that other species do best in dry years, and that overall
Thus both intra and inter-annual variation in flow provide the habitat dynamics that maintain biological diversity and ecosystem function.

The FLOWS method is commonly used across Victoria to determine environmental flow requirements (DNRE, 2002). Like many other environmental flows methods, recommendations from FLOWS are used as an input to long term planning decisions to suit the traditional regulatory approach to providing environmental water. The method combines elements of previous environmental flow determination methods (including hydrological methods, habitat based methods and holistic methods) to better suit the Victorian situation. It should be noted that the FLOWS method is a framework to facilitate determining environmental flows; the rationale is left to those applying the method.

_{FLOWS is based on the natural flow paradigm, which suggests that different parts of the flow regime have different ecological functions, and examines changes to components of the flow regime in order to arrive at recommendations" (Cottingham et al., 2003).}_

The environmental flow requirement represents the key components of the natural flow regime necessary for biological, geomorphological and physicochemical processes (DNRE, 2002). The FLOWS method summarises the key flow components as shown in Figure 2-2 and Table 2-1. Each flow component is described by the magnitude, frequency, duration, timing and ecological significance of the event. A more detailed description of each flow component and their ecological significance can be found in DNRE (2002).
Figure 2-2: Time series showing key components of the natural flow regime

Table 2-1: Common flow components in environmental flow recommendations (DNRE, 2002)

<table>
<thead>
<tr>
<th>Flow Component</th>
<th>Channel flow characteristic</th>
<th>Timing</th>
<th>Frequency</th>
<th>Duration</th>
<th>Key Function</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cease to Flow</td>
<td>No surface flow</td>
<td>Summer</td>
<td>Annual</td>
<td>Vary from days to months</td>
<td>Ecological disturbance Dries habitats and substrates Facilitates organic matter and carbon processing</td>
</tr>
<tr>
<td>Low Flow</td>
<td>Minimum flow in channel. Continuous flow in some part of channel</td>
<td>Summer</td>
<td>Annual</td>
<td>Weeks to months</td>
<td>Connect instream habitats System maintenance</td>
</tr>
<tr>
<td>Freshes</td>
<td>Flow greater than median flow for that period.</td>
<td>Summer Spring</td>
<td>Can be several in each period</td>
<td>Generally days</td>
<td>Biological triggers Inputs to habitats Physico-chemical changes</td>
</tr>
<tr>
<td>High Flow</td>
<td>Connect moist in channel habitats. Less than bankfull. May include flow in minor flood plains.</td>
<td>Autumn Winter Spring</td>
<td>May be several annually</td>
<td>Weeks to months</td>
<td>Inundation of instream habitats Channel connectivity Allows migration Inundation of organic matter Sediment movement</td>
</tr>
<tr>
<td>Bankfull</td>
<td>High flow within channel capacity. Flow in other channels (anabranches etc)</td>
<td>Winter Spring</td>
<td>Generally at least annually</td>
<td>Days to weeks</td>
<td>Channel and habitat forming Sediment transport</td>
</tr>
<tr>
<td>Overbank</td>
<td>Flow extends to floodplain surface flow.</td>
<td>Winter Spring</td>
<td>Can be annual or less frequent</td>
<td>Days</td>
<td>Floodplain connectivity Organic matter inputs</td>
</tr>
</tbody>
</table>

The FLOWS method requires a scientific or expert panel that identifies key environmental assets and objectives through site visits and examination of existing data. These environmental assets are then linked to key flow components. By examining the natural and current flow series, and through use of a hydraulic model, flow
recommendations are then developed to meet the set objectives. The FLOWS method is outlined in Figure 2-3.

![Figure 2-3: Outline of the FLOWS method (DNRE, 2002).](image)

An important problem in using the FLOWS method to define environmental flow requirements is that objectives are set specifically with reference to the current flow regime. Therefore, recommendations often fail to specify important aspects of the natural flow regime that are already provided in the current flow regime. Any changes in operation will not take consideration of this aspect. Methods such as the DRIFT (Downstream Response to Imposed Flow Transformation) method go someway to addressing this issue by comparing the environmental (and social and economic) impacts of a range of flow regimes and discussing them in terms of risk levels (King et al., 2003).

The highly complex dynamics and interactions between flow components and species of these ecological systems are described by an extensive literature (for example, Chee et al., 2005). Environmental flow studies reflect this complexity using proxy variables or indicators of river health rather than changes in population of specific species. As an example, habitat provision may be used as a proxy for a species population. Thus, the objectives of the environmental flow recommendations do not correspond directly (or numerically) to the environmental elements that the community values, making this value more difficult to estimate.
The condition of the instream environment is not isolated to the impact of streamflow, but is affected by a range of other factors. These include catchment land use, fishing pressure, introduced species, riparian vegetation cover, and large woody debris distribution (Gordon et al., 2004, p. 286). It is therefore difficult to isolate environmental responses due solely to flow regime changes.

While the uncertainties in the modelling that underpin environmental flow recommendations are widely acknowledged, they are rarely quantified. For example, the limited data available on specific riverine and riparian plant species in each river basin creates difficulties in determining the vegetation requirements of the flow regime (Roberts, 2001). “Consequently the vegetation component of a flow regime is usually defined using sparse data for just a few species, supported by general ecological theories” (Roberts, 2001). The accuracy of recommendations will be determined by the available data and the funding available for environmental flows study to collect new data. The difficulty in quantifying uncertainty in environmental flow assessments was demonstrated by Stewardson and Rutherford (2006), who systematically assessed the uncertainty in one element (flushing flow) of an environmental flow regime. They found that the 95% confidence limits on the flushing flow recommendation of 157 GL/year were 10GL/year and 360 GL/yr, a confidence interval of 350GL/yr. With only a few extra measurements or data requirements this interval could be significantly reduced. The size of these uncertainties, and the cost and difficulty in quantifying them, has ramifications for investment decisions in environmental flows. Any decision to allocate water to the environment must consider the level of certainty in return on investment.

Apart from the difficulties presented by limited understanding of some aspects of environment’s water needs (Ladson & Finlayson, 2002, p. 5), there are further complications caused by the eco-systems’ varying water requirements over a season and between locations (Environment and Natural Resources Committee, 2003). Even when the environment flows have been determined, there will be difficulty in determining the volumes of water required to ensure these flows. Note that water instream for consumptive deliveries also provides an environmental benefit. The additional environmental volume is what is required instream, over and above the existing instream
volume. In some cases, the environment may be impacted by too much water instream due to irrigation or consumptive deliveries. In turn, the volume instream is based on the decisions made by other water users (how much they have purchased, when they have asked for deliveries etc).

2.4.3. Valuing environmental water

While instream flows generate a range of ecosystem services (Table 2-2), the economic view is that the value of an ecosystem is not valuable in itself, but only in the services it provides to humans. The value of their ecological outcomes will be reflected in the values placed on them by the community.

Table 2-2: Benefits generated by the provision of instream flows (Colby, 1990; Wills, 2006)

<table>
<thead>
<tr>
<th>Benefit</th>
<th>Description</th>
<th>Example</th>
</tr>
</thead>
<tbody>
<tr>
<td>Direct-Use</td>
<td>Based on conscious use of environmental assets (consumption or production activities)</td>
<td>Recreational value</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Local economic development (i.e. tourism)</td>
</tr>
<tr>
<td>Indirect-Use</td>
<td>Based on the contributions of natural resources to support human life</td>
<td>Water Quality enhancements</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Nutrient cycling</td>
</tr>
<tr>
<td>Non-Use</td>
<td>No current tangible interaction</td>
<td>Benefits of preserving so that one has the option to enjoy &quot;option value&quot;</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Willingness to pay for future generations &quot;bequest value&quot;</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Knowing site exists &quot;knowledge value&quot;.</td>
</tr>
</tbody>
</table>

Each ecosystem service requires different flow regimes, which may compete. For example, water quality enhancement may require large flows to flush nutrients from the system. On the other hand, maintaining minimum flow was of more value to recreationists than additional increments to already adequate flow (Daubert & Young, 1981). Non-use values are more related to preservation of the ecosystem and thus require a range of flow components based on ecosystem requirements. Most non-use benefits have public good properties, and as a consequence, there are no markets in place to reveal price and there is little direct market evidence on the willingness to pay for environmental flows. Economists therefore commonly use non-market valuation techniques (such as contingent valuation and choice modelling) to infer willingness to pay for environmental outcomes. There are many texts that provide detailed descriptions of non-market
valuation techniques (including Chee, 2004; Wills, 2006). A brief summary of the common methods and their application is provided in Table 2-3 and Table 2-4.

Table 2-3: Methods for estimating unpriced values (Wills, 2006)

<table>
<thead>
<tr>
<th>Based on</th>
<th>Methods for estimating unpriced values (Wills, 2006)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Based on</td>
</tr>
<tr>
<td></td>
<td>Observed behaviour of beneficiaries</td>
</tr>
<tr>
<td>Direct-Use</td>
<td>Exclusion to create a market</td>
</tr>
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<td></td>
<td>Travel Cost</td>
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<td></td>
<td>Hedonic price analysis</td>
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<tr>
<td>Indirect-Use</td>
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<tr>
<td>Non-Use</td>
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</tbody>
</table>

Table 2-4: Methods for estimating unpriced values (Chee, 2004; Wills, 2006)

<table>
<thead>
<tr>
<th>Based on</th>
<th>Basis of approach</th>
<th>Main techniques</th>
<th>Method</th>
<th>Example uses</th>
<th>Advantages / Disadvantages</th>
</tr>
</thead>
<tbody>
<tr>
<td>Observed behaviour of beneficiaries</td>
<td>Revealed preference</td>
<td>Travel cost method</td>
<td>Evaluates value where consumption is commensurate with the cost to travel to acquire it.</td>
<td>Most commonly applied to outdoor recreation</td>
<td>Tends to only include visible, commonly appreciated values (direct use values). Assumption that travel time equated to value of site is not always valid (eg. Some people may live close to the site because of the value they place on it) Better at estimating value of a site that remains in a constant state, not at estimating the impact changes in value.</td>
</tr>
<tr>
<td>Hedonic Pricing</td>
<td>Value of an asset is based on the valued characteristics of that asset. The economic value of a single characteristic (or ecological component) of the service is derived from the market price of the service using a regression of service value with various units of the relevant characteristics.</td>
<td>Mainly applied in real-estate markets to estimate the contribution of environmental amenities on land and housing values</td>
<td>The method implies that there exists a set of measurable attributes that will predict the price of a commodity when it is traded. Finding suitable variables to measure environmental attributes can be problematic. Large amounts of data required.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Based on</td>
<td>Basis of approach</td>
<td>Main techniques</td>
<td>Method</td>
<td>Example uses</td>
<td>Advantages / Disadvantages</td>
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</tr>
<tr>
<td>Hypothetical responses of beneficiaries</td>
<td>Stated preference</td>
<td>Contingent valuation</td>
<td>Based on a hypothetical market in which people are asked through questionnaires or interviews, their willingness to pay for a certain environmental good/service. This makes it capable of eliciting monetary value for goods that have no exchange value.</td>
<td>Valuation of wildlife, erosion control</td>
<td>Some reservation as it is not based on actual market behaviour Bias depending on survey design Level of understanding and prior knowledge of those surveyed has a strong influence on outcomes. Strongly influenced by income and education level of survey group.</td>
</tr>
<tr>
<td>Choice modelling</td>
<td>Rather than asking survey respondents to put a direct money value on a certain environmental good, they are asked to make a series of choices between alternative scenarios.</td>
<td>To compare environmental policy options</td>
<td>As above, however choice modelling can reduce some of these difficulties.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Estimated gains from exchange / expert judgement</td>
<td>Production approach</td>
<td>Production function</td>
<td>Based on estimating the contribution an ecosystem service makes to some marketed or marketable service.</td>
<td>Impact of mangrove clearing on fishing industry</td>
<td>Lack of data on cause-effect relationship between ecosystem being valued and marketable commodity Difficult to account for interconnectivity of ecosystems and therefore a tendency to double count</td>
</tr>
<tr>
<td>Constructed value</td>
<td>Assesses value based on how much it would cost to replace or restore it after it had been damaged.</td>
<td>Site rehabilitation cost after mining.</td>
<td>From an economic perspective the optimum level of restoration must be determined by the value to society. Thus the monetary values derived are only valid if society is willing to incur the cost should the ecosystem service not exist.</td>
<td></td>
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</tr>
</tbody>
</table>

There is a wide body of literature applying valuation techniques to environmental flows and other water uses. In the United States, the majority of water resource studies conducted by the US Bureau of Reclamation (the main organisational body reclaiming water for the environment) require some form of economic analysis to justify proposed actions. Platt (2001) provides a summary of these environmental flow valuation projects, but the different ecology of the river systems and potentially different attitudes of the communities make it difficult to transfer the project outcomes to the Australian
environment and context. While the method of contingent valuation is transferrable, and there are lessons to be learnt from the USA on applying these methods, the data itself is not transferrable.

While there is much discussion of the theoretical application of non-market valuation techniques to provision of environmental flow in Australia (for example Bennett, 2002), there are limited examples of the application in reality. The most comprehensive study valued New South Wales rivers for use in benefit transfer (Morrison & Bennett, 2004). In this study, choice modelling surveys were conducted to value improved river health across seven rivers in New South Wales. Morrison and Bennett then developed a method to transfer the outcomes of these studies to be used to value improved river health in other streams across New South Wales. The outcomes of this study have also been applied to catchments in Victoria including the Thomson and Macalister environmental flows study (Branson et al., 2005) and the Stringybark Creek environmental flows assessment (URS, 2006).

While non market valuation techniques provide insight to the willingness to pay for providing environmental flows (and remain the most widely used method), there are limitations and scepticism amongst many stakeholders.

- Non market valuation studies provide a large scatter of results (Briscoe, 1996) reducing their accuracy and precluding their use in making tradeoff decisions at the time and spatial scale required to develop environmental demand curves.
- It is an expensive process to survey every river system to determine environmental values. While benefit transfer techniques are available, there is limited data currently available in most areas. The more removed the initial data set, the less likely the benefit transfer outcomes will reflect the real situation.
- Contingent valuation techniques assess the value placed on the environment by the community at a single point in time. These values will change with time, and with the political climate. For example, as the wealth of the community increases, it is likely that the environment will be valued more highly.
2.4.4. Understanding tradeoffs of providing environmental flows

Most environmental flow studies undertaken in Australia relate to the “command and control” regulatory approach. They recommend a fixed environmental target (albeit different at different times of year) and the cost to the community of meeting this is assessed. These are decisions made for long term resource planning rather than day to day operational decisions. The usual method is an iterative process, where an expert panel examines a range of different flow scenarios and balances the environmental gain against economic loss.

This approach is designed to address trade-offs between the needs of the abstractor and the environment...the disadvantage is that the approach does not provide an unambiguous decision. Indeed, the water manager needs to decide on the scenario to employ, which often requires a political decision concerning acceptable levels of environmental degradation for the given economic implications” (Acreman, 2005, p. 103).

One of the difficulties in reviewing this topic is that much of the decision making process is documented in grey literature (either unpublished or government reporting), which is not readily available.

Regional priorities for river management are established in regional planning documents such as River Health Strategies in Victoria. These documents provide a framework to integrate all actions for river management to ensure that rivers of high quality are protected and that ecological condition of other rivers are improved. The Regional River Health Strategies identify “High Priority Reaches” based on the “greatest value to the community”, and rivers that are currently “ecologically healthy” (Goulburn Broken CMA, 2003). Factors such as the presence of endangered species, their national and regional significance and whether a reach is associated with an international or nationally significant wetland are used to determine whether a reach is of high priority. Once high priority reaches are identified and their management systems are in place, strategies are developed to improve the condition of other rivers (or reaches within a river).
In both the North Central region (SKM, 2006a) and the Glenelg Hopkins region in Victoria, the concept of triage was applied to prioritise flow restoration in unregulated rivers. A key element in considering the probability of recovery of a river reach was whether flow was the factor limiting environmental health. Flow was considered in the context of other threats to environmental condition.

The concept of triage originated as a means of efficiently deploying medical care during battles, so that those most urgently in need of care were accorded priority over those whose injuries were less severe and over those whose injuries were deemed terminal. Two criteria feature centrally in care intervention decisions: degree of threat/need and potential for recovery/reversibility of disease. Hobbs and Kristjanson (2003) have written on the application of this approach to “health care” for landscapes. The concept of triage as it has been applied in natural resource management is shown in Figure 2-4.

The Thomson–Macalister environmental flows studies presented their environmental expert panel with a series of flow scenarios (The Thomson Macalister Environmental Flows Task Force, 2004). The panel was asked to make tradeoffs in providing environmental flows, giving certain environmental flow elements lower priorities. Their decisions seem to be based on the current availability (and political availability) of water in the catchment and would thus need review if flow conditions in the river changed.

The DRIFT method (King et al., 2003) is a more rigorous use of an expert panel in assessing scenarios and making tradeoff decisions between consumptive benefits and
natural resource degradation (King & Brown, 2006). DRIFT is a holistic method that assigns severity ratings to changes resulting from different flow management scenarios. Stewardson (2005) applied this concept to the environmental flow statistics used commonly in Victorian environmental flows studies. Stewardson suggested drawing thresholds lines on relevant plots to indicate the point where further departures from the natural regime will lead to an increasingly severe environmental impact (Figure 2-5 and Table 2-5).
Figure 2-5: Hypothetical Severity Ratings from Stewardson (2005). Key for lines indicating severity ratings is given in Table 2-5.

Table 2-5: Severity Ratings (Stewardson, 2005).

<table>
<thead>
<tr>
<th>Severity rating</th>
<th>Severity of change (from reference condition)</th>
<th>Equivalent loss of abundance (relative to reference)</th>
<th>Key for lines shown in Figure 2-5</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>None</td>
<td>No change</td>
<td>---------------------------------</td>
</tr>
<tr>
<td>1</td>
<td>Negligible</td>
<td>0% - 20% reduction</td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>Low</td>
<td>20% - 40% reduction</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Moderate</td>
<td>40% - 60% reduction</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>Severe</td>
<td>60% - 80% reduction</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>Critically severe</td>
<td>80% - 100% reduction (includes local extinctions)</td>
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</tr>
</tbody>
</table>
The Murray Flow Assessment Tool (MFAT), while addressing only the ecological outcomes of different flow regimes, provides a transparent approach to assigning ecological risk. MFAT is a software tool developed to assess the ecological outcomes of different flow scenarios along the Murray River (Young et al., 2003). The ecological outcomes are assessed in terms of the physical habitat condition for native fish, waterbirds and vegetation communities and growth of algal blooms. Daily flow data is entered into the model, which then uses a series of ecological modules to derive an output of ecological assessment indices. The ecological assessment is based around a series of preference curves with the x-axis defined by a flow element (for example daily water depth, flow percentile, duration of inundation) and the y-axis a non-dimensional index where zero indicates intolerable habitat conditions and one represents ideal habitat conditions. A range of different flow elements defines the ecological requirements of a number of groups of species. A series of weightings are used to mark the relative importance of each element. The assessment points along the River Murray can also be weighted according to relative importance. Different flow regimes in the Murray River can be assessed based on ecological outcome using MFAT. The difference from the previously discussed projects is that the expert panel was used in MFAT to determine the ecological preference curves, and these are then used in the modelling framework to make transparent assessments of different flow regimes. The previously discussed methods rely on expert panels to make the trade off decisions.

Dudley et al. (1998) used an optimisation approach to minimise tradeoffs between the environment and consumptive users. Trade-off curves were developed between commercial outcomes and environmental effectiveness (as shown in Figure 2-6). When water is scarce, one gains at the expense of the other. The modelling was used to establish the highest mutual benefit attainable. We can relate the approach from Dudley et al. (1998) back to the concept of marginal value of water to irrigators and the environment. The technical feasibility trade-off curve (Figure 2-6) is often referred to in economics as the Production Possibility Frontier (PPF). This represents the boundary of the attainable combination of environmental and irrigation outputs, based on the volume of water available (and other inputs). The slope of this (or the derivative) is known as the marginal rate of transformation. In other words, for an extra unit of irrigation output,
how much environmental value is lost? The \textit{society indifference curve} (also called a preference curve) shows societies preferences for combinations of environmental and irrigation outputs. The \textit{marginal rate of substitution} represents the slope of society’s indifference curve. In other words, it measures the rate at which society is just on the margin of moving water from consumptive use to environmental use (or vice versa)$^5$. At the point where the production possibility frontier and the indifference curve are tangent (in other words, where the marginal rate of transformation is equal to the marginal rate of substitution), the marginal social benefit of consumptive allocation is equal to the marginal social benefit of environmental allocation, (MSBc = MSBe). This defines how water should be allocated between the two uses.

Dudley \textit{et al.} (1998) determined “Environmental effectiveness” based on representative flow statistics to show deviation from natural conditions (for example mean and median monthly flows, percentile flows, spells analysis). Similar hydrological indicators to represent likely ecological condition have been used in a range of studies (for example Richter \textit{et al.}, 1996; SKM, 2005).

\begin{figure}[h]
\centering
\includegraphics[width=0.7\textwidth]{trade-off_curve.png}
\caption{Illustrative example of a trade-off curve (adapted from Dudley \textit{et al.}, 1998)}
\end{figure}

$^5$ A good description of preference curves and marginal rate of substitution can be found in (Varin, 2006).
The method to determine tradeoffs between ecological outcomes and consumptive users varies greatly between basins. However, in all cases, the approaches depended on subjective, albeit expert, opinion with documentation of these decisions varied across projects. The approaches also tend to suit long term planning decisions and regulated flow requirements rather than adaptive and day to day operational decisions of an environmental manager.

2.5. Options for gaining environmental water

Typically, one or more of the following legislative instruments provides water for the environment:

- **A) Operational rules:** Conditions, rules and restrictions are imposed on entitlement holders for consumptive use (or water authorities) to maintain specified flow regimes within rivers;

- **B) Environmental entitlement:** Water is provided in the form of a specific entitlement (share of the resource) and defined like other water entitlements; and

- **C) Resource cap:** Environmental requirements are determined before defining the volume of water available for other entitlements for consumptive uses.

These legislative mechanisms for providing water to the environment (operational rules, environmental entitlements and resource caps) often exist concurrently in individual river systems. Decreased water availability and increased supply variability, under these differing mechanisms, present significantly differing risks for the environment. This is demonstrated in Figure 2-7, which shows a flow series and the three components that make up the total water allocated to the environment in any one year: Under the operational rules or license conditions, environmental water (shown by area A in Figure 2-7) is provided at a higher priority than all other water users and is the minimum volume that must remain instream before consumptive users can extract water. Entitlements can be allocated to both consumptive users and the environment (shown as allocated water, component B in Figure 2-7). The environmental water allocated as an environmental entitlement in Component (B) has the same conditions, priority and reliability as other consumptive entitlements in Component (B). Only this portion of the environmental
water can be traded as a “property” or “entitlement” and can be directly transferred to or from other consumptive entitlements. If the system is capped, any volume of water exceeding the cap is left instream for the environment (shown as area C in Figure 2-7). This component of the environmental allocation is the lowest priority, and it has the lowest reliability in the allocation system. Where consumptive diversions are capped, any instream flows over and above the cap provide some of the natural variability required by aquatic environments and form part of the current allocation of water to the environment. If however, catchment inflows change, and therefore the available water resources change (for example through reduction as a result of climate change), the environment’s unaccounted allocation is the first to be impacted.

![Figure 2-7: Components of Environmental Water Allocation](image)

Efforts to increase the environmental water reserve which focus on increasing environmental entitlements rather than adjusting the cap or changing conditions on consumptive users have a number of advantages. The environmental entitlement is more reliable than the volume above the cap, and has more flexibility for adaptive management than the volume of water protected by operational rules. This means that the water can be used to meet a wider range of environmental flow objectives, not just minimum flow requirements. However, it also ensures that clear decisions are made on an ongoing basis about the benefit of using the environmental water at a particular location and time (which is not necessarily the case for regulated minimum flows protected through licence conditions).
Assuming environmental water reserve is increased through changes in the entitlement volume, there are three options for sourcing water. An EWM can compulsorily acquire from existing water holders (with compensation), can enter the water market and purchase from willing sellers, or can fund infrastructure projects to make more water available. These options are summarized in Table 2-6, with each of the mechanisms discussed against a set of criteria; political viability, market signals, flexibility and transition.

Table 2-6: Mechanisms to increase the environmental water reserve

<table>
<thead>
<tr>
<th>Mechanisms</th>
<th>Criteria</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Political Viability</td>
</tr>
<tr>
<td>Compulsory buy-back</td>
<td>Unpopular</td>
</tr>
<tr>
<td>Water savings projects</td>
<td>Widely accepted at on-farm scale Varied response for larger projects</td>
</tr>
<tr>
<td>Water market</td>
<td>Limited acceptance (although growing)</td>
</tr>
</tbody>
</table>

2.5.1. Compulsory buy-back

An option discussed widely in the media is compulsory buy-back of water rights. The government has to date ruled out this option. A compulsory buy-back could be structured in a number of ways, for example requiring every irrigator to transfer a set percentage of their allocation to the environmental water manager, or alternatively running a compulsory tender process (Goesch & Heaney, 2003).
Compulsory acquisition of water rights, regardless of the compensation provided, would create large uncertainty amongst the irrigation community in terms of the value and security of their water rights. Irrigators have already made investment decisions based on their current water rights. If there is less certainty of underlying water rights, there will be reduced confidence around future long term investments in infrastructure.

2.5.2. Water savings projects

Governments have funded infrastructure projects that save water, with realised savings secured for the environment. These have ranged from on-farm water savings projects (such as funded through the federal government Water through Efficiency tender (DAFF, 2006)) to large infrastructure projects (such as the proposed Food Bowl Modernisation Project, (Food Bowl Modernisation Project Steering Committee, 2007)).

At the farm level, the concept is that government funds irrigation efficiency improvements (such as moving from flood irrigation to drip irrigation) and a volume of the irrigators’ entitlement is then transferred to an Environmental Entitlement. These programs are often run through tender processes where farmers nominate the volume of water they are willing to transfer and nominate a price based on their own cost estimates for on-farm efficiency measures. This is particularly effective in unregulated systems (that is, with no upstream storage) where the existing water market is relatively thin and thus an EWM must create a market to source water. It also, theoretically, allows an EWM to assess the proposed on-farm efficiency project in terms of the timing of irrigation and relate this back to the timing of environmental water requirements. The government can ask for more information than just price to assess the different tenders.

*Buyers can elicit much more information from irrigators than would normally be available when purchasing water in the market. For example, in irrigation districts with highly saline ground water, preference could be given to bids by irrigators irrigating less than 3 kilometres from the river, or irrigators without reuse systems* (Goesch & Heaney, 2003, p. 12).

On-farm projects place the onus on the landowner to determine the volume of water saved through the improvements and subsequently nominates the volume of entitlement
transferred to the environment. However, the extent of connectivity of groundwater and surface water and the proposed on-farm measures provide risk that the water use efficiency project may also result in a reduced return flow to the river. Tradable property rights must be exclusive and rival. Numerous writers have examined the extent to which water entitlements (for both consumptive users and the environment) are adequately defined (Campbell, 2004; Freebairn & Quiggin, 2006; Siebert et al., 2000; Young & McColl, 2005).

Licences are currently specified for gross water diversions rather than net water usage (Productivity Commission, 2003, p. XXIV). This means that irrigators that improve on-farm efficiency can increase their net water usage. The hydrology of systems (surface and subsurface) is linked, so that most of the extracted water not used by crops or lost through evapotranspiration will return to the main river system through subsurface drainage and become available for other water users. If on-farm efficiency improves, the return flows are reduced, potentially impacting on downstream water users. Part of the current environmental allocation is based on a cap on the consumptive water use (shown by C in Figure 2) whereby water in excess of the cap is left instream for the environment. Using gross water usage to account for consumptive use means that environmental allocations protected by the cap (component C in Figure 2) will be directly impacted if net water usage increases. In other words, the current arrangement for environmental allocation is not exclusive. Any water savings project aimed at improving the environmental allocation must account for this.

Both state and federal governments have also invested in a number of large scale infrastructure projects (for example converting open channels to pipelines) to save water that would have been lost through seepage and evaporation. These projects are considered an investment in rural communities, sharing the water saved between the irrigation community and environment.

One of the main difficulties in using water savings projects to source water for the environment is determining the volume of water saved. Improving infrastructure can lead to a net increase in water through reductions in seepage and evaporation. These losses
are difficult to estimate and there is currently no consistent approach. The method used to calculated system losses is based often on a water balance across the reach (or channel) and thus incorporates the uncertainties in all data collected for the reach (some of which is often unmetered). It is also difficult to accurately asses how much of the seepage component would return to rivers through the aquifer system. By way of example, according to the Foodbowl modernization project in Victoria:

*When the whole system is modernised then the saved water could represent more than 800 GL (less some unknown amount that is currently de-facto recycling through drainage outfall reuse downstream)* (Food Bowl Modernisation Project Steering Committee, 2007).

The Auditor general later reported that the water savings figures were not clearly justified (Victorian Auditor General, 2008, p. 32).

It is difficult to compare the price of sourcing water saved through these projects with other water from other sources when the volume of water gained is so uncertain. One could also question making an investment when the returns on investment cannot be clearly assessed because the risk of not delivering the expected outcome is unknown. The most recent figures for the Foodbowl project suggest that Stage 1 will cost $1 billion, saving 225 GL ($4444/ML) and Stage 2 will cost $1 billion for a saving of 200 GL ($5000/ML). ACIL reported that other water savings projects cost in the range of $200 to $7500 per megalitre (ACIL Tasman, 2008).

Assessment of water savings from large scale infrastructure projects is often based on the current operation of the system. Care must be taken to assess the risk in committing to large scale infrastructure projects or buybacks because the investment may actually alter the water market trades and hence change the actual savings gained (The Wentworth Group, 2003).

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6 As an indication, water purchased on the permanent market at the start of 2008 was around $2000 / ML (ABC News Online, 2008)
2.5.3. Entering the water market

The NWI and the 2007 Water Act both emphasised the need to recognise the environment as a water user that could trade with consumptive users. Examples of the government entering the water market to purchase water for the environment include the Living Murray Initiative, Waterfind Environment Fund, and NSW Riverbank. However, there has been limited participation by the environment in water markets because of specific restrictions limiting it from entering the market\(^7\) and because of the lack of clearly defined, tradable environmental water entitlements (Siebert et al., 2000). By excluding the environment, the water market is incomplete because the market price will not reflect the true opportunity cost of water and therefore, does not result in an efficient outcome.

There are a number of other advantages in using market mechanisms to provide water for the environment. By allowing the environment to participate in the market, environmental goals could be achieved more effectively and at a lower cost to the community (Productivity Commission, 2006, p. 76). To date, governments have preferred to obtain water for the environment by funding or subsidising water saving projects (Marsden Jacob Associates, 2003, p. 19). The government is estimated to have spent $5000/megalitre subsidising water saving projects to secure water for the environment, which is around four times the market price, implying that purchasing water entitlements in the market would have been more cost effective (Cosier, 2006). If instead, this water for the environment was purchased, the market price would indicate the value of this water to the complete set of rights holders. Any increase in the market price would encourage private investment in water saving technology or technology to

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\(^7\) For example, South Australian water policy states that trade in environmental water should not occur (ACCC, 2006b) Other Barriers that are relevant to an EWM include restrictions on water entitlement ownership and constraints on water exports. By way of example, Victorian legislation limits the percentage of Victorian water entitlements that can be held by non Victorian landholders to 10% of the total volume of Victorian water entitlements. Similarly a 4% limit on net outward permanent trade exists in the rules of most irrigation operators. Purchases by the EWM will fall within these restricted categories. While it may not be the intended consequence these rules directly impact on an EWM’s ability to purchase water for the environment.
increase the effective water supply. In the absence of clear market failures, this decision should be left to private investors rather than to the public sector.

While the majority of irrigators agree with the allocation of water to the environment (Syme et al., 1999), it was important that the allocation was “fair”. Interestingly, the perception that allocation to the environment was “fair” was considered more important than any economic loss. Allowing an environmental purchaser to participate in the market would ensure that transactions only occur if both parties are in agreement. Markets may thus be better able to fulfil the perception of a “fair” allocation of water.

Environmental water requirements change from year to year depending on annual runoff and the required flow regime. As scientific knowledge around ecosystem requirements advances, it is also likely that understanding of the volume, timing, and duration of water required for the environment will improve. As society’s wealth increases, the value placed on the environment is likely to increase and thus society will be prepared to invest more in protecting the environment. Adaptive management to accommodate these changes is more easily achieved through the market rather than through regulatory approaches.

Water entitlements often differ in their access and supply reliability characteristics. The utility of market participants can increase, purely by free trade improving access to non-homogenous products i.e. greater product choice (for example Dixit & Stiglitz, 1977; Helpman & Krugman, 1985). This concept is relevant in the context of environmental water trade. As stated previously, the environment typically requires variability in flow and the flexibility to manage environmental risks under varying natural conditions. The market could therefore provide an EWM with the flexibility to better manage environmental flows, and in particular allow response to changing circumstances, by increasing the flexibility and the specificity with which water of particular characteristics can be procured (Grafton et al., 2004, p. 43). With this opportunity, EWMs may choose to hold a portfolio of water products. The Productivity Commission (2006) provide a good outline of water products (for example options, leases, temporary and permanent purchases) and the benefit they would provide an EWM.
The existing property rights to water have been designed for consumptive water users. It is easier to define a property right as “rival” if it has been extracted from the river system. However, many environmental allocations remain instream. It becomes difficult to define how far downstream these entitlements should still be protected. For example, if environmental allocations are used off stream (for example, to water a wetland) and returned to the river system, can they still be protected as an environmental entitlement to be used again to maintain downstream river health?

The environmental outcomes that can be achieved through the market may be hindered by problems associated with coordinating the actions of various state based EWMs. For the most part legislative control of water rests with the states, reflecting the division of powers established by the Commonwealth Constitution, with private parties granted rights to access water for various purposes. Allocating and managing environmental water varies between state jurisdictions, notwithstanding that many of these jurisdictions are hydrologically connected.

The Commonwealth Water Act (2007) has the objective of addressing the issues of managing a single hydrological basin across a range of state borders. However, while the Act establishes a Commonwealth environmental water holder, responsibility for providing environmental water still rests with the states. While the water act will provide some over arching coordination, individual states will still hold environmental water entitlements separate to the Commonwealth account.

There are broader implications associated with the environment entering the market. Consideration should be given to whether some of the existing policies for providing water for the environment impact on transaction costs. Ensuring that environmental flows and regimes are provided through the use of various operative rules imposes the need for monitoring trade to ensure its compliance with these rules. Where the trade

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8 Under the Environmental Protection and Biodiversity Conservation Act 1999, the Australian Federal Government can control the taking, use and returns of water that adversely affect an environmental matter of national significance. Also under the Commonwealth’s external powers it can be responsible for natural resource management as it applies to Australia’s international obligations and International Environmental Conventions. The Commonwealth used these powers and others to take control of some aspects water management in the Murray Darling Basin through the Water Act 2007.
approval process involves a charge being levied to cover the cost of undertaking the approval and/or delays the approval process, the transaction cost on all market participants reduces the efficiency of the market outcome.

If an EWM could instead use the market to counteract any impacts on flows as they occur, the approval process and thus transaction costs could be reduced. Of course, this may require substantial ongoing monitoring in itself and where there is a delay between the time issues are identified and then resolved through the market, the negative impact may already have occurred.

If an EWM enters the water market to reduce the extent of over-allocation in a catchment, this would have an impact on the market price of water entitlements, due to the increase in demand for entitlements. The extent of this price increase would depend on the supply elasticity in the market and the relative size of any purchases. The intended purchases of the government are significant, and therefore, a price increase is inevitable. However, any increase in the market price as a result of the environment’s participation does not in itself represent a poor outcome. Rather the reallocation of water and any associated adjustments represent an efficient outcome which should provide a signal to the private sector to independently increase investment in water saving technology and in other water supply augmentation options (Freebairn, 2005, p. 4).

While purchasing water on the market may lead to the cheapest option for reclaiming water and reveal correct signals in the market, there may be additional social and economic implications if large volumes of water leave an irrigation district. Irrigation authorities may be left with stranded irrigation infrastructure or maintaining infrastructure for only a small number of remaining irrigators. There is currently a cap of 4% on permanent trade out of any given irrigation district within a year to try to minimise the impact of water leaving a region. There are also exit fees in place if water is sold from a district. The ACCC has undertaken a review on structuring these exit fees (ACCC, 2006a). The reduction in irrigated activities in a district will also have flow-on effects to agricultural suppliers and other local businesses (Goesch & Heaney, 2003).
2.5.4. Optimisation approach to least cost water purchases

There have been many research exercises to understand the least cost approach to providing a fixed environmental flow target using optimisation and simulation modelling techniques\(^9\).

ABARE developed the Murrumbidgee River Options Model (MROM) to explore the use of different types of options contracts and how they could be used to meet high flow environmental events in the Murrumbidgee River (Beare et al., 2004; Hafi et al., 2005). High flow events are not required every year and thus can be provided by less reliable water. The options model combined a hydrological module, storage and operations module, agronomic module and a crop allocation module. The model shows the price and volume of water available at different allocation thresholds for the options contracts. The options are then valued against alternative water supply options (such as permanent entitlements).

BDA Group (2006) looked at the use of market buy backs to provide environmental water in the Murray river and examined two types of environmental water demands: a fixed environmental demand representing minimum flow requirements and a variable environmental demand representing high flow (or less frequent events). Analysis of the watering requirements for the Icon sites\(^{10}\) in the Murray River indicated that additional volumes of water required ranged between 0 and 310 GL per year. The study showed that while a portfolio of water products may be more expensive for an environmental manager, there was a greater likelihood of meeting the variable environmental demand than when allocations or entitlements were purchased in isolation.

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\(^9\) “Simulation and optimisation models are the two basic categories of water resources models. The main difference between these two groups of models is that the optimal allocation water is independently determined for each one-time interval of analysis in the first model while in the second, the analysis is carried out for a multi-period interval.” Optimisation modelling aims to find \(x\) to maximise \(y=f(x)\) subject to a range of constraints. Simulation modelling varies the value of \(x\) and identifies the effect on \(y=f(x)\) and then the user decides the better set of \(x\) values (Reca et al., 2001).

\(^{10}\) The Living Murray project refers to “Icon sites” that are priorities for environmental water and management.
Hollinshead and Lund (2006) constructed a three stage linear optimisation model to identify strategies for seasonal water purchases for application to California’s Environmental Water Account (EWA). The optimisation model minimises the average cost of EWA purchase over a single year by looking at the combination and timing of long-term, spot market and option purchases. The EWA protects fish in the San Francisco Bay and Sacramento-San Joaquin Delta by reducing consumptive pumping and instead purchasing water on the market to reimburse consumptive demands for foregone pumping. Purchase decisions occur in three stages, with increasing information (climate and other constraints) becoming available as the year progresses. Water year type (climate condition) was found to be the best predictor of likely EWA costs.

While these existing studies contribute to determining the best use of markets to achieve a certain environmental outcome, these studies are limited by the assumption that environmental demand remains constant. The models do not account for how environmental demands change over time and how these demands compare to consumptive demands for water. Crop functions to determine crop water demands have been used as inputs to optimisation of irrigation allocations (for example, Reca et al., 2001). A similar approach could be applied to determining environmental water demands.

2.5.5. **Structure of entitlements**

Assuming the environment will benefit from entering the water market, it is critical to clearly define the environment’s right to water by creating specific statutory entitlements. Universality of water entitlements allows trade to occur between the Environment and consumptive users. Currently only one of three component of the environment’s total allocation is provided this way (i.e. component B in Figure 2). Converting all the water rights of the environment into specific tradable entitlements may create legislatively simpler trading arrangements, but also requires explicit statement of the priority given to these environmental entitlements. For example, water allocated to the environment as a result of the cap on diversions could be created as a low priority (or lower reliability) entitlement as per current outcomes. Similarly water provided through license conditions
could be created as a high priority entitlement. Stating environmental rights in this way also adds to the transparency of governance arrangements. Even when considering consumptive water entitlement holders, no state has water entitlements that are universal. For example, most do not integrate the management of surface and ground water sources (Poff et al., 1997).

For consumptive water users, the aim is to minimize cost, whilst maximizing the combination of a number of attributes including reliability of supply and quality of water. Environmental water requirements are more complicated and variable, and therefore one form of property right may not suit all environmental needs. Recall that the environmental flow requirements comprise a range of different flow components, and each of these flow components can be described by their “magnitude, frequency, duration, timing, and rate of change” (Productivity Commission, 2006). The current water allocation system has been designed to meet consumptive needs, and therefore, focuses primarily on the magnitude of water available in a year. While all water demands are variable, this is particularly the case for the environment. The volume of water and reliability of water required will vary substantially between years. A range of water products may be required to meet this variable requirement (including the frequency, duration, timing and rate of change) in a timely and cost effective way (ACCC, 2006b; Grafton et al., 2004). However, this is not a criticism of the virtues of universality or the benefits of the environment’s entitlements being more clearly defined. Rather, it highlights that there may be benefits in the environment holding a range of water products (i.e. low and high reliability entitlements) as is explicitly the case in current arrangements for consumptive water users. Often water entitlements have different access and supply reliability characteristics (for example high security and low security entitlements).

Many states have begun to ‘unbundle’ water rights. The first step in this process has been to separate water rights from land titles. Historically, the water allocation system was based on riparian access, thus linking access to a water licence to ownership of land. The unbundling process allows non-landholders to purchase or hold water rights, providing government, non-governmental organisations, or individuals the opportunity to hold
water rights on behalf of the environment without owning land. However, in South Australia, NSW, Queensland and Victoria, legislation appears to restrict/limit ownership of water entitlement by non-water users/non-landholders, and therefore by default, an EWM. This issue has become increasingly politicised with the Federal governments agenda to buy-back water for the Murray River clashing with state legislation restricting the volume of water held by non-water users and limiting trade out of irrigation districts (Ker, 2008).

A further advantage of unbundling is the potential to cost effectively address environmental requirements in terms of frequency, duration and timing of flows. The frequency of a flow component can be linked to the reliability of an allocation. For example, an environmental flow component that is required every year will need a higher reliability allocation than a component that is required only once every five years. The duration, timing of flows and rates of rise and fall relate to the ability of an EWM to manage releases from storage and the volume of water that can be held in storage. Due to the flow magnitude of high flow events required by the environment, the ability to carry over water in storage would significantly increase the likelihood of the EWM being able to manage for these events. An alternative to this would be to use a combination of permanent and temporary purchases (or even options) to build up a portfolio of different water products.

Further stages in unbundling may include transforming the current water rights into capacity share arrangements for storages and delivery infrastructure, and ensuring usage licences reflect differences in the external costs of water usage and waste disposal for different water uses and locations. The same concept can be applied to capacity shares of rivers that are used to deliver irrigation water. A key concern in regulated11 river systems is that summer flows are unseasonably high due to irrigation transfers. Under the current licensing arrangements, there is no mechanism for the environment to reduce flows in a stream as existing market mechanisms only allow for purchasing additional water. The

11 Recall that in this context a regulated system refers to a river that has an upstream storage, where downstream flows are regulated by storage release.
creation of a capacity share along the river reach would allow an environmental manager
to instigate maximum flow restrictions to ensure high summer irrigation flows do not
cause adverse impact on the environment.

The environment requires more flexibility in water products than is currently available. Further unbundling of water rights may address some of these issues and allow an EWM to target specific flow requirements and cost effectively meet variable demands. Table 2-7 summarises the issues in defining property rights to meet environmental requirements.

<table>
<thead>
<tr>
<th>Environmental Requirements</th>
<th>Potential solutions</th>
<th>Issues to be overcome</th>
</tr>
</thead>
</table>
| Large inter-annual variability | Carry over storage  
Purchase of temporary water | Managing potential spills  
Difficulty in purchasing adequate volume and potential price impact of large purchase |
| Varying frequency of flow components | Portfolio of water products, High and Low security water | Increased transaction costs of managing multiple products |
| Varying timing and duration of flow events | Capacity shares in storage and river channels | Knowledge of instream flows and required release depends on decisions from other water users |
| Universality of entitlements (ability to trade with consumptive users): | | |
| (i) Exclusivity and rivalry | Change to net water usage rather than gross water usage | Increase in transaction costs |
| (ii) Security of entitlement (e.g. from climate change) | Change of current Cap to a formal entitlement so that risk is shared across all water users | |

### 2.5.6. Regulated versus unregulated systems

The literature that discusses the provision of environmental flows mainly focuses on increasing environmental allocations in regulated systems. However, increasing environmental flows in unregulated systems posses a number of different problems:

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12 Note that in this context a regulated system refers to a river that has an upstream storage, where downstream flows are regulated by storage release.
Water markets in unregulated systems are either thin or non-existent. This means that there is less information about water price. Where there is no existing market, other mechanisms may be required to engage land owners.

Purchased water can not be actively managed. In a regulated system, upstream storages allow the environmental water reserve to be actively managed and for water to be released when required. In unregulated systems, water purchased for the environment will remain instream at the time that the consumptive water user would previously have pumped water. There is no capacity to adaptively manage. This requires additional consideration of where and when water can be purchased.

Thus the methods used to increase environmental allocations in regulated systems may not be transferable to unregulated systems. When an unregulated system is selected as a basin of high priority for streamflow restoration, the decision is based on environmental value and level of risk of the whole watershed. Prioritisation for unregulated systems must be on this very broad scale because water purchase will be very opportunistic as very few land-owners will sell water at any given time.

Because flows in unregulated systems cannot be actively managed, and the comparatively thin nature of the water market, the research described in this thesis will focus on environmental release decisions in regulated systems.

2.6. Management of environmental water

To trade in the water market, an EWM will require a corporate governance structure that ensures environmental objectives are met. Three potential governance arrangements for an EWM will be considered; a government department or other entity, an independent statutory authority, and a franchise manager. A general summary of these three options and their associated advantages and disadvantages is contained in Table 2-8.

\[ \text{Table 2-8} \]

\[ \text{While many other variations of an EWM governance structure are possible (for example, a government owned corporation) for the purposes of this discussion, these three have been chosen to highlight the various trade offs present in determining the most appropriate structure. Again, many variants in terms of details can be envisaged for each of the three broad options.} \]
The water industry, like many others, has experienced the broader shift away from direct government provision of services. The government has traditionally been the sole provider because of:

- difficulties in measuring targets and outcomes for the purpose of creating appropriate management incentives and accountability (see section 2.6.1).
- difficulties managing complex interrelated issues associated with the broader public policy goals (see section 2.6.2).

However, when the government plays the role of the EWM it is essentially in a competitive market and this result in:

- inefficiency arising from environmental water management becoming a partisan issue and being used to achieve other unrelated public policy objectives (see section 2.6.3); and
- issues arising from the vertical integration of water planning and ongoing management of environmental water (see section 2.6.5).

The provision of public goods such as water for the environment need not be the sole domain of government. Other structures (such as an independent statutory authority or a franchised manager) provide alternatives that negate some of the issues of government interaction in a competitive water market. However, this will be at the expense of the difficulties raised in relation to non-government provision.
Table 2-8: Summary of potential governance arrangements for environmental water managers

<table>
<thead>
<tr>
<th>Governance structure of EWM</th>
<th>Advantages</th>
<th>Disadvantages/ Issues</th>
</tr>
</thead>
<tbody>
<tr>
<td>Government department or other entity</td>
<td>Public accountability through being ultimately responsible to parliament and voters.</td>
<td>Less incentive to develop efficient operations and economical efficient options for delivering environmental objectives.</td>
</tr>
</tbody>
</table>
|                                                           | Does not require as clearly defined measurable targets for contractual purposes. | Subject to partisanship
|                                                           | Can manage issues associated with the broader public interest.                | Conflicts of interest arising from the vertical integration of water planning and ongoing management of environmental water. |
| Independent Statutory Authority                           | Reduces probability of management of environmental water becoming a partisan issue. | May not be the most appropriate body where dealing with matters relating to the broader public interest. |
| [extent to which the authority is at arms length will be affected by details relating to the structure and governing legislation] | Often greater transparency of processes and the basis for decisions | Vulnerability to funding cuts. |
| Franchised Manager [limited period tender]                | General efficiency benefits arising from competitive procurement.            | Private sector generally more risk adverse, and therefore, may add additional cost margin in the context of climatic and regulatory uncertainty. |
|                                                           | Innovation of new approaches from commercial tender process.                  | Agency costs in forming and monitoring contracts.                                      |
|                                                           |                                                                            | Principal agent problems: Difficulties in defining measurable targets for purposes of defining contractual arrangements and monitoring outcomes. |
|                                                           |                                                                            | May not be able to integrate land and water management under this arrangement.        |

2.6.1. Measurement and Monitoring Issues

The general issues are similar to measuring and monitoring the performance of managers of corporations for the benefit of shareholders, although with additional sources of uncertainty. In a general sense, we have a relationship between desired environmental outcomes (such as areas of forests and wetlands, species numbers and biodiversity, and water quality) and the starting state of the environment, inputs to protect and increase the outcomes under control of the EWM (such as volume of water allocated and utilized for
different environmental components), and a number of other inputs not controllable by the EWM (such as climate).

Performance of an EWM can be ensured by combining *ex ante* rules (measurement and monitoring of the inputs and procedures) and *ex post* monitoring of the observed outcomes. When outcomes are readily observed, *ex post* monitoring provides the most effective means to control an EWM’s performance. Even then, performance will be confounded by non-controllable forces and the variations they produce add complexity and measurement challenges. In many cases environmental outcomes will lag behind the inputs that caused them. Conversely, when the performance attributes are difficult to measure, *ex ante* rules on inputs and procedures may be more efficient (for example the Victorian Environmental Flows Monitoring and Assessment Framework, Cottingham *et al.*, 2005). In practice, government management tends towards *ex ante* rules to monitor performance, with franchise and statutory authorities commonly assessed using *ex post* monitoring.

Monitoring of environmental condition has been widely canvassed and frameworks have been developed to assess the impact of environmental flows (NSW Department of Land and Water Conservation, 2008). These monitoring programs are primarily designed to understand how environmental water achieves predicted outcomes, and applying this knowledge to future management. With this stated aim, there are well documented issues in developing a monitoring framework:

- the scale of changes in flow regimes in the context of total water use and natural flow variation;
- river ecosystems are affected by many factors other than flow;
- some responses to environmental flows may take years or even decades to occur; and
- it is only possible to monitor a small component of these large and highly variable systems (Connor & Young, 2003, p. 16).

There are added complications in developing a program to monitor the performance of an EWM. The first issue is determining the objective of the EWM. The ultimate
performance goal is the delivery of environment amenity (a public good) and providing this cost effectively. This can be phrased in a number of ways. For example, the objective of the EWM is to maximize environmental gain given a set budget, or the objective is to achieve a set environmental outcome at minimum cost. In either scenario, an EWM’s performance is measured against indicators of environmental condition. However complexities and uncertainty in ecological responses to flow may make this measure of performance unfair. Indeed, the uncertainty in the performance criteria may complicate the franchised manager approach. There are however many examples from human resources and economic performance indicators where innovative lag and lead indicators have been developed and incorporated into contractual arrangements.

2.6.2. Integrated Catchment Management

While environmental flows are a vital component of river health, many other catchment issues play a role. For example, cattle access to waterways significantly affects water quality and habitat integrity. When flow is not the only factor limiting river health, these other threats need to be managed in conjunction with the provision of flows. Many environmental assets are on the river floodplain, in wetlands removed from but linked to the main river stream. These wetlands often have management plans that range from required watering to weeding programs. A watering program alone will not necessarily result in the desired environmental outcome. The advantage of a government department or independent statutory authority is that they have influence over a range of environmental management programs, only one of which is managing environmental water. An example of this is the current Victorian arrangement where the Catchment Management Authorities are government funded and operated, and are responsible for all catchment issues. While a franchised manager could work with land managers and other environmental organizations, they are limited in their ability to plan an integrated approach. When the EWM is integrated into other forms of catchment management this better ensures the entity has access to all of the potential levers of change and is fully responsible for the outcomes achieved.
2.6.3. Partisanship in environmental water management

When the government acts as the EWM, the provision of environmental water can become a partisan issue and become subject to political gaming. Furthermore, the management of environmental water can be used to help achieve other unrelated public policy goals. Where the government interacts with a competitive market, and either does not attempt to achieve the indirectly related public policy goals through more direct means or with greater transparency, this can impact on the efficiency of water markets. For example, if the government, under political pressure from rural communities, chooses not to provide environmental flows in a drought year the market price for water will underestimate its true value. The true costs of water will then not feed through to other markets (such as fruit and vegetable markets) to ensure efficient allocations. Another example would be if the government responded to political pressure and allocated extra funding to improve flow conditions in rivers in marginal electorates.

CSIRO has suggested that an EWM may not be as able to attract donations from individuals and corporations that it might otherwise attract unless it is at arms length from the government (Connor & Young, 2003, p. 16). Furthermore, this report highlights that an EWM may suffer perceptions issues if they are not at arms length where “they may find it difficult to engage in counter-cyclic trading without being accused of having conflicts of interest and/or being accused of impropriety” (ACF, 2006, p. 5).

The recent Commonwealth Water Act (2007) considers this issue. It establishes the Commonwealth environmental water holder to manage the Commonwealth’s environmental water holdings (Part 1, Division 6). While the Minister may make rules (by legislative instrument) by which the holder must operate, the holder is not subject to direct input or instruction from the Minister (Section 107 and 109).

2.6.4. Competition

A potential attraction of the franchise manager is the opportunity it provides to introduce competition and new ideas into the EWM task, relative to the government department and statutory authority options. A suggested mode of operation is for government to tender
supply of the EWM for a period of, say ten (or more generally X) years as happens for the supply of other government funded goods and services, including, for example, defence equipment, infrastructure, computing and human resources services. Tenders could come from public corporations, NGOs, and also government departments or agencies. In addition to the tenders being evaluated and compared on the way in which they deliver the outputs or the inputs, the tender could seek proposals on how these inputs and outputs would be measured and monitored.

The tender approach places more weight on the integrity of the measurement and monitoring arrangements. However, it makes more explicit the necessity to focus on these issues, whereas the government option provides a cover to continue with a vague and uncertain set of objectives. Once established, the tender approach reduces the opportunities for political intervention.

2.6.5. Separating planning and management

A competitive market requires that participants behave non-strategically and have no relative bargaining power. Under current institutional arrangements the same government department may be responsible for determining the initial distribution of water to the environment and making environmental trading decisions. Therefore, with the introduction of environmental trade, governments may have an incentive to increase the initial environmental entitlement in order to subsequently sell the excess to make a profit. A clearer division of powers is required.

The Australian Conservation Foundation has noted that “environmental water should be managed to deliver environmental outcomes, not address the budget shortfalls of government agencies” (ACF, 2006, p. 5).

2.6.6. The role of Non Government Organisations (NGOs)

The question of who should be able to manage or purchase water on behalf of the environment links in to the question of who should pay to return environmental water. There are two common approaches to who should pay for natural resource management; user pays or polluter pays. In the case of environmental flows, the approach has been for
the Government, on behalf of the community, to fund the return of environmental flows as a public good. There will, however, be instances where segments of the community are willing to pay more than the value being contributed by government. In these instances, NGOs may wish to purchase additional environmental water funded through public donations. This is an approach that has been adopted widely in the western USA (Horne et al., 2008).

Conner and Young (2003, p. 21) note that the specific structure of an EWM will then determine whether donations can be made directly to the EWM.

To be eligible to receive donations that are tax deductible, the environment organization must take the form of (a) a body corporate, (b) a co-operative society (c) a trust or (d) an unincorporated body established for a public purpose by the Commonwealth, a State or Territory (Challen et al., 1996).

Thus the role of NGOs in receiving donations to source environmental water will vary depending on the structure of the EWM itself.

2.7. Summary

Instream flows for the provision of environmental benefits have historically been provided through regulatory policies such as license specifications and trading rules. Recent water reforms have established environmental water reserves with the same legal rights as other water users. This marks a shift in Australia’s approach to managing environmental water.

A formal environmental water reserve allows environmental managers to actively manage environmental water. When volumes of water are limited, adequate flow can not be provided for all environmental assets. Therefore, trade-off decisions between competing environmental demands are required. These competing demands may be different river reaches, or different environmental flow components (high winter flow rather than low summer flow) in the same reach. These decisions must be transparent and achieve the greatest overall benefit to the environment.
With the emergence of water markets, the least cost approach to providing fixed environmental flow targets has been extensively researched using optimisation and simulation modelling (as discussed in section 2.5.4). These projects usually assume that the required volume of environmental water is fixed (all be it at different times of year) and then investigate how various market mechanisms can be used to purchase water to meet this fixed volumetric target. While these existing studies contribute to optimizing the use of the market to achieve specific environmental water volumes, they are limited by assuming that environmental demand remains constant rather than constantly changing over time.

Currently in Australia, environmental flow releases are prioritized according to advice provided by expert panels regarding which flow components should be given priority. These usually compare environmental flow recommendations with the current availability of water and subsequently, decide which components should be provided. These approaches lend support to long term policy decisions between water allocations to the environment and consumptive users. However, active management of a parcel of environmental water requires operational decisions made on a shorter time frame. The decision must account for changes in environmental demands over time and readily adapt to new information.

Government policy to increase the Environmental Water Reserve uses a combination of purchasing water on the market and investment in water savings projects. While the environment entering the water market may cause the price of water to rise, this sends the correct signals and incentives about the true value of water. The positive element of on-farm assistance and education that comes through on-farm efficiency projects also has an important role in the transition process. Ideally, this transitional assistance should be provided separately to the purchase of water in the market, through schemes such as farm extension officers. There is still a role for large scale efficiency projects, however these need to be coordinated with the location of water purchases on the market to ensure that investments are not occurring in unsustainable irrigation areas.
3 Research Design

3.1. Introduction

Chapter 2 provided an overview of current environmental flow policies in Australia and options for better institutional arrangements for environmental water. Policy has moved from a predominantly regulatory approach to providing an environmental entitlement that can be actively managed. Policy makers are still grappling with unresolved questions:

- How should water be divided between consumptive users and the environment?
- How should the environmental allocation be increased if needed?
- Once an environmental allocation is set, how can this water best be used?
- How should environmental water be delivered (for example, to wetlands) to minimise wastage of water and impact on neighbouring landowners?

The first two questions have been extensively discussed in the literature. The structure of entitlements to suit environmental water requirements was discussed in Chapter 2 as were the options of increasing the existing environmental allocation (section 2.5). These two questions will not be further addressed in this research. It is the third question that is the focus of this thesis. Once the environment is allocated a parcel of water, decisions need to be made about how best to use this water to achieve environmental outcomes. While there may be discomfort in making “tradeoff” decisions between environmental outcomes, this is the reality of our current limited supply of water. Existing environmental flows studies in Australia tend to target regulatory approaches to providing environmental flows, and thus do not provide adequate information to allow decision making between flow elements and different time steps. Prioritisation of environmental water tends to be subjective without transparent and repeatable decision steps in place.

The aim of this study is to develop a conceptual framework for prioritising environmental water releases in a flexible and transparent manner. Existing approaches use expert scientific panels to assess scenarios and choose the best environmental outcome (for example, Brown & King, 2000). While these methods may work for long term water resource planning decisions, they can not be easily adapted to assess new situations as
they arise. They are also subjective and dependent on the expert panel involved (Burgman, 2005).

In other aspects of water allocation, Australia has opted for a market approach to move water to the highest value use. If a market can be used to allocate water between different irrigators (each with very different timing and volume requirements, or different demand curves for water), is it possible that the same concepts can be used to allocate water between different rivers or components of the ecosystem (each with different demand curves for water)? While it is not being suggested that individual fish and birds place bids with an auction house, the concept of using marginal value and “trading” water to optimise overall benefits can be applied. If an environmental manager has a fixed volume of water (i.e. the resource is capped), and the demand curve for each of the different elements of the environment is defined, an optimisation approach can be used to allocate water to achieve the optimal environmental outcome. As environmental demands change over time, the optimal allocation between elements of the environment will change over time (the same way that water moves between different irrigators at different times).

To use this approach, the shape of the demand curve for water for each environmental component is required. An optimisation model is then constructed to allocate water between the various environmental components at different time steps. This is a significant change in the approach to prioritising or allocating environmental water. As such, rather than jump into a complicated model, the process begins with a simplified representation of a case study basin. This allows assessment of which model inputs are playing a significant role in the decision making process. Layers of complexity are added as needed.

This chapter begins with detailed research questions to assist in meeting the research aim. An outline of the overall method is then provided, followed by a brief introduction to the case study basin (the Goulburn River Basin).
3.2. Research questions

The aim of this study is to develop a conceptual framework for prioritising environmental water releases in a flexible and transparent manner.

The following research questions have been identified.

1. How can environmental water requirements be quantified to indicate the marginal value of water to the environment?
2. What factors influence the decision to release environmental water in a given month?
3. Does an optimisation approach provide the flexibility and transparency to make release decisions?

3.3. Method

The method to prioritise environmental flow components and make release decisions from storage is demonstrated using the Goulburn River Basin as a case study (described in detail in section 3.4).

Prioritization of environmental water use, and indeed determining the volume of environmental water allocation, is impeded by our limited understanding of ecosystem responses to flow. While quantification of environmental flow requirements has progressed, it is still a new practice, drawing on incomplete scientific knowledge. The timing and magnitude of required flows, the ecosystem response to flows, and how the ecosystem response should be “valued” or compared to the value provided to society from other water uses are all uncertain. Before environmental release decisions can be optimised, an understanding of the marginal value of water to the ecosystem is required. Chapter 4 explains the importance of marginal value to the decision making process and suggests that an environmental response curve can be developed for each ecologically significant flow element in a given reach by combining the shape of the habitat rating curve (the relationship between habitat area and flow) and the natural frequency of habitat provision.
It should be noted that environmental response curves rather than environmental demand curves are developed for each flow element. The distinction is that an environmental response curve relates flow to ecological outcome, while a demand curve relates flow to a dollar value. No contingent valuation has been undertaken as part of this research. The marginal value is assessed in terms of ecological outcome rather than dollar value.

Once environmental response curves (relating flow to ecological outcomes) for each component is established, these curves can be used in an optimisation model to determine the release pattern that will achieve the best overall ecological outcome.

An important point of clarification is required here. The optimisation model aims to determine the optimal release pattern for \textit{environmental allocations}. Under current arrangements in Australia, a storage operator releases water from storage to provide downstream needs. These releases include legislated flow releases, such as minimum flow requirements, and releases to meet the needs of irrigators downstream. If an environmental manager now holds a parcel of water in storage, as an environmental allocation, the environmental manager must indicate to the storage operator the timing of when they want this water released from storage. Irrigators make their decision based on their crop requirements and order water from storage to meet their demands. An environmental manager must order water from storage to meet ecological requirements downstream and achieve the best ecological outcome. Other releases, such as irrigation releases, spills or legislated flows, are not within the environmental manager’s control. This is the same as decisions for an irrigator, where the release decisions of an environmental manager are outside the irrigator’s control. The optimisation model is aimed at helping the environmental manager make release decisions. The model makes the decision on when to release environmental allocation water, given that all other releases are not in the environmental manager’s control. When should an environmental manager make a decision to release as apposed to store water for future years? When would they release a high flow compared to a low flow? There are currently no tools available to make these operational level decisions. This optimisation model is a first step in demonstrating how a modelling approach could improve the decisions.
The aim of the optimisation model is to determine the monthly release pattern for environmental water. Initially a single year model is constructed, allowing monthly release decisions to be made over a twelve month period. Subsequently, this is extended to a multi-year model, allowing decisions made in one year to influence environmental demands in later years. The single year model is used initially because it simplifies the model inputs, making it easier to isolate the elements that drive the decision making process. Further layers of complexity can then be gradually added.

The single year optimisation model is described in Chapter 5. The model determines the optimal monthly release pattern by calculating the risk to each environmental flow component, at each relevant timestep, given the passing flow. Figure 3-1 shows the model structure. Passing flows from irrigation and storage spills are considered exogenous (outside the control of the EWM). Note there is also an environmental flood release and compensation flow that are legislated releases outside the control of an environmental manager. These releases are also considered exogenous. These exogenous flows will vary depending on storage levels and climate conditions. The environmental release decision is only for the volume of environmental allocation available to the environmental manager that can be actively managed. The annual volume of environmental allocation is predetermined in the model. The decision is then how to release this water on a monthly timestep to achieve the best environmental outcome. In the optimisation model, release of this environmental entitlement water is made based on the relationship between environmental risk and flow for each of the flow components.

![Figure 3-1: Schematic of model inputs, constraints, decisions and outputs](image-url)
The model is used to assess the sensitive trigger points in the decision making process. What variables influence the release decision? The model is a simplified representation of the catchment and many assumptions have been made in constructing model inputs. The environmental release decision will be highly sensitive to some of these model inputs, and not to others. This helps inform where knowledge gaps are and where research energy should be focussed.

Environmental water requirements will change with time. The environmental response curves developed in Chapter 4 incorporate this. Release decisions in one year will influence the demand for water in future years. If the system is already stressed, it will be increasingly important to provide water in the next time step. The multi-year model (presented in Chapter 6) is constructed to look at how different climate sequences influence the release decision over time. The multi-year model also introduces an additional decision variable with the option to hold environmental water in storage from one year to the next (carry over water).

Both the models are deterministic; that is, the model knows the inflows for all time steps at the start of the modelling period. In reality, an environmental manager would not have knowledge of the climate and inflow conditions in future years. A deterministic model has been chosen as it makes the optimisation process easier to assess and, is therefore, an important first step in testing this approach to allocating environmental water. However, a stochastic model would form an important next step.

The Goulburn Basin in North Victoria will be used as a case study to illustrate the approach. The Goulburn Basin is introduced in more detail in the following section. The Goulburn River is simplified in the model and represented by only two reaches, with generalised streamflow and storage inputs. It should be noted that the case study is purely to demonstrate that the method developed is a worthwhile and possible process, rather than to achieve exact figures for the Goulburn Basin itself. That said, the study basin is represented accurately enough to adequately apply the methodology developed and analyse the outcomes.
Section 2.5.6 highlighted that the environmental water requirements and options to manage environmental water are significantly different in regulated and unregulated river systems. For this reason, only the regulated portion of the Goulburn Basin is represented, and not the environmental water requirements in unregulated tributaries.

While environmental flow recommendations are usually made on a daily timestep, a monthly timestep is used in the optimisation model as regulated flows are fairly consistent during each month and can be adjusted to meet daily requirements.

### 3.4. Simulation or Optimisation?

There has been a vast array of simulation and optimisation models developed to assist in water resource planning. These are described in detail in the literature (Wurbs (1993) provides a good overview and reference list for other modelling papers). A simulation model represents a system to make predictions on how it would behave with a given set of inputs. Alternative input scenarios are run through the simulation model to predict outcomes over a range of conditions. For example, prediction about reliability of water supply could be made by running a number of different climate and streamflow inputs through a simulation model of a given river and supply system. Similarly, different operating rules for the system could be modelled. On the other hand, optimisation models determine a set of values for decision variables that lead to the optimal outcome (maximum or minimum) for an objective function, subject to a series of constraints. The constraints and objective function are linked to the decision variables through a series of mathematical expressions. In the context of water supply and operations, the objective function usually relates to economic benefits and costs or reliability of water supply. The decision variable is often to do with release decisions from storage and end of period storage volumes. Constraints refer to storage capacity, channel capacities and streamflow requirements (Wurbs, 1993).

“There is no simple answer to the question of which models and analyses techniques should be used for a particular modelling application” (Wurbs, 1993, p. 465). While models are often described or categorised as simulation or optimisation models, the reality is that many models include a combination of both simulation and optimisation
modelling techniques. Some balance is preferable, allowing the model to be as flexible as possible, while still describing the system in enough detail.

There are limited examples of environmental water requirements included in water resource models (Harman & Stewardson, 2005). Modelling of environmental water requirements has tended to be done through existing water allocation models (such as REALM in Victoria). Environmental water requirements are currently specified as fixed targets (refer to section 2.4.2). In response to this, a number of studies have looked at release decisions to conform to environmental flow targets. This modelling often uses a combination of optimisation and simulation, optimising the operational rules for the system and simulating the response to a number of different environmental flow targets (for example, Hughes et al., 1997; Hughes & Ziervogel, 1998). These models make releases to “piggy back” natural high flow events, thus minimising additional releases required from storage. Harman and Stewardson (2005) go on to look at real-time tributary flow data as a trigger for environmental flow releases from dams. These studies have a number of things in common (1) they all use a combination of optimisation and simulation techniques and (2) they are all based on fixed environmental targets or scenarios of fixed environmental targets.

The approach adopted in this research uses a combination of simulation and optimisation modelling. An optimisation model has been constructed to determine the optimal environmental release given a set of constraints. One of the key innovations of this research is to treat the environmental requirements as having a demand curve for water rather than fixed targets. To benefit from this approach an optimisation technique provides the most flexible means of isolating the release pattern that achieves the optimal environmental outcome. The system operation is represented using a series of modelling constraints. However, in combination with the optimisation model, a series of different scenarios are run through the model to represent different climate and storage scenarios. Similarly, when capacity constraints are added to the optimisation model, a simulation approach is used to look at different capacities. This limits the complexity of the model structure and speed of solving the optimisation problem.
3.5. The Goulburn River Catchment: Study Basin

This section describes the Goulburn River catchment, outlining the operation of the system, the key water users, environmental flows studies and available data and modelling for the study basin.

3.5.1. Catchment Description

The Goulburn River is located in Northern Victoria, Australia. It is Victoria’s largest river and one of the major tributaries of the River Murray, discharging to the Murray River east of Echuca. A map of the catchment is shown in Figure 3-2.

The Goulburn River basin supports a large agricultural industry, estimated to be worth A$1.3 billion per year (Goulburn Broken CMA, 2003). The agriculture in the catchment is diverse. The southern catchment is used for hard wood timber productions, with some areas reserved as national parks. Moving north through the catchment, land is used to produce sheep, and beef and dairy cattle. Areas in the northern catchment also produce huge quantities of fruit and vegetables. The Goulburn valley also has large areas of dryland cropping.

Climate also varies across the catchment. The high country of the south east endures cool winters with persistent snow and an average annual rainfall greater than 1600 mm. Moving north into the valley and floodplains, rainfall decreases and can be less than 450 mm per year, with potential evaporation up to three times that amount (G-MW, 2008).

Goulburn Murray Water manages the water supply system and supplies Shepparton and Central Goulburn Irrigation Areas, along with Rochester Irrigation Area in the neighbouring Campaspe Basin and the Pyramid / Boort Irrigation Area in the west of the neighbouring Loddon Basin.

Lake Eildon, the major storage on the Goulburn River, has a catchment area of 3,900 km² with major tributaries of the Delatite, Howqua, Jamieson and Big River flowing into the storage. At a full supply level Lake Eildon holds 3,390,000 ML and has a surface area of 13,840 ha (SKM, 2006b, p. 3). The Goulburn River downstream of Lake Eildon is highly
regulated. While the average annual inflow to Lake Eildon is similar to the average annual outflow (approximately 1900 GL), the seasonality is completely reversed. Water released from Lake Eildon flows down to Goulburn Weir (downstream of Nagambie) where large volumes of water are diverted to the Waranga West and Goulburn East Channels and into the neighbouring catchments to meet irrigation demands. At full supply level there is a pool of 25,000ML and a surface area of 1,120 ha at Goulburn weir (SKM, 2006b, p. 7). The mean annual flow downstream of Goulburn Weir has more than halved from 2,631 GL under natural conditions to 1,034 GL under current conditions.

Streamflow along the Goulburn River has been significantly modified by Lake Eildon and Goulburn Weir. The effect of regulation on streamflow has been an extreme seasonal inversion between Lake Eildon and Goulburn Weir, with high summer flows and low winter flows. Diversions at Goulburn Weir have significantly reduced the total volume of water flowing downstream of the weir. Gauged stream flow data is available both downstream of the two major structures\textsuperscript{14}. This data was used to derive current and natural flow series over the period August 1975 to July 2000 (Cottingham \textit{et al.}, 2003). A comparison of current and natural flow regimes downstream of the two major structures in the Goulburn River is shown in Figure 3-3 and Figure 3-4. Figure 3-3 shows flow duration curves which show the probability that a certain flow (y-axis) will be exceeded (with probability of exceedance on the x-axis). In keeping with the description above, Figure 3-3a shows that flows between 1000 and 10,000ML are more likely to be exceeded with releases from storage in summer month than they would have been under natural conditions. Figure 3-3b demonstrates that diversions at Goulburn Weir have decreased the probability of a given flow being exceeded (for example, a flow of 1000 ML is exceeded 82\% of the time under natural conditions and only 28\% of the time based on historical flows). Figure 3-4 shows the average monthly flow downstream of both structures, demonstrating the seasonal pattern and how it has altered from the rivers natural state. The expansion of the water market has led to increasing inter-valley transfers (IVTs). The Goulburn will play an important role in meeting inter-valley

\textsuperscript{14} A full list of data available in the Goulburn Catchment is provided in Appendix A.
demands. IVTs are likely to increase flows leaving the Goulburn system between January and March, significantly increasing flows downstream of Goulburn Weir.

Apart from its agricultural significance, the Goulburn River is listed as a Victorian Heritage River and a high priority for environmental management (Department of Natural Resources and Environment, 2002a).

Figure 3-2: The Goulburn River Catchment. Adapted from Victorian Data Warehouse Map Data\(^{15}\) (Department of Sustainability and Environment, 2008a)

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\(^{15}\) Note that environmental flow reaches refer to the reach numbering defined as part of the original environmental flows study. The study reaches for this research are different as only two of the five environmental flow reaches are incorporated in the modelling.
Figure 3-3: Flow Duration Curve (a) Downstream of Lake Eildon and (b) Downstream of Goulburn Weir
Figure 3-4: Average monthly flow (a) Downstream of Lake Eildon and (b) Downstream of Goulburn Weir
3.5.2. Current Water Entitlements

Consumptive entitlements

Current irrigation entitlements in the Goulburn System total around 985 GL (refer to Table 3-1), with an irrigated area of 360,000 ha (Goulburn Murray Water, 2006/07).

Table 3-1: Goulburn System Irrigation Areas and Entitlements (Goulburn Murray Water, 2006/07)

<table>
<thead>
<tr>
<th>Irrigation Area</th>
<th>Entitlement (GL)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Shepparton</td>
<td>165</td>
</tr>
<tr>
<td>Central Goulburn</td>
<td>351</td>
</tr>
<tr>
<td>Rochester</td>
<td>172</td>
</tr>
<tr>
<td>Pyramid-Hill Boort</td>
<td>203</td>
</tr>
<tr>
<td>Goulburn Diverters</td>
<td>94</td>
</tr>
<tr>
<td><strong>Total Goulburn</strong></td>
<td><strong>985</strong></td>
</tr>
</tbody>
</table>

The Bulk Entitlement for the Goulburn system (Victorian Government, 1995) specifies that Goulburn Murray Water may take up to an annual average total of 1,919 GL over any ten year consecutive period. The releases from Lake Eildon must not exceed 1,410 GL over the same period (Clauses 6.1 and 6.2, Bulk Entitlement, Victorian Government 1995)\(^{16}\).

Each season, annual allocations are calculated based on the volume actually held in storage minus losses (evaporation and seepage incurred through storage and delivery) minus existing commitments (including urban water supply and legislated minimum environmental flows). Annual allocations are presented as a percentage of entitlement available. For example, if allocations are 43%, irrigators can expect to receive 43% of their entitlement that year. Allocations are updated throughout the irrigation season. In recent years, low inflows in the Goulburn system have meant a series of low allocation years for irrigators.

\(^{16}\) Note that this volume is to ensure that irrigation entitlements can be met, along with supplies to other water providers and system losses.
Goulburn Valley Water and Colliban Water both hold Bulk Entitlements to meet urban demand. The total urban entitlement is 44 GL/yr (Department of Sustainability and Environment, 2007).

The State Electricity Commission of Victoria (SECV) is entitled to an annual volume of 52 GL/yr which accrues at a rate of 13 GL/month from May to August. If Lake Eildon spills, any unused allocation accumulated in storage from previous years is cancelled (DSE, 2003, section 4.2.8).

**Environmental Water Reserve**

Recall from chapter 2, the environmental water reserve comprises all water available to the environment. The Goulburn Basin Environmental Water Reserve comprises the following components:

- passing flows released as a condition of consumptive bulk entitlements;
- all other water in basin not allocated for consumptive use (in other words, unregulated flows); and
- specific environmental entitlement.

Goulburn Murray Water holds the main Bulk Entitlement (Victorian Government, 1995) for the Goulburn system. This allows Goulburn Murray Water access to an annual volume of water to supply primary entitlements (domestic and stock, irrigation licences and other suppliers, for example, Goulburn Valley Water), subject to a range of conditions. It is these conditions specified in the Bulk Entitlement that define the majority of the Environmental Water Reserve.

The conditions specified in the Bulk Entitlement for storage volume and delivery infrastructure, along with the annual diversion volume, ensure that the unregulated flow portion of the environmental water reserve is protected.

The Bulk Entitlement specifies that Goulburn Murray Water must ensure the following passing flows:

- A minimum flow immediately downstream of Lake Eildon pondage of 120 ML/d (as specified in Clause 11). This passing flow increases to 250 ML/d in any
month where the volume of inflow to Lake Eildon during the previous 24 months exceeds the trigger flow specified in Table 3-2

Table 3-2: 24 Month trigger inflows to Lake Eildon (Schedule 6)

<table>
<thead>
<tr>
<th>Month</th>
<th>Jan</th>
<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
<th>May</th>
<th>Jun</th>
<th>Jul</th>
<th>Aug</th>
<th>Sep</th>
<th>Oct</th>
<th>Nov</th>
<th>Dec</th>
</tr>
</thead>
<tbody>
<tr>
<td>Trigger (GL)</td>
<td>2785</td>
<td>2786</td>
<td>2782</td>
<td>2785</td>
<td>2782</td>
<td>2796</td>
<td>2802</td>
<td>2801</td>
<td>2779</td>
<td>2780</td>
<td>2776</td>
<td>2788</td>
</tr>
</tbody>
</table>

- If, in a given year, the minimum flow downstream of Eildon has increased and the 12 month exceedance flow up to October is in excess of 800,000 ML, a volume of up to 80,000 ML may be required for release during November. This release must not exceed a rate of 16,000 ML/d (Schedule 6). This parcel of water is designed to inundate wetlands and has historically been available on average every 10 years.
- A minimum flow downstream of Goulburn Weir of a weekly average of 250ML/d and a minimum on any one day of 200 ML/d (Clause 11).
- Any additional flow required to maintain a minimum average monthly passing flow at McCoys Bridge (downstream of Goulburn Weir) of 350 ML/d and a daily requirement of no less than 300 ML/d between November and June. The required monthly average is 400 ML/d from July to October, with a daily requirement of no less than 350 ML/d (Clause 11).

The Bulk Entitlement also allocates a volume of 30,000 ML per year available to maintain water quality in the waterway (Clause 12.3d).

There is currently no Environmental Bulk Entitlement available for the Goulburn River. There is a Goulburn Murray Bulk Entitlement of 141 ML (Victorian Government, 2007), a low reliability entitlement aimed at meeting Living Murray objectives. It is anticipated that the Food Bowl Modernisation project will lead to water savings of an estimated 225 GL/yr to be shared between irrigators, Melbourne and the environment. The environment will obtain an entitlement of 75 GL, and as opposed to the existing entitlements, this entitlement is specifically to be used in Victorian tributaries (particularly the Goulburn, Campaspe and Loddon Rivers) rather than to meet Murray river objectives (Food Bowl Modernisation Project Steering Committee, 2007).
3.5.3. **Existing Water Allocation Model**

The Goulburn Simulation Model (GSM) represents the water supply system for the Goulburn, Broken, Loddon, and Campaspe Basins\(^{17}\). This is an existing model used as a water resource planning tool for the Goulburn River storages. It should not be confused with the optimisation model constructed as part of this research. It is provided here as background to the Goulburn catchment as it is the major water resource planning tool and provides data that can be used in the optimisation model\(^{18}\). Note the GSM water allocation model encompasses not only the Goulburn basin, but also the neighbouring basins to allow modelling of inter valley transfers.

The primary objective of the GSM is to define security of supply for major user groups of the Goulburn system (DSE, 2003). The model is thus run over a long period (approximately 100 years of record) at a monthly timestep. The operation of the system (storage targets, irrigation demands and allocations) are modelled and calibrated to historical data to ensure that the model indeed reflects actual system reliability. Detailed documentation of the GSM can be found in DSE (2003). Aspects of the GSM relevant to this research are discussed in more detail in Chapter 5, with details listed in Appendix A.

**Calculating Allocations**

As described in Chapter 2, water users hold an entitlement for water. Each year, an allocation is calculated as the percentage of the entitlement available for use in the given season.

The Goulburn basin restriction policy determines the allocation of water resources available for use within a water year, and between water years (DSE, 2003). In the

\(^{17}\) The GSM is constructed using the **REsource ALlocation Model** (REALM). REALM is a simulation modelling package which is used to represent any water supply system using a series of arcs, nodes and storages with rules and constraints on their capacity and operation. “It uses a linear programming routine within each time step to perform a water balance and optimises the assignment of water within the network in accordance with user defined operating rules.” (DSE 2003, pg 2). More details on REALM can be found in the Department of Sustainability REALM Manual (Department of Sustainability and Environment, 2008b).

\(^{18}\) Refer to section 5.3 for further details on how the GSM is used to generate inputs for the optimisation model.
Goulburn system, the current restriction policy is to supply the maximum seasonal allocation in the current irrigation season subject to:

- the seasonal allocation in the current season is not less than 100% of water right;
- and
- seasonal allocation in the current season only exceeds 100% once it is ensured there is enough water remaining in storage to meet the following year’s commitments (assuming that natural inflows to storages in the intervening period will be equal to the 99% exceedance probability flow).

An initial allocation is announced at the start of the season to allow irrigators to make planning decisions. This allocation is calculated assuming only the 99% exceedance probability inflow to storages. The allocation can then be increased during the season as more information about the real inflows becomes available.

The allocation is calculated based on the available resource in storage and the forecasted inflows to storage. The water available is calculated as the sum of water currently in storage, plus predicted inflows, minus passing flow requirements. The storage volume is calculated as the volume in storage at the start of the month in Lake Eildon, Waranga Basin and Greens Lake. Goulburn Murray Water has calculated marginal monthly inflows based on 99% exceedance flows.

*Using a one year planning period and Lake Eildon as an example, the May flow (31GL) is calculated from the 12 month flow estimate ending in April (318 GL) minus the 11 month flow estimate ending in April (287 GL). This method is continues with reducing durations down to the one month, 99% exceedance April inflow” (DSE, 2003, p. 27).

A similar process is used for the two year planning period, however the marginal inflows are calculated using durations from 24 months down to 13 months.

As the irrigation season progresses, the inflows will usually be greater than the 99% exceedance estimates used in the allocation procedure. Each month the allocation is reviewed to reflect the actual available water at that time step. If necessary the allocation is increased to reflect the greater resource. On the infrequent occasions when inflows are
less than estimates, the seasonal allocation is not reduced, but the previous month’s allocation is maintained.

When making an allocation part way through the season, the volume of demand supplied to date needs to be taken into account. The delivery of irrigation water within a season is also based on historical use patterns. In each month, after estimating the water use limit (the available resource and allocation relationship), the cumulative restricted crop requirement can be determined. The unrestricted crop demand is calculated using PRIDE modelling and based on the soil moisture and crop type (Erlanger et al., 1992). The restricted demand curve (typically an s-shape) is determined based on historical usage patterns through the season. More detail can be found in Zaman (2005).

3.5.4. Environmental Flow Studies

An environmental flow study for the Goulburn River was completed in 2003 (Cottingham et al., 2003). The project study area included the Goulburn River and its associated floodplain, downstream from Lake Eildon to the confluence of the River Murray (Figure 3-2). The River was divided into five reaches to represent different conditions along the river:

- Reach 1: Lake Eildon to Molesworth
- Reach 2: Molesworth to Seymore
- Reach 3: Seymore to Nagambie
- Reach 4: Nagambie to Loch Garry
- Reach 5: Loch Gary to the River Murray

Reaches 1, 2 and 3 are downstream of Lake Eildon and thus highly regulated. Reaches 4 and 5 are downstream of Goulburn Weir, where the majority of irrigation releases are diverted from the River. Thus the current hydrology of reaches 4 and 5 is significantly different to that of reaches 1, 2 and 3.

The environmental flows study used the FLOWS method (Department of Natural Resources and Environment, 2002b), which is discussed in more detail in Chapter 4. An expert panel identified key environmental objectives and values and flow stressors that
may impact on environmental values. The Flow Events Method (FEM) (Stewardson & Gippel, 2003) was used to assess each of the flow stressors.

Table 3-3 summaries the identified issues and recommendations made as part of the environmental flows study.

**Table 3-3: Summary of issues requiring environmental flow recommendations (Cottingham et al., 2003)**

<table>
<thead>
<tr>
<th>Issue</th>
<th>River Attribute</th>
<th>Reach</th>
<th>Flow Component</th>
<th>Flow Recommendation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inversion of the flow regime in Reaches 1 – 3:</td>
<td>In-channel macrophytes</td>
<td>1,2,3</td>
<td>Summer low flows</td>
<td>Adoption of a precautionary approach suggests indicative summer-autumn base flows below 1,000 – 3,000 ML/d in Reach 1. Further investigations are required to better quantify environmental flow recommendations</td>
</tr>
<tr>
<td>High water velocity</td>
<td></td>
<td></td>
<td>Summer low flows</td>
<td></td>
</tr>
<tr>
<td>Duration of bench inundation</td>
<td>Aquatic macrophytes</td>
<td>1,2,3,4</td>
<td>Spring low flow</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Macroinvertebrates</td>
<td></td>
<td>Summer low flow</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Biochemical processes (e.g. cycling of carbon and nutrients)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Availability of riffle habitat</td>
<td>Macroinvertebrates</td>
<td>1,2,3</td>
<td>Summer low flow</td>
<td></td>
</tr>
<tr>
<td>Availability of shallow water habitat</td>
<td>In-channel macrophytes</td>
<td>1,2,3</td>
<td>Summer low flow</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Small fish</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Frequency of freshes</td>
<td>Geomorphology</td>
<td>4,5</td>
<td>Summer freshes</td>
<td>Current frequency of freshes maintained, with natural magnitude and duration</td>
</tr>
<tr>
<td></td>
<td>Aquatic macrophytes</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Macroinvertebrates</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Fish</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Frequency of wetland inundation</td>
<td>Geomorphology</td>
<td>1,2,3,4,5</td>
<td>Spring Flood</td>
<td>Annual flood of varying magnitude (15,000 – 60,000 ML/d peak magnitude). No action required if floods occur naturally.</td>
</tr>
<tr>
<td></td>
<td>Wetland vegetation</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Macroinvertebrates</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Fish</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Duration of bench inundation</td>
<td>In-channel macrophytes</td>
<td></td>
<td>Spring and summer low flow/freshes</td>
<td>Experiment to evaluate extended duration of bench inundation</td>
</tr>
<tr>
<td></td>
<td>Macroinvertebrates</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Availability of deep water habitat</td>
<td>Fish</td>
<td>4,5</td>
<td>Summer low flow</td>
<td>Minimum flow of 610 ML/d measured at Murchison</td>
</tr>
<tr>
<td>Rate of rise and fall in river levels</td>
<td>In-channel macrophytes</td>
<td>1,2,3,4,5</td>
<td>Rate of rise and fall</td>
<td>No specific flow volume required. Care is required to avoid rates of rise and fall exceeding 95th percentile values of the natural flow regime.</td>
</tr>
<tr>
<td></td>
<td>Macroinvertebrates</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Fish</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
After the initial environmental flows study, the environmental flows panel was reconvened specifically to review environmental flow recommendations addressing potential flow increases in reaches 4 and 5 due to irrigation transfers to downstream irrigation districts (Cottingham et al., 2007). The study provided a more detailed list of flow stressors and corresponding recommendations for reaches 4 and 5. This environmental flow study moved beyond a single threshold recommendation for each flow component and made recommendations expressed as the frequency distribution of different events. As an example, Figure 3-5 shows the final recommendations for summer months in reach 4. They are complicated to interpret. Solid shapes and sold lines represent upper bounds for each percentile year, while open shapes and broken lines represent lower bounds. To minimise environmental risk, flows should occur between the upper and lower bounds. If we take for example a median year, summer discharges pose little risk if they fall within the shaded area. Based on the median year, flows of 500 ML/d should be exceeded at all times. There is little risk if flows fall between the range of approximately 500 ML/d and 2,000 ML/d for 70% of the time (based on the lower bound). Flows above 2,000 ML/d can occur between 30% and up to 70% of time (based on the upper limit) (Cottingham et al., 2007). While this approach provides a more detailed flow regime and incorporates concepts of risk, it is still aimed at assessing long term flow regimes rather than providing input for operational (short term) decisions.

![Figure 3-5: Upper and lower bounds and exceedance levels for flow duration for 10th, 30th, median, 70th, 90th percentile years and all years (1975 – 2000) for the pre-regulated flow regime in reach 4 over summer months (adapted from Cottingham et al (2007)).](image)
4 Environmental Response Curves

The development of procedures to simultaneously analyse and optimize outcomes of water management decisions for the environment and consumptive water users is complicated by the stark differences in conceptual frameworks used to understand the effect of flow volumes on natural ecosystems and water-dependent businesses. Reconciling these differences is not trivial. This chapter addresses these differences to provide a sound conceptual basis for integrating ecological and economic assessments of water resources planning into an optimisation model. The basic premise of this chapter is that current methods for representing environmental flow requirements do not permit adequately transparent, optimal decisions for short term planning (for example, operational decisions). As argued in the previous chapter, better tradeoffs and clearer decisions about environmental water would be made if its marginal benefit function was known. This is not an attempt to replace existing environmental flow methods used for water resource planning, but rather to present existing data to allow operational decisions for the environmental entitlement.

This chapter begins by introducing the concept of marginal value and demonstrates its application to environmental water requirements (section 4.1). A distinction is drawn between environmental response curves (relating environmental outcome to flow) and environmental demand curves (relating dollar value to flow). Current methods for determining environmental flows were briefly discussed in section 2.4.2. The literature on environmental response curves is reviewed in section 4.2, followed by a description of the development of environmental response curves, highlighting the difficulties (section 4.3). This approach is then applied to the study basin, the Goulburn River, in section 4.4. The chapter ends with a brief discussion of how to convert and environmental response curve into a demand curve (section 4.5).

4.1. The importance of understanding marginal value

The basic concept of markets was discussed in section 2.3. Recall that water markets function because economic gains can be captured by transferring water to a different
“location, season or purpose of use” resulting in higher net returns than the current pattern of use (Saliba & Bush, 1987). Applying this same concept to make “trades” between different environmental flow components, as a means of allocating environmental water, results in maximum social gain. When environmental water allocations are inadequate for all environmental requirements, a tradeoff between flow components or river reaches is required. Rather than Irrigator A and Irrigator B, Figure 2-1 could show winter high flow (flow component A) and summer low flow (flow component B). Comparing the marginal benefit from different flow components (or, flow on different river reaches) allows for allocation between the two components to achieve the greatest benefit.

The analogy of trade between irrigators can similarly be used to describe trade between the environment and irrigators. To achieve maximum social gain, water should be distributed between the environment and irrigation so that the marginal value of water to the environment and the marginal value of water to irrigation are equal. This requires a common currency to compare environmental outcomes and consumptive water users, which inevitably means valuing environmental outcomes in monetary terms. This complexity is not addressed in detail in this thesis, however is briefly discussed a later sections of this chapter.

In order to participate in a market, and maximize gains, a trader must know their own demand for the goods being traded. In the majority of markets (including the water market between irrigators) the goods traded are described as rival, excludable and private. A good is “rival” when one person’s enjoyment prevents the enjoyment of another person. A good is “excludable” when it is easy to exclude non-payers from enjoying the good. One individual’s willingness to participate is independent of another participant’s willingness to trade. Willingness to participate is based on the perceived individual benefit reaped by those who trade. The sum of all the individual private benefits is the benefit reaped by all participants and is referred to as the social benefit. Individual participants in a market (for example, irrigators in a water market or a more general example, an individual purchasing apples) know what they are willing to pay for a good. Each participant also requires an understanding of which factors will determine their
willingness to pay. The willingness of an irrigator to purchase water will be determined by, among other things, its price, what crop is grown, how much water the crop needs, what alternatives to water are available, and the sale price of the crop. Similarly, the number of apples a person purchases is based on (for example) their income, the price of the apples, the enjoyment they will gain from additional apples and the alternative foods available. Individuals can determine their own willingness to pay for particular goods because they hold all the information.

In contrast, most of the benefits of environmental flows have public good characteristics: they are non-rival and non-excludable, which complicates the process of determining a demand curve for environmental water. The steps required to determine a demand curve for the environment are outlined in Figure 4-1. The steps are to quantify the impact of change in flow on ecological condition, its effects on environmental service provision, and the value of environmental service provision to society. In reality, each stage is encumbered by unknowns and uncertainties; the relationships between flow and ecology are scientifically uncertain and complex, and environmental amenity is a public good with no market value (as compared to the case with private goods such as irrigation).

![Figure 4-1: Conceptual steps in deriving environmental demand curves](image)

If an environmental demand curve is so difficult to determine, why persist with this approach? In essence, demand for water has outstripped supply and allocation choices must be made and these will not be economically efficient (result in the maximum possible social gain) without knowing the marginal benefit of water for the environment.
Discussions about environmental water requirements usually focus on allocating water between consumptive users and the environment. If these discussions are to include the efficiency and transparency of trading between competing uses, then the value of providing additional water must be understood in comparative value units. For the environment to be treated on equal terms, its demand must be represented in a manner that allows its value to be compared with consumptive use. Because irrigators use the water market to allocate water, consumptive users know its marginal value in dollar terms. Thus the value of additional environmental water must be expressed in dollar terms. This requires an environmental demand curve in its full form (developed from an environmental response curve and contingent valuation). However, if the focus is on how to best use a parcel of environmental water to achieve environmental outcomes, the environmental response curve alone can provide substantial information to support these decisions. Understanding the marginal value of water to different environmental attributes in terms of environmental outcome (rather than dollar value) allows management of the environmental water allocation to achieve maximum outcomes.

This might be best explained using a simple example where we assume that willingness to pay per “unit of environmental outcome” is fixed, and thus the demand curve will follow the shape of the environmental response to flow. Consider first the consequences of using current environmental flow recommendations, which provide single values or thresholds for each flow component. While these values are based on the best available scientific information, they constitute a threshold or single minimum flow, which implies no benefit in providing flow below this threshold (Figure 4-2a). Recall that the benefit curve represents the total value compared to the quantity of water provided (as in Figure 2-1a) and the demand curve is the marginal value relative to the quantity of water (the derivative of the benefit curve). If a threshold flow is set, the shape of the environmental total benefit curve is implied, but there are few environmental flow components where this assumption would hold true. Wetland inundation might be an exception, where a trigger flow must occur before the wetland commences to fill. However even in this case, it only holds for a single wetland and where the duration of hydrological connection between the wetland is not important. If multiple wetlands are considered, the response curve becomes more continuous.
The shape of the environmental total benefit curve will significantly affect the “trade” decision between environmental flow components and decisions between the environment and consumptive use. Figure 4-2 provides examples of environmental total benefit curves for a minimum flow recommendation. The y-axis shows the total social benefit from providing the environmental flow and the x-axis shows the daily flow (ML/day). Figure 4-2a represents the current environmental flow recommendations, where a minimum threshold of 20 ML is set. When the benefit curve is linear, halving the flow exactly halves environmental and social benefit (Figure 4-2b). In Figure 4-2c, providing a small flow of 5ML/day will provide half the benefit of a 20 ML flow, but a large increase in flow (above 15ML/day) is required for any additional gains. This may be the case in a river where benches or additional channels must first receive water. In Figure 4-2d, there is significant benefit in having a small amount of water in the river, but the curve flattens as flow increases. Each of these possible total benefit curves has been translated into a marginal benefit curve (or demand curve) in Figure 4-3.

Figure 4-2: Possible total benefit curves for an environmental flow

Figure 4-3: Associated marginal benefit curves for an environmental flow
We can now consider how using these marginal benefit curves could inform allocation decisions. Consider an environmental manager who has 30 ML of water to release on a given day. The manager’s decisions are between allocations to a flow component with the total benefit curve shown in Figure 4-2d (with diminishing marginal returns) and another flow component. For the purpose of examples we will compare the consequences of trades off between component “d” and each of the curves shown in Figure 4-2a, b or c. This is analogous to a market between two traders as depicted in Figure 1. We can model this process by plotting the marginal benefit of each flow component on reverse axis, as shown in Figure 4-4. Recall equilibrium (and the economically efficient solution) occurs where the marginal benefit curves intersect. When the alternate environmental flow component is a threshold value (Figure 4-2a), the environmental manager would allocate 20 ML to flow component A and only 10 ML to flow component d (as depicted in Figure 4-4a). If the alternate flow component is linear (Figure 4-2b), only 12 ML should be allocated to component A with the remainder allocated to component d (as depicted in Figure 4-4b). The case shown in Figure 4-4c is slightly more complicated as the marginal benefit curves intersect in numerous places. To ensure a stable equilibrium point, 17 ML should be allocated to component d so that there are diminishing marginal returns (i.e. when the curve is downwards sloping\(^{19}\)).

\[\text{Figure 4-4: Trade off decisions using marginal benefit curves for an environmental flow (note that the marginal benefit (MB) for component d in a graph is shown on reverse x axis in grey).}\]

\(^{19}\) Note that when there is a convexity in the total benefit function determining the most efficient allocation is more complicated. It is often not clear about the relative merits of the different points where the lines intercept. Further calculations are often required to distinguish between the equilibrium points.
This example demonstrates how environmental response curves can transparently provide maximum gain from water allocated to the environment. The shape of the response curve fundamentally changes decisions about how water is allocated. While there may be difficulties in initially determining the response curve, it is worth persevering because even relatively crude assumptions should lead to improved outcomes compared to current environmental allocation methods.

The remainder of this chapter discusses how the relationship between flow and ecological outcomes can be described. To avoid confusion with the final demand curve, the relationship between flow and ecological outcomes will be referred to as the environmental response curve. The environmental response curve will form the basis of the environmental demand curve once the value of the ecological change is understood. It is within this framework that the environmental response curve can be translated into an environmental demand curve for water.

4.2. A review of the literature on environmental response curves

The concept of environmental flow requirements is very recent and there is little data linking hydrology and ecological response. Because of differing levels of data and funding, each basin uses a different approach (and accuracy) to determine environmental flow. Demands on government resources mean it is not always possible to conduct a detailed environmental flow study prior to setting water allocations (National Competition Council, 2004).

While current environmental flow studies often consider flow–ecological relationships, final recommendations are stated to allow long term planning decisions or meet a regulatory legislative structure. The existing approaches usually recommend a single value or threshold for each flow component, or assess specific flow regimes. However, environmental water outcomes would be better reflected by the marginal value of water to the environment; that is, how the river ecosystem responds to each additional allocation of water. This concept uses the natural flow paradigm, but applies it
differently. The form of the environmental response function describing ecosystem response to river flows, along with the reversibility of environmental impacts, has not been fully established. Nevertheless, where water is limited, response functions are required to make rational decisions about environmental water management (St stewardson, 2005). The concept of ecological response curves is not new, however there is little discussion in the literature about how these curves should be structured to allow transparent decision making. Acreman (2005) suggests that understanding of links between hydrology and ecology has been hampered by differences in research methodology between the two disciplines.

If response curves are to be used in operational decisions about environmental water use, they must be comparable and change with time to ensure natural variation. This is not to diminish the importance of species specific response curves, but rather establishes the separate need for representing environmental water requirements in a form that allows decision making.

Ecological response curves presented in the literature to date tend to be at broad catchment scales (for example, ELOHA described in Arthington et al. (2006); and The Nature Conservancy (2008); Walsh et al. (2005) in the context of ecological response to urbanisation) or specific to an individual species (for example, Extence et al., 1999). Often ecological response curves have been used to try and identify threshold values below which there would be significant changes in river ecology (Acreman, 2005).

A desktop review of existing ecological–flow studies (both within Australia and internationally) showed strong evidence for ecological and geomorphological responses to flow modification in rivers (Lloyd et al., 2004), with 87% of reviewed studies supporting this theory. The review also aimed to quantify the responses of Australian rivers and wetlands to flow modification and, if possible, establish simple relationships or thresholds for flow related ecological change. The study concluded that there is no simple linear relationship between the size of ecological change and the size of hydrological change (severe ecological changes can occur with even relatively small
changes in flow). Loyd et al. (2004) identified three limiting factors in determining a
general relationship between flow and ecological response:

(i) lack of consistency between studies;
(ii) scale mismatches between hydrological change and ecological response; and
(iii) time lags between cause and effect.

Sheldon et al. (2000) develop an approach to estimating ecological reference curves
relating observed ecological condition to predicted ecological condition based on flow
statistics. These flow statistics included annual proportional flow deviation (changes in
monthly flow as a proportion of the expected natural flow), frequency of high flow
events, frequency of medium flow events, and duration of low flow events and cease to
flow events. Flow conditions were compared to the ecological condition for each river to
generate a series of hypothetical relationships between change in a given flow statistic
and ecological condition. The curves are generated assuming a standard response to
flow, whereas “in reality the shape of the response is likely to vary, depending on the
flow attribute in question and the response being examined” (Sheldon et al., 2000, p.
415).

The MFAT (introduced in section 2.4.4) conducts ecological assessment based around a
series of preference curves (with the x-axis defined by a flow element (for example, daily
water depth, flow percentile, duration of inundation) and the y-axis a non-dimensional
index where zero indicates intolerable habitat conditions and one represents ideal habitat
conditions) (Young et al., 2003)\textsuperscript{20}. Preference curves describe the preferred habitat
condition (for example, flood timing) for selected species. A range of different flow
elements define the ecological requirements of a number of groups of species. The
curves are given confidence limits (A to C) based on evidence sourced from literature
through to expert opinion.

\textsuperscript{20} Note the MFAT was not applied to the Goulburn River as the Goulburn River Flows study was
underway at the time.
4.3. Developing Environmental Response Curves

An environmental response curve shows the relationship between flow and ecological outcome. As previously discussed, ecological response to flow is complex and a number of characteristics including volume, timing, duration and frequency of flows are all important. Each environmental attribute of a river system responds differently or relies on different elements of the flow regime, yet all are interconnected. The difficulties in developing environmental response curves are summarized into three main points which are subsequently discussed in detail.

- How to reconcile a variety of environmental attributes
- How to represent the complexity, variability and uncertainty in ecological response to flow
- How to incorporate the variability required in the flow regime

4.3.1. Reconciling a variety of environmental attributes

Measuring “ecological outcome” is difficult because of the large range of environmental attributes in a river and the complex interactions between these attributes. We might begin by considering a response curve for each attribute; for example, describing the variation in Murray cod (an iconic fish in Australia) abundance with change in flow. However, many environmental attributes are co-dependent, as shown by Chee et al. (2005), who mapped the relationships between catfish and various ecological attributes in the Wimmera River system (Australia). Developing response curves to include each of these interactions would be fraught with difficulty. There are few data linking flow volumes to quantifiable ecological responses with most environmental flow studies referring to indicators of river health such as habitat condition.

Nevertheless, for an environmental manager to decide between releasing different flow elements (or water at different times) a number of environmental response curves representing each flow element are required. While the environmental response curves together encompass the variability in the duration, timing and volume of flow required, the tradeoffs between components require a single measure of ecological outcomes. Ultimately, considering the role of the water market, this could be a dollar value for each
attribute. However, it is not yet possible to produce a demand curve (relating flow to dollar value) for each ecological attribute independently. Therefore, to circumvent this problem, an alternate single descriptor of environmental outcome is required. This is then used in deriving response curves for each flow element, to allow comparative decisions.

Possible descriptors can be divided into two categories; those that describe ecological outcome in terms of potential gains and those that describe ecological outcome in terms of potential risks or losses. Ecological potential describes the possible ecological condition of a “well managed, multiple use, sustainable system” (Gordon et al., 2004, p. 238). A scale can then be used to describe the likelihood of achieving this ecological outcome, where 100% means that the rehabilitation vision is achieved and 0% means no ecological gains have been made. The alternative is to use the term “Environmental Risk” to describe the likelihood of environmental damage, where 0% represents no risk of environmental damage, while 100% represents risk of irreversible damage. Similar broad terms to describe ecological response have been adopted in the Downstream Response to Imposed Flow Transformations (DRIFT) method (King et al., 2003) and Ecological Limits of Hydrologic Alteration (ELOHA) (Arthington et al., 2006). An alternative would be to combine the terms and have a scale where 0% represents no change, 100% represents full movement towards a positive environmental outcome and -100% represents a disastrous environmental outcome. A further alternative is that 0% represents certainty that the environmental is not at risk, while 100% means there is certain to be some damage. The scale of the damage is not represented.

The difficulty in having a single descriptor of “environmental outcome” is that, regardless of the term used, without clear definitions it is difficult to turn the descriptor into a vision of the actual impact on a river system. However, for the majority of environmental flows studies that have attempted this (as previously discussed) the approach of a scale of 0 to 1 or 0% to 100% is used to represent environmental risk. Therefore, a similar approach is adopted in this research. The environmental response curves will thus provide a relationship between flow and environmental risk. The final demand curve, once values are incorporated, will represent the price the community is willing to pay to have a
certain level of environmental risk in the river. What level of risk is acceptable at what price?

Rivers have been described as having hierarchical ecosystems, where hydrological change may influence some or all of the various hierarchies.

...rivers may respond to three scales of hydrological behaviour: the flow regime (long-term statistical generalization of flow behaviour); flow history (the sequence of floods or droughts); and the flow pulse (a flow event). If rivers are indeed nested hierarchies then a change in hydrological behaviour at the scale of a flood pulse will (with time) extend throughout the hierarchy (Sheldon et al., 2000, p. 404).

While there may be some risk to the ecosystem based on changes to a flood pulse, these are likely to impact ecological responses at the organism level (individual survival, spawning success etc). Continued changes that result in a change to the flow history will then impact on communities and population levels. If the hydrological change continues, the risk of ecosystem level responses (species changes, and adaptations) becomes significant with a change in the flow regime.

The time scale for the movement of ecological change through this hierarchy, from responses by organisms through to population and community levels and finally to ecosystem level, will also depend on the organism, or group of organisms, in question. Those with relatively rapid generation times (algae, invertebrates, small fish, annual plants) may show ecosystem responses in the same time frame as a population response in a relatively long-lived organism (large fish, perennial plants trees, waterbirds) (Sheldon et al., 2000, p. 404).

This demonstrates some of the many complexities in defining what is actually meant by the term risk and what would constitute a high risk of irreversible damage. Is “irreversible damage” based on changes at the organism level or ecosystem level? This may depend on the objectives for the particular region under consideration. The notion of environmental risk used here is based purely on deviation from natural conditions with no
attempt to quantify how this might manifest in terms of organisms, populations and ecosystems.

**4.3.2. Ecological response to flow**

The information and data available for each river system will constrain the method for relating flow regime to ecological outcomes. Whatever the method, the important concept is that a function is required to relate flow to ecological outcome so that the marginal gains can be understood. Many methods for determining environmental flow requirements are based on the natural flow paradigm (see section 2.4.2). Implicit in methods based on the natural flow paradigm is the assumption that the natural flow regime provides zero environmental risk, which is an unlikely concept especially in terms of evolution. Further clarification may be required where ecological values have adapted to new regimes in highly modified catchments.

Acknowledging that the method will depend on data availability, one approach is to base the shape of the environmental response curve on the habitat rating curve (in other cases, response curves related directly to key species may be more appropriate). A relationship between habitat provision and flow regime, referred to as a habitat rating curve, can be determined by hydraulic characteristics of the river system. A habitat rating curve shows how habitat changes with each additional megalitre of water. Recent environmental flows studies (for example, Cottingham *et al.*, 2007) identify flow stressors, that is, the proximate cause of ecological responses to change in flow regime. These flow stressors are characterised by various metrics. For some ecological components, stress will be directly related to habitat provision. In other instances, flow stressors are related to frequency and duration of particular events. This research focuses on flow stressors that can be described using habitat provision as a measure of likely ecological condition. It should be acknowledged that the response function is for changes in habitat, and not direct biological changes. The premise is that changes in habitat represent likely changes in biological condition and although the ecosystem responses are multifaceted and complex, using habitat response as a surrogate is a practical way forward. The natural flow series for a reach can be translated into a series of the habitat metrics using the
relationship of the habitat rating curve. This will show the natural frequency of habitat provision, where the natural frequency can set the limits of the environmental response curve.

Figure 4-5 demonstrates an environmental response curve developed from habitat rating curves. The process is first described conceptually and then applied to the Goulburn River in Section 4.4. Figure 4-5a shows an example habitat rating curve. The figure shows that as flows increase to around 500 ML/d, the area of habitat (measured in area of habitat available (m²) per metre of river length) also increases but then decreases. In Figure 4-5b the habitat rating curve translates the natural flow series into a habitat series, from which the frequency of area of habitat under natural conditions can be determined. In this example, referring to the habitat frequency plot in Figure 4-5b, habitat areas of between 3 and 6 m²/m are provided naturally (the graph is predominantly populated between these two points). This corresponds (using the habitat rating curve in Figure 4-5a) to flows of between 30 and 3,400 ML/d. Figure 4-5c shows the information from the habitat rating curve, combined with the natural frequency data, to provide a flow–ecological risk relationship. This relationship defines the environmental response curve.

Flows between 30 and 3,400 ML/d provide habitat areas which are well within the natural range and pose a low risk to the environment (recall that flows between 30 and 3,400 ML/d correspond to habitat between 3 and 6 m²/m). If flows fall below 30 ML/d, the area of habitat falls below 3 m²/m. This rarely occurs under natural conditions and risk is considered to increase in proportion to loss of habitat. Similarly, flows exceeding 3,400 ML/d correspond to a decrease in available habitat area, thus posing a substantial environmental risk. The extremes of the environmental response curve, where environmental risk reaches 100%, are set by expert opinion on known ecological limits.

In cases where the ecology is not well understood, the extremes of the historical flow regime can be used as reference points for 100% risk. The gradient of the environmental response curve, as shown in Figure 4-5c, is based on the proportion of habitat area lost with the change in flow. In other words, the shape of the response curve follows the shape of the habitat rating curve (shown in Figure 4-5a). This is clearly seen with the “spike” that occurs at around 10,000 ML/d, seen in both the habitat rating curve and the environmental response curve.
Recall that, if willingness to pay is constant per unit of environmental risk, the environmental response curve represents the total benefit curve. The environmental response curve shown in Figure 4-5c represents the “total benefit” to the environment. The demand function for environmental water is given by the marginal benefit function (the benefit of each additional megalitre of water to the environment) in Figure 4-5d. In other words, Figure 4-5d is the derivative of Figure 4-5c.

Figure 4-5: Flow chart describing the use of habitat rating curves to generate environmental response curves (a) Habitat rating curve (b) Natural frequency of habitat provision (c) Environmental response curve (d) Marginal environmental response curve (note that the marginal benefit is the change in environmental risk where a reduction in risk is a positive benefit. The log scale means that the gradient of the environmental risk curve between 3,400 and 10,000 appears large in figure c, when in actual fact it is very small).
4.3.3. Variability in flow regime (Changes in the environmental response curve)

The previous section provided an example of a method to develop an ecological response curve from which a curve representing the environments demand for water can be derived. In economic terms, demand for different inputs and outputs depends on a range of factors. For example, an irrigator’s demand for water will depend on the crop’s commodity price, or the prices of alternative farm production inputs to water (for example, feed prices). The same concept applies to the environmental response curve where the ecosystem’s demand for water varies with a range of factors. This is reflected by a shift in the environmental response curve. It is neither necessary, nor perhaps desirable, that habitat be maximized in all time steps as this would create very artificial conditions for the biota of interest.

Long-term studies of naturally variable systems show that some species do best in wet years, that other species do best in dry years, and that overall biological diversity and ecosystem function benefit from these variations in species success (Poff et al., 1997, p. 8).

Both the intra- and inter-annual variation in flow provide the habitat dynamics that are important to maintain biological diversity and ecosystem function. The recent Goulburn environmental flow study developed recommendations that specifically emphasized the importance of inter-annual variability, describing each flow requirement using a range of flow magnitudes and recurrence intervals (Cottingham et al., 2007).

In the following sections, factors that can influence the environmental response curve are discussed along with a description of how they might shift the curve. In other words, factors other than flow in a particular time step are being introduced to the response curve function. The extent to which these factors influence the environmental response curve will depend on the ecological values and flow regime under consideration.

Recall that, response curves (as demand curves) are dynamic; that is, demand for goods change. Similarly, an environmental response curve represents the demand for water at that particular time step. Each response curve then informs how flow and environmental
risk are related at future points in time. For example, if the flow regime in previous years posed a low risk, it may be acceptable to release less water in the next period.

**Diversion from Natural variability**

If the response curve does not shift, the same flow (or similar flow) would tend to be prescribed every year (assuming curves for other flow elements also remained constant). The base curve (Figure 4-5c) shows that a flow of 30 ML/d would be preferable as it provides maximum total benefit, with no additional benefit of providing higher flows (Figure 4-5d). A shifting response curve is required to capture the variability that occurs in natural systems.

The variability is described by the frequency distribution of habitat over the range of natural flows. The frequency distribution of flows in some “current” series (up to a particular point in time) can be compared to the natural situation (Figure 4-6). When a specific habitat area is occurring more frequently than it would under natural conditions (point a in Figure 4-6), the risk of maintaining a corresponding flow should increase to discourage providing these flows in future (i.e. the slope of the environmental response curve must increase). Similarly, when a specific habitat area is occurring less frequently than under natural conditions (point b in Figure 4-6), the risk of providing the corresponding flows should decrease in future years (i.e. the slope of the response curve must decrease). The importance of variability is incorporated in the demand curve by comparing the current frequency distribution of habitat to the natural frequency distribution.

![Figure 4-6: Frequency distribution of habitat provision under current and natural conditions](image-url)
The shift in the environmental response curve can be based around the percentage change from natural (positive or negative) to current habitat provision. The application or degree of the shift might use expert opinion and the level of variability in the particular system. This requires further exploration.

This same process can be applied for each time step to obtain a new response curve. The effect of this continuous adjustment in the response curve accounts for the desirability of natural variation in flow, as advocated by the natural flow paradigm.

This concept can be further developed to specify that high flow events occur in the naturally wetter years by presenting different frequency distribution plots for average, wet and dry year conditions. The response curve would then shift to reflect the range of likely flows in each year type.

![Figure 4-7: Frequency distribution of habitat provision by year type](image)

**Ecological resilience and antecedent conditions**

There is a maximum duration that each element of an ecosystem can survive without required flow levels. Beyond this duration, some elements will be permanently lost. A management decision (or value decision) to persist with a particular flow state will depend on the value of ecological services threatened because their maximum duration is being approached. The risk of the decision can be described by values or weightings placed on each environmental response curve. The purpose of the ecological response curve is to show risk to the current ecology when flow is not provided, but not to make the value based decision.
As the critical duration (where recovery will no longer be possible) is approached, the risk to the ecosystem increases. With each time step, the slope of the response curve will adjust to reflect the increasing risk provided by the progressive increase in duration between required events. To demonstrate how this might be applied, we return to the previous example. If some population (e.g. fish or macroinvertebrate) will be irreversibly damaged when a particular habitat area is not provided for three years, the response curve may change as shown in Figure 4-8. As the years in which flow requirements are unmet increase, the environmental response curve becomes steeper; in this example, within two years, the gradient of the curve is vertical. This implies that if the required flow is provided in the following year, there will be no consequence, but in all other cases (even if part of the flow is provided) there is 100% environmental risk (i.e. high risk of irreversible environmental damage). This demonstrates the increasing importance of providing adequate flow as the duration between events increases.

In respect of risk alleviation, some decision is required as to what constitutes “provided” or “not provided”. For example, if risk is below 20%, is this considered adequate? Similarly, once a system is stressed for an extended period, the ability to recover will depend on the resilience of the system. Once the critical duration is reached, and adequate habitat has not been provided, a readjustment of the base curve may be required. If the ecological value has been irreversibly affected, the shape of the response curve will change or indeed become redundant.
Event duration

Many environmental flow requirements specify both duration and volume. For example, freshes must be of adequate duration to ensure correct ecological triggers. If an investment to provide part of the flow event has been made, then further investment should be based on the cost effectiveness of providing flow at the next time step. The response curve only changes when events have been provided recently (that is, if an X duration event has not been provided in the past X years then the risk increases).

It is possible that the relationship between required duration and flow could also be represented by a curve, so that the result would be a 3D solution surface for environmental outcome as a result of flow magnitude and duration.
Interaction between flow components

Rating curves describing ecologically significant flow components should not be considered in isolation. Many flow components (or habitat rating curves) may describe elements that are important for maintaining the same objective in the river system (for example, target fish or target water quality). Failure or success in providing one flow component may affect the success of another flow component. In other words, if a particular flow component \((X)\) is not provided at time \(t\), it may impact the shape of the response curve of another flow component \((Y)\) at time \(t+1\). This is an extremely important aspect of environmental flow provision and one of the complexities in determining environmental water requirements. While it is acknowledged that this is a key issue, it has not been directly addressed in this thesis. This thesis aims to demonstrate a process, and layers of complexity would need to be added at a later stage. Interaction between flow components is one such layer.

4.4. Environmental Response Curves for the Goulburn River

In the previous sections, the concepts underpinning environmental response curves were discussed\(^{21}\). Here, the concepts are applied to the Goulburn River. The environmental flow study made flow recommendations for five reaches. In this study, the Goulburn River is being represented by only two reaches; one immediately downstream of Lake Eildon and one immediately downstream of Goulburn Weir. Therefore, environmental response curves are developed for each of the relevant flow elements for the environmental flow study Reach 1 and Reach 4 only. To avoid confusion later in this thesis, the reaches will now be referred to as follows:

- Reach 1: downstream of Lake Eildon; and
- Reach 2: downstream of Goulburn Weir.

These response curves are being developed as inputs to a conceptual model to test how different aspects of the response curves inform allocation decisions, and is limited to data

\(^{21}\) Recall that Environmental response curves represent the total benefit of providing water to the environment, rather than the marginal benefit.
readily available through the Goulburn Environmental flows study (Cottingham et al., 2003). It is anticipated that these curves will require further refinement from ecologists at a later stage. The optimisation modelling (outlined in Chapter 5) will inform which elements of the response curves will require the most attention when this review occurs.

Section 4.2 discussed a range of factors that may cause the response curve for a particular flow element to shift. In developing response curves for the Goulburn, the process has been simplified. Instead of continuous shifts in the environmental response curves, there are a number of “states” that the environmental response curves can move between. These include climatic conditions (wet, dry or average) and the number of years since flow was last provided (last year, two year ago, three years ago). The definitions of whether a flow component has been “provided” are based on the duration requirements for each flow component.

The environmental flows study identified a range of different flow stressors, each related to different habitat curves. Flow elements that occurred in the final recommendations and related to a number of objectives in the environmental flow study have been developed into environmental response curves to incorporate into the optimisation model. The key flow elements are listed in Table 4-1.

Table 4-1: Environmental flow elements for each reach

<table>
<thead>
<tr>
<th>Reach 1 – downstream of Lake Eildon</th>
<th>Reach 2 – Downstream of Goulburn Weir</th>
</tr>
</thead>
<tbody>
<tr>
<td>High water velocity (summer low flow)</td>
<td>Shear Stress (summer and winter)</td>
</tr>
<tr>
<td>Riffle habitat</td>
<td>Slow shallow habitat</td>
</tr>
<tr>
<td>Shallow water habitat</td>
<td>Deep water habitat (summer and winter)</td>
</tr>
<tr>
<td>Wetland inundation</td>
<td></td>
</tr>
</tbody>
</table>

As discussed in Section 3.5.4, the environmental flow study for the Goulburn Basin used the Flow Events Method, producing habitat rating curves for each flow element. The response curves are based on the habitat rating curves and have been adjusted to represent wet, average and dry conditions based on the level of habitat that would have been naturally available (from frequency plots). As previously discussed, the exact application depends on the data available for that particular flow element. Because of the many
limitations in the method and the subjective nature of the final curves, the curves used in the model were reviewed and discussed with two members of the expert panel for the Goulburn Environmental Flows study (Roberts, J., 2008, pers. comm., 28 May; and Hillman, T., 2008, pers. comm., 24 June).

The environmental response curves were developed to demonstrate the concept of using an optimisation approach to make allocation decisions with regard to environmental water. In Chapter 5, the optimisation model is used to test how sensitive the decision choices are to the shape of the environmental response curves. This will inform priorities for, and how much more detail is required in, further research into developing these response curves.

The method used to develop response curves for Reach 1 and 2 are described in sections 4.4.1 and 4.4.2 respectively.

4.4.1. Reach 1 – Downstream of Lake Eildon

Velocity

Macrophytes (aquatic plants living near or in water) can be affected by flow velocities but their ideal range for growth and expansion is 0.1 – 0.6 m/s (Cottingham et al., 2003). This is particularly important in the summer and autumn months when plants are in their most productive phase. Slow velocity water (< 0.1 m/s) allows biofilm and algae to collect on plant leaves and diminishes CO₂ uptake. Sustained periods of slow velocity will eventually deplete underground energy supplies (starch), causing irreversible damage and death (Roberts, J., 2008, pers. comm., 28 May). Thus the duration of slow velocity poses a threat but there is little quantifiable evidence as to what critical duration actually threatens the resilience of macrophytes to slow velocities. However, expert opinion provided the following approximations:
- Slow velocities could be tolerated for one month per year, with recovery occurring in the other summer months of the same year.\(^{22}\)
- If slow velocities lasted for 4 months in a year, growth would be arrested. Moderate velocities (range of 0.1 – 0.6 m/s) would be required in the following year to ensure regeneration.
- The risk of irreversibly damaging macrophytes would be high if slow velocities occurred throughout the summer and autumn months (more than 4 months per year), and re-colonisation would be required.

Fast water (between 0.6 – 0.9 m/s) causes mechanical damage to the plants by stripping their foliage. As long as the bed itself remains intact, the plants will regenerate when flow velocities become favourable. Duration of high velocities is not important as a single event can damage plant foliage. Moderate flows are required in the following year to ensure that macrophytes regenerate. However, if high velocity flows disrupted the bank and the underground root network and stores are destroyed, irreversible damage is likely and recolonisation would be required (Roberts, J., 2008, pers. comm., 28 May). In the Goulburn River downstream of Eildon, it is assumed that bed disturbance starts at a velocity of 1.9 m/s, which is the maximum velocity experienced under natural conditions. The timescale and complexities of macrophyte re-colonisation are poorly understood. Nor is it certain that they would regenerate if this stage was reached. As macrophytes provided habitat and nutrients for a range of fauna, their loss may cause long term consequences in other aspects of the ecosystem. This was therefore chosen to represent the endpoint, or 100% risk, in the environmental response curve.

This exemplifies the complexities in defining an endpoint or point of “irreversible” damage. The frequency distribution plot in Figure 4-9b shows that velocities of 1.9 m/s were reached in a very small number of years. Furthermore, the current presence of macrophytes in the Goulburn River shows that they were not eliminated by these high velocities and (at least partially) recovered from them. Any environmental flows method

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\(^{22}\) Note that winter flows that are high enough to clear biofilm or algae, significantly increase the chance of full recovery in summer months. This interaction has not been included in the environmental response curves.
or decision process is plagued by these questions of end points and the unknowns that surround the ecosystems response. In this approach, 100% environmental risk really represents “risk” rather than a defined state. At these velocities, based on the best information available, the risk of irreversible damage is extremely high and management of environmental water should attempt to avoid these risks.

Figure 4-9 shows the development of environmental response curves for velocity. Figure 4-9a shows the habitat rating curve (the relationship between flow and velocity). Flows considered to provide velocities favourable to macrophytes (0.1 – 0.6 m/s) are considered to pose 0% environmental risk. Flows at which bed disturbance commence is pose an irreversible disturbance and thus flows above this trigger pose 100% risk. Between these defined points, the risk will vary depending on the climate and the number of years since favourable flows have been provided. These risks have been developed using the following approach:

- For the base curve (flows are provided in the previous year, \( r = 1 \)), the slope of the curve between 0% risk and 100% risk at the higher flows (or velocities) is based on the proportional increase in velocity. Thus the shape varies depending on the climate type (wet, dry or average) due to the value set for the upper limit. In a wet year, the upper limit is set to 1.9m/s. In a dry year, under natural conditions, velocity does not exceed 0.9m/s and in an average year, velocities do not exceed 1.2m/s. These values are considered to pose 90% risk.
- At lower velocities, while macrophytes will survive, it is assumed there is still some risk from slow velocities. A risk of 40% at zero flow was chosen to represent this.
- If adequate flows have not been provided for two years \( (r=3) \), the response curve acts as a trigger. In other words, if velocities are not between 0.1 and 0.6 m/s, the system will fail. This is the same for wet, average and dry years.
- The response curves for one year \( (r=2) \) since favourable flows were provided are derived by linear interpolation between \( r = 1 \) and \( r = 3 \).
Figure 4-9: Developing Velocity Response Curve (a) Habitat Rating Curve (Cottingham et al. 2003) (b) Proportion of time velocity occurs in dry, wet and average years (Summer and Autumn) (c) Environmental response curve for an average, dry and wet year (d) Dry year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (e) Average year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (f) Wet year Environmental Response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided
**Riffle Habitat**

Increased flows downstream of Lake Eildon have drowned out much of the riffle habitat in Reach 1. Riffle habitat is important for fish and macroinvertebrates, and is a fast-flowing zone with turbulent flow (Froude number > 0.18) at the surface and a depth of less than 0.3m (Jowett, 1993).

Figure 4-10 shows the development of environmental response curves for riffle habitat. Figure 4-10a shows the habitat rating curve (the relationship between flow and riffle habitat area) and Figure 4-10b shows the frequency of habitat provision over summer and autumn, which are the relevant months for this flow component. Flows providing habitat area well within the natural range are considered to pose 0% environmental risk. As long as some riffle habitat is available, macroinvertebrates and smaller fish should be maintained. Therefore, only when riffle habitat area falls below 1 m$^2$/m is there considered to be 100% risk. Between these defined points, the risk will vary depending on the climate and the number of years since favourable flows have been provided. These risks have been developed using the following approach:

- For the base curve ($r=1$), the slope of the curve between 0% risk and 100% risk at the higher flows (or velocities) is based on the proportion of habitat lost as flow decreases. Thus, the shape will vary depending on the climate type (wet, dry or average). In dry and average years, habitat areas range mostly between 3 and 6 m$^2$/m (corresponding to flows of between 30 and 3400 ML/d) and in wet years, habitat area ranges from 1 to 6 m$^2$/m.
- If adequate flows have not been provided for two years ($r=3$), the response curve acts as a trigger. In other words, the response curve is vertical between 0% risk and 100% risk.
- The response curves for one year ($r=2$) since favourable flows were provided is derived by linear interpolation between $r = 1$ and $r = 3$.

Note that there is little data available to determine how many years macro-invertebrates and fish will survive (unaffected) if inadequate areas of riffle habitat are provided. It is assumed here that an unacceptable level of risk is reached after three years without adequate riffle habitat but there is little data to support or challenge this. As with many
of these response curves, this forms only a first attempt at developing the curves. As
knowledge on ecosystem response improves, so too will the environmental response
curves. Furthermore, if modelling demonstrated that decision making was highly
sensitive to duration of inadequate riffle habitat, it would point to the priority for research
to establish the critical duration accurately.

Information regarding the duration and consistency of riffle habitat required is also
limited (Hillman, T., 2008, pers. comm., 24 June). It is assumed here that adequate
habitat must be provided for at least four out of the six autumn and summer months.
Figure 4-10: Developing Riffle Habitat Response Curve (a) Habitat Rating Curve (Cottingham et al. 2003) (b) Proportion of time habitat occurs in dry, wet and average years (Summer and Autumn) (c) Environmental response curve for an average, dry and wet year (d) Dry year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (e) Average year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (f) Wet year Environmental Response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided.
**Shallow water habitat**

Shallow water habitat (referring to areas with a depth of less than 0.3 m) is important for in-channel macrophytes and low-flow fish recruitment in summer. The shallow habitat is important for macrophytes, allowing light penetration for growth. Because of the warmer temperature, summer is when macrophyte grow and fish are recruited, and is thus the preferred season for low flows. The environmental flows study recommended January, February and March and thus provision of shallow water habitat requirements are only relevant in these months.

Little is known about the actual depth required by macrophytes and fish during this time, and the current environmental flow recommendations are based on the natural flow paradigm. Adopting the same philosophy, the environmental response curves are based on the habitat rating curve in combination with the natural frequency of habitat provision (the method as outlined in Chapter 4.3.2).

The following information was used to construct the response curves:

- In average and dry years, habitat ranges between 5.25 and 7 m\(^2\)/m (corresponding to flows of between 90 and 2000 ML/day). In wet years, habitat ranges between 4 and 7 m\(^2\)/m (corresponding to flows of between 15 and 2000 ML/day). Flows in these ranges (for the corresponding year type) are considered to pose zero risk.
- In dry years habitat areas occasionally reach as low as 4.75 m\(^2\)/m and in average years as low as 4 m\(^2\)/m. Habitat areas beyond this limit are assumed to pose 100% environmental risk as they are outside the range of natural occurrence. In wet years, habitat areas reach as low as 2.5 m\(^2\)/m.
- If flow was provided in the previous year \((r = 1)\), the risk between 0% and 100% is based around the percentage habitat lost (in other words, follows the shape of the habitat rating curve).
- If adequate flows have not been provided for two years \((r=3)\), the response curve acts as a trigger (i.e. flows must be between 90 and 2000 ML/d in dry and average years and between 15 and 2000 ML/d in wet years).
- The response curves for one year \((r=2)\) since favourable flows were provided is derived by linear interpolation between \(r =1\) and \(r =3\).
Figure 4-11: Developing Slow Water Habitat Response Curve (a) Habitat Rating Curve (Cottingham et al. 2003) (b) Proportion of time habitat occurs in dry, wet and average years (Summer and Autumn) (c) Environmental response curve for an average, dry and wet year (d) Dry year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (e) Average year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (f) Wet year Environmental Response curve with 0yr (base), 1yr and 2yr since adequate.
**Wetland Inundation**

Lake Eildon and the regulation of flows to accommodate irrigation requirements has caused a major reduction in floodplain and wetland inundation along the Goulburn River.

The Goulburn Environmental Flows study developed a relationship between flow in the Goulburn River and the area of different types of wetlands to be inundated (Figure 4-12a). The relationship shows that at flows of 15,000 ML/d in the Goulburn, small areas of shallow freshwater marsh, deep freshwater marsh and freshwater meadows are inundated. At flows of 40,000 ML/d the majority of deep freshwater marsh and freshwater meadows have been inundated, with around 75% of shallow freshwater marsh areas inundated. At 60,000 ML/d effectively all areas have been inundated. The difficulty in defining a response curve arises from the fact that the area of wetlands inundated is predefined and spatially distributed. While a flow of 40,000 ML/d will inundate 75% of wetlands, the same 25% of wetlands will miss out on watering in every instance. Thus, a range of different flows are required to ensure the survival of the full range of habitats. It is the frequency, or recurrence, of these events that is important.

A relationship between discharge and average recurrence interval (ARI)\(^\text{23}\) was developed as part of the Goulburn Environmental Flows Study. The important concept is how large a change in the recurrence interval would cause a change in the type of wetland or habitat available. An initial attempt to describe the potential shift in the array of wetlands is provided below (Roberts, 2008). These are based on the overlap in types of vegetation present in wetlands with different inundation frequencies.

- Wetlands naturally inundated every year (a recurrence interval of 1:1 years) would begin to change condition if inundated every second year (a recurrence interval of 1:2 years). Any further increase in time between inundations would pose a high risk of irreversible change.
- Wetlands naturally inundated between every second year (a recurrence interval of 1:2 years) and every fifth year (a recurrence interval of 1:5 years) can move comfortably within this range of intervals (e.g. only minor changes would occur if

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\(^{23}\) *The average (or expected) time duration between flows exceeding a specified magnitude.*
a wetland naturally inundated every second year become inundated only every fifth year). An increase to every tenth year would pose a high risk of irreversible change.

- Wetlands currently inundated every tenth year (a recurrence interval of 1:10 years) are at high risk if the inundation frequency increases beyond this. Limited scientific data available suggests that biodiversity loss through reduced seed banks occurs if recurrence intervals extend beyond 10 years.

Based on the above principles, an upper limit can be placed on the recurrence intervals acceptable for different flows (as shown in Figure 4-12c), and this would constitute 100% risk.

Defining a response curve based on recurrence intervals is problematic when constructing a single year optimisation model. ARI is a statistic calculated from long term data sets and a reduced data set will produce different approximation of the ARI. In environmental flows studies, the frequency of events over a long term data set is often expressed as the probability of an event happening one year in X years. This is interpreted to mean that if flow has not been provided in X-1 years, then the event should occur in year X. While this ensures that the same ARI occurs, the pattern of events (or allowable duration between events) is artificial.

The year types used in the model (dry, average and wet) are based on 100 years of data and divided by the 20th percentile and 80th percentile annual inflows. Wet and dry years therefore occur every one in five years, while average years make up three out of five years. The recurrence interval of different climatic year types is therefore used to guide wetland inundation to occur at the correct ARI. The natural variation (or pattern) of wet, average and dry years will therefore define when different floodplain events occur. This will minimise the introduction of an artificial watering pattern. In a single year model, this may force large events to happen more frequently than required. However, in a multi-year model (even of short duration) the model will always attempt to provide smaller events over large events where possible, and is therefore more likely to follow the natural ARI.
The following were used to guide the environmental response curves:

- The environmental flow study recommended an annual flood of at least 15,000 ML/d (a doubling of the recurrence interval as this size flood would naturally have occurred twice a year). Flows below this are assumed to pose 100% risk.
- In dry years, flows of 34,547 ML/d must occur every year (a recurrence interval of 1:1 years) to ensure zero risk (the natural recurrence interval). Interpolation via a linear curve is used to connect 100% risk and 0% risk. This is based on the relatively linear relationship between wetland area inundated and flow (Figure 4-12a). If an adequate flow was not provided in the previous year, a minimum flow of at least 34,547 ML/d is required to ensure that the reoccurrence interval does not exceed a recurrence interval of 1:2 years.
- In average years, events of at least 47,793 ML/d must occur to ensure 0% risk (corresponding to an event that naturally occurred 1:2 years). A linear curve is used to connect 100% risk and 0% risk. If an adequate flow was not provided in the previous year, a minimum flow of at least 47,793 ML/d is required to ensure that the reoccurrence interval does not exceed 4 years.
- In wet years (occurring one in five years), events of at least 59,828 ML/d must occur to ensure 0% risk (corresponding to the natural 1:5 year event). If adequate flows were not provided in the previous year (assuming this will occur approximately every second wet year), flows of at least 68,929 ML/d must occur to ensure 0% risk (corresponding to a natural recurrence interval of 1:10 years). If adequate flows have not been provided in the previous two years, a minimum flow of 68,929 ML/d is required.

Note that this approach assumes that the frequency of wet and dry years will continue at historical frequencies. However, climate change is likely to alter the frequency of these events. If these changes are to be reflected in the natural recurrence interval of floodplain events, the reduced frequency of dry and wet years should be incorporated into the response curves. This has not been addressed in this research.
Determining the required duration of events to cause wetland inundation is problematic. While the flow at which inundation commences is known, some floodplain areas will pond water long after inflows have ceased and other areas will require ongoing inflows to maintain adequate habitat over a long enough duration. The relationship between the flow volume and duration of event is not well understood.
Figure 4-12: Developing wetland inundation Response Curve (a) Area of wetlands of various types filled by increasing flows in the Goulburn River (Cottingham et al. 2003) (b) ARI of floodplain events (c) Environmental response curve for an average, dry and wet year (d) Dry year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (e) Average year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (f) Wet year Environmental Response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided.
4.4.2. Reach 2 – Downstream Goulburn Weir

Shear stress

Shear stress is a flow element affecting geomorphological processes and impacting on macroinvertebrate conditions. The literature suggests that composition and numbers of macroinvertebrates change in response to sediment loads, with increased sediment deposits causing a decrease in abundance of some species (Hogg & Norris, 1991). Fine sediment that settles on the substrate and surfaces of snags and vegetation interferes with plant and particularly biofilm production, and also has a direct effect on macroinvertebrates. In the environmental flow study (Cottingham et al., 2003), flow stressors were developed related to the amount of time that shear stress is above or below a certain level. As recommendations were made on a seasonal basis in the environmental flows study, environmental response curves were prepared for summer/autumn (Figure 4-13) and winter/spring (Figure 4-14) separately. The habitat rating curve for shear stress is shown in Figure 4-13a and Figure 4-14a (it will obviously not vary across seasons as it is based on hydraulic characteristics of the river). The frequency distributions plotted in Figure 4-13b and Figure 4-14b show that there are significant differences in the range of shear stress provided naturally in summer and autumn compared to winter and spring months.

The following were used to guide the environmental response curves:

- In an average year, during autumn and summer months, shear stress ranges between 2.5 and 7.25 N/m². In a dry year, shear stress ranges from 1.5 to 5 N/m², occasionally reaching as high as 6.5 N/m². Shear stress ranges from 3.25 to 7.5 N/m² in wet years. In winter-spring, average year shear stress ranges between 4.5 and 7.5 N/m², with values extending up to 11 N/m² at a lower frequency. In dry years, shear stress ranges from 4.5 to 7.5 N/m², occasionally reaching as low as 3.25 N/m². Shear stress ranges from 4 to 11 N/m² in wet years. Habitats within this range (and corresponding flows) were assumed to pose 0% risk. Note that the minimum values are based on the natural flow series and the habitat rating curve. This is consistent with the method used elsewhere in the study. However, levels of shear stress vary spatially across a river. There will always be areas of low
shear stress at the river margins, or even on the floodplain during high flows. It is
the high shear stress levels that are of greater concern.

- There is little information available for determining the maximum level of shear
  stress that macro-invertebrates will tolerate (Hillman, T., 2008, pers. comm., 24
  June). As macro-invertebrate communities are currently present, it is assumed
  that they can survive the maximum shear stress presented under the current flow
  regime. Using historical data downstream of Goulburn Weir, the maximum value
  occurring under current conditions was calculated as 19 N/m² for summer and
  autumn, and 45.5 N/m² for winter and spring. Shear stress values exceeding the
  maximum values experienced under current conditions are assumed to pose 100%
  risk. (It should be noted that these values are significantly higher than the 90th
  percentile values).

- For the base curve (r=1), the slope of the curve between 0% risk and 100% risk at
  the higher flows (or velocities) is based on the proportion of habitat lost as flow
  decreases.

- If adequate flows have not been provided for two years (r=3), the response curve
  acts as a trigger i.e. The response curve is vertical between 0% risk and 100%
  risk.

- The response curves for one year (r=2) since favourable flows were provided are
  derived by linear interpolation between r = 1 and r =3.

Once again, there is little data available for determining the number of years
macroinvertebrates and fish will survive (unaffected) if high levels of shear stress occur.
It is assumed here that an unacceptable level of risk is reached after three years of high
levels of shear stress.

Information regarding the duration of high shear stress that can be withstood is also
limited. It is likely that a short burst of high energy would have little impact assuming
the macroinvertebrates are shifted but allowed to re-settle and the biofilm is not seriously
dislodged (Hillman, T., 2008, pers. comm., 24 June). It is assumed here that adequate
habitat must be provided for at least eight months of the year (four months over autumn
and summer combined and four months over winter and spring combined).
Figure 4-13: Developing Shear Stress Response Curve for Summer and Autumn months (a) Habitat Rating Curve (Cottingham et al. 2003) (b) Proportion of time habitat occurs in dry, wet and average years (c) Environmental response curve for an average, dry and wet year (d) Dry year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (e) Average year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (f) Wet year Environmental Response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided.
Figure 4-14: Developing Shear Stress Response Curve for Winter and Spring months (a) Habitat Rating Curve (Cottingham et al. 2003) (b) Proportion of time habitat occurs in dry, wet and average years (Summer and Autumn) (c) Environmental response curve for an average, dry and wet year (d) Dry year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (e) Average year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (f) Wet year Environmental Response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided.
**Slow shallow habitat**

Slow shallow habitat, an important component of fish habitat, is an area where the depth is less than 0.5m and velocity less than 0.05 m/s. It is relevant over summer months. The timing of provision of slow shallow habitat for fish is complicated by lumping “fish” into a single category (which, some argue is a failure of habitat models (Gordon *et al.*, 2004)). Larger fish require slow shallow habitat for spawning and thus require only short durations, however at very specific times. Whereas smaller fish survive predominately in the slow shallow zone (Hillman, T., 2008, pers. comm., 24 June). Slow shallow areas also tend to be on the edge of streams, and are important for the collection and uptake of nutrients entering the system. Thus provision of slow shallow habitat should also be considered in conjunction with drying cycles to allow organic matter to collect at the stream edge and the rate of change of flow levels. These aspects have not been considered here in the development of the slow shallow habitat response curve.

Figure 4-15 shows the development of the slow shallow habitat response curves. The slow shallow habitat rating curve is shown in Figure 4-15a. It should be noted that the habitat rating curve for this flow element only has data for flows up to 4450 ML/d (Cottingham *et al.*, 2003). It appears that the slow shallow habitat would then decrease with higher flows. For the purposes of the environmental response curve, a linear decline to zero area at flows of 86,400 ML/d is assumed. This is based on the shape of the shallow habitat rating curve for Reach 1 as a similar form is expected.

Figure 4-15b shows the frequency of habitat provision. In all year types, slow shallow habitat area ranges from 2 m²/m up to around 6.5 m²/m (corresponding to flow in the range of 500 ML/d up to 5850 ML/d). Flows within this range are considered to pose 0% risk. Although some variation in the frequency of habitat occurs within this range, it is not considered enough to distinguish between wet, dry and average years. The environmental response curve for slow shallow habitat is thus the same in all year types. It is assumed that a minimum habitat of 1 m²/m is possible before 100% risk is reach.
For the base curve \((r=1)\), the slope of the curve between 0\% risk and 100\% risk at the higher flows (or velocities) is based the proportion of habitat lost as flow decreases.

If adequate flows have not been provided for two years \((r=3)\), the response curve acts as a trigger. In other words, the response curve is vertical between 0\% risk and 100\% risk.

The response curves for one year \((r=2)\) since favourable flows were provided are derived by linear interpolation between \(r=1\) and \(r=3\).

Once again, there is little data available to determine how many years fish will survive (unaffected) if inadequate areas of slow shallow habitat area are provided. It is assumed here that an unacceptable level of risk is reached after 3 years of inadequate habitat.

Information regarding the duration of the required slow shallow habitat is also limited (Hillman, T., 2008, pers. comm., 24 June). It is assumed here that adequate habitat must be provided for at least two out of the three summer months.
Figure 4-15: Developing Slow Shallow Habitat Response Curve (a) Habitat Rating Curve (Cottingham et al. 2003) (b) Proportion of time habitat occurs in dry, wet and average years (summer) (c) Environmental response curve for an average, dry and wet year (d) Average year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (e) Dry year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (f) Wet year Environmental Response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided.
Deep water habitat

Deep water habitat is an important flow element for large fish (e.g. Crook et al., 2001). Deep water habitat is defined in the Goulburn Environmental flows study as areas with a depth greater than 1.5m (Cottingham et al., 2007). The original environmental flows study recommended a minimum flow of 610 ML/d or natural (corresponding to a habitat area greater than 10m²/m) (Cottingham et al., 2003). The recommendation is relevant all months of the year. In the later environment flow study, recommendations were made on a seasonal basis (Cottingham et al., 2007). Therefore, environmental response curves have been developed separately for summer and autumn (Figure 4-16) and winter and spring (Figure 4-17).

The habitat rating curve for deep water habitat is shown in Figure 4-16a and Figure 4-17a (it will obviously not vary across seasons as it is based on hydraulic characteristics of the river). The frequency distributions shown in Figure 4-16b and Figure 4-17b show that there are significant differences between the range of naturally provided shear stress in summer/autumn and winter/spring months. It is assumed that larger areas of deep water habitat provide increasing environmental benefits. Therefore, upper limits are not included in the environmental response curves. During summer and autumn months, in average and dry years, deep water habitat areas rarely fall below 24 m²/m (650 ML/d). In wet years, the lower limit is 34 m²/m. In winter month and spring months, these limits increase to 36 m²/m in a dry year, 48 m²/m in an average year and 58 m²/m in a wet year. Flows providing habitat levels above these values are assumed to pose 0% risk.

The area of deep water habitat required will vary substantially depending on the actual fish species present and the abundance of fish in the area. It is also likely that if there are external pressures (such as fishing), providing a greater number of deep water pools will further protect the fish population (Meredith, S., 2008, pers. comm., 24 June). For the purposes of developing the environmental response curve, a lower limit of deep water habitat area has been set based on the lower limit experienced based on current operations of the Goulburn Weir. In summer and autumn, the minimum area of habitat experienced is 5.6 m²/m and in winter and spring, the minimum area of habitat experienced is 4.5m²/m. A value of 5m²/m of deep water habitat is taken to represent 100% risk.
For the base curve (I=1), the slope of the curve between 0% risk and 100% risk at the higher flows (or velocities) is based the proportion of habitat lost as flow decreases.

If adequate flows have not been provided for two years (I=3), the response curve acts as a trigger i.e. The response curve is vertical between 0% risk and 100% risk.

The response curves for one year since favourable flows were provided (r=2) are derived by linear interpolation between $r = 0$ and $r = 3$.

Once again, there is little data available to determine how many years fish will survive (unaffected) if inadequate areas of deep water habitat area are provided. It is assumed here that an unacceptable level of risk is reached after three years of inadequate habitat. It is assumed that adequate habitat must be provided for at least six months of the year (three in summer months and three in winter months).
Figure 4-16: Developing Deep water Habitat Response Curve for Summer and Autumn (a) Habitat Rating Curve (Cottingham et al. 2003) (b) Proportion of time habitat occurs in dry, wet and average years (c) Environmental response curve for an average, dry and wet year (d) Average year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (e) Dry year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (f) Wet year Environmental Response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided.
Figure 4-17: Developing Deep water Habitat Response Curve for Winter and Spring (a) Habitat Rating Curve (Cottingham et al. 2003) (b) Proportion of time habitat occurs in dry, wet and average years (c) Environmental response curve for an average, dry and wet year (d) Average year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (e) Dry year Environmental response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided (f) Wet year Environmental Response curve with 0yr (base), 1yr and 2yr since adequate habitat was provided.
4.5. Valuing the response curve – creating a demand curve

While environmental response curves provide information that assists in allocating a parcel of environmental water, an environmental demand curve (including a value assessment) is required to make allocation decisions between the environment and consumptive users. While this is not the focus of this research, it is included since value assessments on different environmental assets could also improve the allocation of environmental water between environmental flow components and reaches.

Chapter 2.4.3 highlighted some of the approaches and the related problems in determining the value of providing water for the environment. Despite these difficulties in applying non-market valuation techniques to develop environmental demand curves, it remains the most promising method. Even if it is assumed that the community is willing to pay a constant price (or has a constant “value”) of environmental water, the ecological response curves will allow tradeoff decisions between environmental flow components for the purpose of providing “non-use” benefits such as existence value. As more information becomes available about the values in a specific catchment, these can be built into the demand curve. In the simplest form, this can be done using value weightings of the ecological response functions. The ecological response functions still provide information about the marginal value of water to the environment. The exact dollar values become more important when trying to make decisions about how much should be spent on providing environmental flows and how tradeoffs with consumptive use should be made.

In the absence of a contingent valuation study for the Goulburn catchment\textsuperscript{24}, it is difficult to translate the environmental response curves developed into demand curves. However, to demonstrate the process of develop a demand curve we can look at the riffle habitat environmental response curve and developing a demand curve using two different

\textsuperscript{24} Note that the Ecosystems Services Project undertook to collate an inventory of ecosystem goods and services provided in the Goulburn Catchment. The report is at a catchment wide scale and not specific to river ecosystem services. The report relates the various services to key land uses or industries in the region. No dollar values are placed on these services. (CSIRO, 2003).
potential marginal value curves. In the first instance, consider a constant willingness to pay (Example 1 in Figure 4-18). The community in this example places the same value on each additional reduction in environmental jeopardy ($1000) (Figure 4-18b). As the willingness to pay is constant per percentage point reduction in environmental risk, the shape of the environmental demand curve will follow the shape of the marginal environmental response curve. It is likely that as the environmental risk decreases (implying that the area of environment in pristine condition is increasing), the community’s willingness to pay for additional improvements will decrease (Example 2, Figure 4-18e. That is, the community has a diminishing marginal value per unit benefit. When there is high risk to the environment, the willingness to pay is higher. With each additional reduction in risk, the willingness to pay reduces.
Figure 4-18: Environmental demand curve for water – Example 1: constant willingness to pay and Example 2: diminishing willingness to pay for water.²⁵

²⁵ Note that for flows between 4450 ML/d and 17250 ML/d the marginal value is negative. However, log scale means that negative values in the marginal response curve plots are small, and the y-axis is therefore only shows the positive scale.
4.6. **Summary**

It is argued in this chapter that the environment should be considered like other water users with a demand curve representing the value of water. Understanding the marginal value of water to the environment will help to achieve the maximum benefit of environmental water by allocating limited available water between flow components or reaches; and, at a higher decision level, to determine how water should be allocated between consumptive use and the environment.

Developing environmental response curves is the first step in forming environmental demand curves. The method used to develop environmental response curves will vary depending on the data available for each catchment. Details of the shape of the response curves will need to develop with time as new knowledge becomes available. While the approach developed here has necessary simplifications, it will be possible to build in layers of complexity and refinement to the method over time.

The concept of developing environmental response curves using habitat rating curves has been introduced. The proposed environmental response curves have a number of key components that will assist in making allocation decisions:

- **A comparable measure of environmental outcome:** in this research, environmental risk has been used as a measure of environmental outcome. Further development defining environmental risk (and what it means on a temporal and spatial level) is still required.

- **Environmental response is dynamic:** environmental water requirements will depend on the provision of flows in previous years and the resilience of the system. For this approach to be adopted, the resilience (how long can it last without water) and the recovery (how many years of adequate flow are required for regeneration after a stress period) of the ecological attribute must be understood.

The environmental response curve is translated into a demand curve by linking the ecosystem services provided from improved instream health to the value the community places on these services. Despite limitations, contingent valuation is the most commonly
used method to value environmental services. Use of environmental demand curves in trade off decisions with consumptive use will require contingent valuation to translate the response curves into dollar values. The link between different flow elements and ecosystem services will also require improved understanding. Without this, the demand curves will only represent the non-use values provided by environmental flows.

While many aspects of the response curve and demand curve must be further developed, this chapter introduces a concept for describing environmental demands that will assist operational decisions for environmental water allocation. Understanding the marginal value of water to the environment will allow environmental water reserves to be managed to optimal community benefit. Understanding the true value of each additional unit of water will provide a more transparent process for prioritizing environmental flow components and making release decisions.

By way of illustration, the approach to developing environmental response curves was applied to the Goulburn River. It is anticipated that these response curves will improve over time as more data becomes available and our knowledge of environmental responses improves. In particular, a review of the resilience (time between events) and recovery potential (speed of regeneration) is required to improve the environmental response curves developed here. It is also likely that while habitat curves provide a starting point, they could be more specifically targeted to individual (or groups of) species requirements (for example, distinguishing between large and small fish). Ideally this could be taken further and linked to a comparative valuation (even if relative) of the different components. The interaction between the various flow components to provide for individual species can then also be incorporated. Despite these limitations, the environmental response curves are adequate to look at the relative importance of different aspects in determining allocation decisions for environmental water.
5 Optimising environmental releases in a single year

5.1. Introduction

The key purpose of producing environmental response curves and understanding the marginal benefit of flows to the environment is to use this information to better inform decisions about allocating environmental water. Chapter 4 discussed the approach to developing environmental response curves. This chapter (chapter 5) goes on to use environmental response curves in an optimisation model. The model aims to determine the optimal monthly release pattern for environmental flows, given an annual environmental allocation, over a single year. How can a set environmental water allocation best be used to minimise risk to the environment?

As previously discussed (Chapter 2 and 4), the creation of an environmental entitlement allows an environmental manager to actively decide about environmental water release. Using an optimisation model and environmental response curves also provides a transparent and consistent approach to making these decisions.

In this chapter, a single year optimisation model to determine monthly environmental release patterns for the Goulburn River is developed. A number of different scenarios are run (including variable annual allocations and climate conditions) to determine what factors most influence the optimal environmental release pattern. The model is also used to investigate the sensitivity of release decisions to the shape of environmental response curves and which flow elements primarily drive the decision process.

The chapter begins by discussing the model in the context of the economic ideas presented in earlier chapters (section 5.2). The chapter then presents a conceptual description of the model (section 5.3). The detail of the model in terms of algebraic structure is then provided (section 5.4). Section 5.5 describes the derivation of model inputs, including consumptive uses and environmental requirements. Section 5.6 presents the modelling results, with a discussion presented in section 5.7.
5.2. Economic context

Earlier chapters (and in particular chapter 2) provided background on environmental policy and the economic context of environmental water. Here, the optimisation model is described using these economic concepts. Recall Figure 2-6 and the discussion of Production Possibility Frontier and Society Indifference curves. These same concepts can be used to illustrate the purpose of the optimisation model. Figure 5-1 is an extension of the original Figure 2-6.

Initially, the optimisation model is used to determine an optimal release pattern for the current environmental allocation. The inherent hypothesis is that the current use of the environmental allocation is not optimal. Consider this in the context of Figure 5-1. The figure shows a range of attainable combinations of environmental outcomes and commercial outcomes. While the current use of environmental water sits with in the category of “attainable outcomes”, the optimisation model allows environmental outcomes to be improved without impacting on commercial (or irrigation) outcomes. In other words, the optimisation model moves us on to the production possibility frontier (move from point $A$ to point $B$). The process of optimising the environmental release has provided a gain of $ab$ (shown in Figure 5-1).

![Figure 5-1: Conceptual illustration of using the optimisation model to move onto the production possibility frontier.](image)
Recall that the point where society’s indifference curve (which defines society’s preferences) and the production possibility frontier are tangent defines how water should be allocated between the environment and commercial uses. This thesis does not attempt to determine this allocation. However, the optimisation model is used to determine the optimal release pattern for a range of environmental allocations. In other words, if society’s preferences change and the environment is allocated more water, what gain in environmental effectiveness is achieved (measured in this thesis by the term environmental risk). This concept is shown in Figure 5-2, where the change in allocation (based on the new indifference curve) results in an environmental improvement of $ab$.

![Figure 5-2: Conceptual illustration of using the optimisation model to show how environmental effectiveness changes as a result of shifts in the society indifference curves.](image)

### 5.3. Model Scope

A linear optimisation model was constructed, representing the key components of the Goulburn System relevant to the environmental flow reaches 1 and 4 (immediately downstream of Lake Eildon and immediately downstream of Goulburn Weir). The model objective is to minimise environmental risk in a single year by determining the optimal monthly release pattern for environmental allocations. The monthly environmental
release (which must total less than or equal to the annual environmental allocation) is determined by minimising the environmental risk across both reaches, given their different environmental requirements and flow conditions. The model is structured around a single year, allowing environmental releases to be made from Lake Eildon at each of the monthly time steps\textsuperscript{26}.

The model determines the optimal monthly release pattern by calculating the risk to each environmental flow component, at each relevant timestep, based on the passing flow. The passing flow is a combination of releases made for consumptive purposes (which are independent of the environmental manager’s decision), legislated releases (such as environmental flood water and compensation flows) and environmental allocation releases (the decision variable).

As discussed in Chapter 3.4, the Goulburn System is highly complicated and the existing water resource allocation model (the GSM) reflects this operational complexity. The aim of this research is to demonstrate an approach to allocating environmental water and as such does not require this level of complexity. Rather, the optimisation model is simplified to represent only the key components of the Goulburn System relevant to Reach 1 (downstream of Lake Eildon) and 2 (downstream of Goulburn Weir)\textsuperscript{27}.

Data from the GSM are used to represent the exogenous inputs to the optimisation model. The full record of data from the GSM has been used to create single year monthly series as input to the optimisation model (Figure 5-3). Note that the GSM is only used to obtain input data for the optimisation model; the two models do not interact. The following single year monthly series were derived from GSM data as exogenous inputs to the optimisation model:

- **Irrigation releases**: releases made to meet downstream irrigation demands.

\textsuperscript{26} Note that the model is based around a water year rather than a calendar year, running from July to June to ensure that each irrigation season is within a single water year.

\textsuperscript{27} Recall from section 3.5.4 that there are five environmental flow reaches. The study reach 1 is the equivalent of the environmental flow reach 1, while the study reach 2 is the equivalent of environmental flow reach 4.
- **Hydropower releases**: releases made to meet the requirements of the Pacific Hydro power station at the Lake Eildon Pondage (note that this water can be used by alternative uses downstream).

- **Spills and Flood Pre Releases**: storage spills and releases made from storage to prevent potential flooding from spills later in the season (based on target storage levels).

- **Compensation Flows (or minimum flows)**: releases made to meet legislated minimum passing flow requirements under the Bulk Entitlement conditions for the Goulburn River.

- **Environmental Floods**: releases made to meet legislated environmental flood events under the Bulk Entitlement conditions (note that these occur infrequently based on the storage level in Eildon and natural inflows in the preceding 24 months).

---

**Figure 5-3: The use of GSM in relation to the optimisation model**

As the model represents only a single year of operation (with monthly timesteps), it is assumed that changes in Lake Eildon’s operation will not be substantial enough to impact on spills and flood releases. Therefore, the storages have not been explicitly modelled in the single year optimisation model. Instead, as described above, the calculated passing flow from each storage (Lake Eildon and Goulburn Weir) is based on the outputs from the GSM. The passing flow from Lake Eildon includes irrigation releases, spills and flood releases, hydropower releases, legislated environmental flood releases, legislated minimum flows and legislated env. floods.
compensation releases and environmental allocation releases. The majority of irrigation water is diverted at Goulburn Weir, and thus passing flows at Goulburn Weir represent spills and environmental releases.

It is assumed that spills and flood releases will remain the same (as described by the GSM output) regardless of the environmental release decisions. However, the irrigation releases calculated from GSM will adjust as changes are made to the annual environmental allocation. Any increase made to the Environmental Allocation results in a corresponding reduction to the irrigation release as the increased environmental allocation must come from purchasing irrigation water. This is described in more detail in later sections.

The elements included in the optimisation model are shown in Figure 5-4. In this figure, “ENV RELEASE” represents the release decision for environmental allocation water. All other components of the passing flow are exogenous. This includes the legislated environmental flood (shown as env flood) and legislated minimum flow (min flow).

![Figure 5-4: Schematic of Model structure](image-url)
Environmental Response Curves are used to represent the environmental requirements. The development of environmental response curves was discussed in Chapter 4. Recall the response curve will change depending on a range of factors (discussed in section 4.3.3). In the optimisation model, it is assumed that changes in response curves will depend on the annual climate and the number of years since adequate environmental flow was last provided. Note that the environmental response curves represent the total benefit of providing water to the environment. The optimisation model calculates the marginal value (based on the gradient of the curves) in order to solve for the optimal release pattern.

5.4. Model formulation

The objective of the model is to minimise environmental risk in a single year by determining the optimal monthly release pattern. The monthly environmental release (which must total less than or equal to the annual environmental allocation) is determined by minimising the environmental risk across both reaches, for each flow component given their different environmental requirements and passing flow conditions. The passing flow is a combination of releases made for consumptive purposes (which are predetermined and independent of the environmental manager’s decision) and environmental releases (which is the decision variable). The model is structured around a single year, allowing environmental releases to be made from Lake Eildon at each of the monthly time steps. Environmental releases from Lake Eildon flow downstream and also pass Goulburn weir to provide benefit in both reaches. A water year, running from July to June, rather than a calendar year, is modelled to ensure that each irrigation season is within a single water year.

5.4.1. Objective Function

The objective function (to minimise total annual risk) is calculated as the sum of the risk caused to each flow component at each timestep. This implies that the risk is additive and assumes that the scale of risk is consistent across the flow components. It may be that an objective function related to specific species condition for a range of species (based on a number of contributing flow components for each species) is a better
representation of reality (as was used in the MFAT project, Young et al., 2003). It could be argued that complexity of interactions between flow components means that an additive objective function is too simple. However, these interactions are not currently understood and, therefore, the simple additive approach to risk has been adopted. This approach allows the role of each flow component to be readily understood. If warranted, for greater realism, constraints or changes to the objective function can be added at a later stage to develop interactions between components. Using a staged approach will make it clear which aspect of the objective function (or constraints) most impacts on the overall outcome.

5.4.2. Decision Variables

The only decision available to the environmental manager is the timing of the environmental releases of environmental entitlement from storage. Recall that the environmental flood and minimum (or compensation) flow are both legislated and therefore exogenous inputs to the model.

The purchase of capacity constraints (limiting the volume of irrigation water flowing down the river) could also be included as a decision variable with a cost per unit. However, in this model it is included as a model constraint and a number of scenarios with difference capacity restrictions have been run. This avoids additional complexity and uncertainty in calculations. Including capacity constraints as a decision variable would require constraints in terms of dollar values and budget to allow comparison between environmental releases and capacity constraint decisions.

A number of additional decision variables are included in the model as accounting or structural tools to allow the optimisation calculations to occur in linear form (explained in more detail in the following section).

5.4.3. Model structure

The objective function of the model minimizes the total annual environmental risk across both environmental reaches, for all flow components (given certain climatic conditions, initial storage level, number of years since each flow component was provided and
environmental allocation) (Equation 1). Binary inputs specify which environmental flow components are relevant to each reach and at each timestep.

Once the optimization has occurred, a secondary optimization function is required to ensure that the lowest volume of environmental water is used, maintaining the calculated total risk (i.e. the environmental benefit) (Equation 2). Rather than being linear, environmental response functions are non-linear and are represented in this analysis by a series of linear segments forming a piecewise linear function. As the environmental response functions are piecewise linear, the objective function and constraints have been written using binary variables to transform the problem into a linear form (Equations 3 and Equations 6 to 12). This allows the model to be solved using mixed integer programming.\(^\text{28}\) The passing flow is the sum of the environmental release and all other releases (including irrigation, spills and regulated flows) defined in Equations 4 and 5.

Constraints require that the total annual environmental release is less than or equal to the annual environmental allocation (Equation 13). If the annual environmental allocation is increased (for example, by purchasing additional entitlements) it must be accounted for by a decrease in the irrigation entitlement. The irrigation adjustment constraint modifies the volume of irrigation releases to account for any increase in the environmental allocation.

\(^\text{28}\) A good description of integer programming and piecewise linear functions can be found in Winston (2004, pp. 490 - 6)

A brief summary from Winston (2004) is provided here for context.

A piecewise linear function consists of several straight line segments. The points where the function changes gradient are called breakpoints. By using a set of binary variables, a piecewise linear function can be represented in linear form. Suppose that a piecewise linear function \(f(x)\) has breakpoints \(b_1, b_2, \ldots, b_n\). For some \(k\) \((k = 1, 2, \ldots, n-1)\), \(b_k \leq x \leq b_{k+1}\). Then, for some number \(z_k\) \((0 \leq z_k \leq 1)\), \(x\) may be written as

\[
x = z_kb_k + (1 - z_k) b_{k+1}
\]

Because \(f(x)\) is linear for \(b_k \leq x \leq b_{k+1}\), then

\[
f(x) = z_k f(b_k) + (1 - z_k) f(b_{k+1})
\]

Therefore a piecewise linear problem can be transformed into a linear optimisation problem, wherever \(f(x)\) occurs in the optimisation problem, by replacing \(f(x)\) by \(z_1 f(b_1) + z_2 f(b_2) + \ldots + z_n f(b_n)\). The following constraints are then added:

\[
\begin{align*}
Z_1 &\leq y_1, \\
Z_2 &\leq y_1 + y_2, \\
Z_3 &\leq y_2 + y_3, \\
\vdots &\leq \vdots, \\
Z_{n-1} &\leq y_{n-2} + y_{n-1}, \\
Z_n &\leq y_{n-1}
\end{align*}
\]

\[
y_1 + y_2 + \ldots + y_{n-1} = 1
\]

\[
z_1 + z_2 + \ldots + z_n = 1
\]

\[
x = z_1 b_1 + z_2 b_2 + \ldots + z_n b_n
\]

\[
y_i = 1 \text{ or } 0 \quad (i = 1, 2, \ldots, n-1)
\]

\[
z_i \geq 0 \quad (i = 1, 2, \ldots, n)
\]
allocation (Equation 14). The new irrigation release for a particular month is the old irrigation release minus a percentage reduction, based on the proportion of the annual irrigation occurring in that month.

While most environmental flow elements in the model are minimum requirements which must be achieved in all relevant months, the environmental response curve for wetland inundation must be structured to allow one event to occur each year (albeit in any one of the relevant months) rather than a continual flow across all relevant months. The model is constrained to consider individual risk contributed by the wetland response curve in only one of the applicable months and to automatically choose the month where the wetland constraint is best met. The wetland constraint adjusts the calculation of Equation 3 to ensure that an inundation event is only required in one of the three months (Equation 15 and 16). Equation 17 ensures that irrigation releases are less than the capacity restriction on flow downstream of Lake Eildon.

Formally the optimization problem is to choose $x_{phrst}$ to

$$\min\ totalrisk_{phrst} = \frac{1}{12} \sum_{t=1..12} \sum_{d=1..2} \sum_{q=1..9} w_d \times W_q \times u_{dq} \times c_{qt} \times m(x_{phrst} + f_{dph})$$

Equation 1

$$\min\ totalrelease = (\sum_{t=1..12} x_{phrst})$$

Equation 2

Where:

$$m(x_{phrst} + f_{dph}) = \sum_{i=1..n_{phq}} z_{iphqr} indrisk (b_{qphqr})$$

Equation 3

$$f_{1ph} = eirr_{ph} + spills + electricity + envflood + minflow$$

Equation 4

$$f_{2ph} = girr_{ph} + spills + electricity + envflood + minflow$$

Equation 5

Subject to:
Constraints for piecewise linear transformation to linear form

\[(x_{phrst} + f_{dph}) = \sum_{i=1..n_{qpr}} z_{iphqrt} b_{iphqrt} \quad \forall d, q, t\]  

\[\sum_{i=1..n_{qpr}} z_{iphqrt} \leq y_{iphqrt} \quad \forall q, t\]  

\[z_{iphqrt} \leq y_{iphqrt} + y_{(i-1)phqrt} \quad \forall q, t, i \in 2..n_{qpr} - 1\]  

\[z_{nphqrt} \leq y_{(n-1)phqrt} \quad \forall q, t\]  

\[\sum_{i=1..n_{qpr}} z_{iphqrt} = 1 \quad \forall q, t\]  

\[\sum_{i=1..n_{qpr} - 1} y_{iphqrt} = 1 \quad \forall q, t\]  

\[y_{iphqrt} \in \{0,1\} \quad \forall q, i, t\]  

Supply constraint

\[\sum_{t} x_{phrst} \leq s\]  

Irrigation adjustment constraint

\[eirr_{qph} = eirr_{qph} - \left(\frac{eirr_{qph}}{\sum_{T=1..12} \max(eirr_{qph}, girr_{qph})}\right) \times (s - \text{current}) \quad \forall t\]  

Wetland constraint

\[m(x_{phrst} + f_{dph}) \geq (\sum_{i=1..n_{qpr}} \text{indrisk} (b_{qph})) - 100 \times \text{pcount}_{14t}\]  

\[\sum_{t} \text{pcount}_{14t} = 1\]  

(where the wetland inundation component is \(q = 4\) and is relevant in reach \(d = 1\))
Capacity constraint

\[ eirr_{ph} \leq cap \]  \hspace{1cm} \textbf{Equation 17}

Where the model objectives are:

- \( totalrisk_{phr} \): average risk over a year for all reaches and flow components in relevant timesteps
- \( totalrelease \): total environmental release over a single year (sum of monthly releases)

And the decision variables are:

- \( x_{phrst} \): environmental release for the month \( t \), given the conditions year type \( (p) \), storage level \( (h) \), risk factor \( (r) \) and allocation \( (s) \).

And the calculation variables are\(^\text{29}\):

- \( y_{nphqrt} \): binary variable to convert piecewise linear to linear form
- \( z_{nphqrt} \): variable to convert piecewise linear to linear form
- \( pcount_{dqt} \): a binary variable to count when flow components have been provided.

And the model parameters are:

- \( b_{iphqr} \): the flow at breakpoint \( i \), for component \( q \) and conditions \( p \), \( h \) and \( r \)
- \( cap \): capacity limit for irrigation flow downstream of Lake Eildon (GL)
- \( c_{qt} \): a binary input which takes the value 1 if flow component \( q \) is relevant in month \( t \) and 0 otherwise
- \( current \): the current environmental allocation (75GL)

\(^{29}\) Calculation variables are separated from decisions variables as they refer to variables used to undertake calculations in the model, but that do not represent any real life decision.
\( e_{irr_{qph}} \) irrigation release from Lake Eildon for time \((t)\), year type \((p)\) and storage level \((h)\) (GL)

\( g_{irr_{qph}} \) irrigation release from Goulburn Weir for time \((t)\), year type \((p)\) and storage level \((h)\) (GL)

\( f_{d_{tph}} \) passing flow (based on irrigation releases, spills and legislated flows) at reach \((d)\) for time \((t)\) given conditions year type \((p)\), storage level \((h)\)

\( indrisk_{qphqr} \) the risk at breakpoint \((i)\), for component \((q)\) and conditions \((p),(h)\) and \((r)\)

\( n_{pqqr} \) the number of breakpoints for each flow component \((q)\), for conditions year type \((p)\), and risk factor \((r)\).

\( u_{dq} \) a binary representation of which flow components \((q)\) are relevant for each reach \((d)\).

\( w_d \) weighting of each reach \((d)\)

\( W_q \) weighting of each component \((q)\)

Where the indexed subscripts are

\( d \) environmental reach, \( d \in \{1 \ldots 2\} \)

(representing Reach 1 downstream of Lake Eildon and Reach 2 downstream of Goulburn Weir)

\( h \) initial storage level, \( h \in \{1 \ldots 3\} \)

( representing a low, medium and high storage levels)

\( p \) climate or year type, \( p \in \{1 \ldots 3\} \)

( representing a dry, average and wet year)

\( q \) environmental flow element, \( q \in \{1 \ldots 9\} \)

( representing the flow elements shown in Figure 4-9 and Figure 4-17)
\( r \) risk factor, \( r \in \{1\ldots3\} \)

(representing 0 years to 2 years since flow component was last provided)

\( s \) annual environmental allocation (in GL)

\( t \) time in months, \( t \in \{1 \ldots 12\} \)

The mathematical model formulation was translated into Mosel code so the software Xpress-MP could be used to run the optimisation (Dash Optimisation, 2007). Xpress-MP is a commercial software package developed by Dash Optimization. It was chosen because (1) it efficiently handles high volume of decision variables and (2) the user interface and coding in Mosel language is easy to use. Xpress-MP’s suite of optimization tools includes numerous optimizer algorithms. The simplex method was used to solve the linear optimisation problem\(^{30}\).

\(^{30}\) The simplex method (invented by George Dantzig) is the most commonly used method to solve linear optimisation problems. Descriptions of the simplex method can be found in the following texts: (2004) and (2007)

An brief extract form Chinneck (2007) is included here:

A corner point solution is any point where two or more constraints intersect. Two corner point solutions that are connected by a single line segment are adjacent corner-point solution. Here are three key properties of linear programs that drive the design of the simplex method:

1. The optimal point is always at a feasible corner-point.
2. If a corner-point feasible solution has an objective function that is better than or equal to all adjacent corner-point solutions, then it is optimal.
3. There are a finite number of cornerpoint feasible solutions.

...We now have enough information to provide a bird’s-eye view of the simplex method. It has two main phases:

1. Phase 1 (start up): find any corner-point feasible solution
2. Phase 2 (iterate): repeatedly move to a better adjacent corner-point feasible solution until no further better adjacent corner-point feasible solutions can be found. This final corner-point feasible solution defines the optimum point.
5.5. **Model Inputs**

As previously discussed, many of the inputs to the optimisation model are derived from the GSM (which was introduced in chapter 3). Details of the GSM model and the approach to deriving inputs can be found in Appendix A. The main model inputs are releases made from storage that can not be controlled by an environmental manager (Irrigation releases, spills etc) and the environmental response curves. A summary of the model inputs follows.

5.5.1. **Climatic year types and storage volume**

The environmental and irrigation demands are both influenced by climate and thus the model inputs need to vary depending on year type. The annual allocation varies depending on the predicted inflow and the existing storage available in the system. Therefore irrigation releases will vary depending on the initial storage in the system. The natural inflow series to Lake Eildon, along with the storage volume in Lake Eildon, were therefore used to define a series of different model inputs for three different year types (wet, dry and average) and three initial storage conditions (low, medium and high).

The natural inflow series to Lake Eildon (the arc EILDON INFLOW from the GSM model shown in Appendix A) was analysed to determine the 20th and 80th percentile annual inflows. Based on these flows, years were determined to be either wet (flows greater than the 80th percentile), average, or dry (flows less than the 20th percentile) year types.
Lake Eildon storage level (the EILDON ESTO from the GSM model shown in Appendix A) was assessed based on storage levels at the start of July and analysed to determine the 20th and 80th percentile storage levels. Based on these storage levels, years were determined to have either high (storage greater than the 80th percentile), medium, or low (storage less than the 20th percentile) storage levels at the start of the season.
The data available through the GSM is 114 years of record (based on irrigation seasons running from Jul – Jun) (refer to Appendix A for description of available data in the Goulburn River). This data is based on the current level of development and water usage in the basin. The number of years falling into each category for climate and storage level is shown in Table 5-1.

Table 5-1: Allocation of data to each year type or storage category

<table>
<thead>
<tr>
<th>Climate</th>
<th>Storage Level</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Low</td>
<td>Medium</td>
</tr>
<tr>
<td>Dry</td>
<td>5</td>
<td>16</td>
</tr>
<tr>
<td>Average</td>
<td>13</td>
<td>40</td>
</tr>
<tr>
<td>Wet</td>
<td>2</td>
<td>13</td>
</tr>
<tr>
<td>TOTAL</td>
<td>22</td>
<td>69</td>
</tr>
</tbody>
</table>

5.5.2. Passing Flows Reach 1 – downstream of Eildon

The passing flow at Lake Eildon includes irrigation releases, minimum compensation flows (legislated), environmental floods (legislated), hydropower releases (SECV), spills and pre flood releases. A twelve month series was derived from GSM model outputs for each of these releases from Lake Elidon. In each instance, a series was derived for each combination of year type and initial storage condition. The final series are shown in Figure 5-7 to Figure 5-9.

Irrigation releases form the predominant component of flows downstream of Lake Eildon. As expected, the irrigation demands occur predominantly over summer. This causes a seasonal inversion of flows downstream of Lake Eildon, with high summer flows and lower winter flows now occurring. In average and wet years, when the storage level in Lake Eildon is high, there are significant spills and flood pre releases over spring and winter months. Particularly in wet years (Figure 5-9c) this goes some way towards returning the seasonality to a natural condition.

There are two components of the environmental releases required under the Bulk Entitlement (BE) and these have been shown separately: environmental flood releases and compensation flow releases (described earlier in section 3.5.2). Both these release
requirements are constraints listed in the Goulburn Bulk Entitlement (Victorian Government, 1995). However, the conditions of these releases differ significantly and the releases have therefore been shown separately in the monthly release patterns. The compensation flows represent minimum passing flow requirements specified in the BE. The environmental flood represents large releases required under the BE once inflows to Lake Eildon exceed a trigger volume (a total of 80,000 ML is then released). As the approach to derive monthly patterns is based on monthly averages over a number of years of data, an environmental flood release is only considered to occur if the average flood release for a given series is greater than the minimum actual flood event (6961 ML). It can be seen in Figure 5-7 to Figure 5-9 that in keeping with the BE, environmental flood releases only occur in November.

Hydropower commission releases do not vary; they are constant releases over the months May to August.
Figure 5-7: Releases from Lake Eildon based on Irrigation, Compensation flow, hydropower releases and spills. Monthly release patterns for dry year (a) Low Storage (b) Medium Storage, (c) High Storage
Figure 5-8: Releases from Lake Eildon based on Irrigation, Compensation flow, hydropower releases and spills. Monthly release patterns for an average year (a) Low Storage (b) Medium Storage, (c) High Storage
Figure 5-9: Releases from Lake Eildon based on Irrigation, Compensation flow, hydropower releases and spills. Monthly release patterns for a wet year (a) Low Storage (b) Medium Storage, (c) High Storage
5.5.3. Passing Flows Reach 2 – downstream of Goulburn Weir

Most of the irrigation water released from Lake Eildon is diverted at Goulburn Weir and only a small volume continues further downstream. A volume of water spills at Goulburn Weir as upstream natural inflows are not all collected and diverted. This means that the flow downstream of Goulburn Weir more closely follows the natural monthly patterns (and seasonality) than the reach above Goulburn Weir.

A substantial portion of the compensation release from Lake Eildon continues past Goulburn Weir to meet minimum flow requirements downstream of Goulburn Weir and McCoys Bridge. Similarly, a portion of Environmental Flood releases made at Lake Eildon will spill at Goulburn Weir.

Figure 5-10 to Figure 5-12 show monthly passing flow for Reach 2 (downstream of Goulburn Weir) for dry, average and wet conditions.
Figure 5-10: Releases from Goulburn Weir based on Irrigation, Compensation flow, and spills. Monthly release patterns for a dry year (a) Low Storage (b) Medium Storage, (c) High Storage.
Figure 5-11: Releases from Goulburn Weir based on Irrigation, Compensation flow, and spills. Monthly release patterns for an average year (a) Low Storage (b) Medium Storage, (c) High Storage.
Figure 5-12: Releases from Goulburn Weir based on Irrigation, Compensation flow, and spills. Monthly release patterns for a wet year (a) Low Storage (b) Medium Storage, (c) High Storage.
5.5.4. **Environmental Response Curves**

The concept of environmental response curves was discussed in Chapter 4. Environmental Response curves were developed for reaches 1 and 2 in the Goulburn River (Section 4.4). The environmental response curves represent the total benefit of providing water to the environment rather than the marginal value.

**Relevant seasons and reaches**

The environmental response curves for each flow component (for each year type $p$ and risk factor $r$, where risk factor represents the number of years since flow was provided) are provided as model inputs. Binary values are then used to tell the model which flow components are relevant to each reach ($u_{dq}$ where $d$ represents the reach and $q$ represents the flow component) and the months in which the flow components apply ($c_{qt}$, where $q$ represents the flow component and $t$ represents the month). For example, shallow water habitat is relevant to Reach 1 and not to Reach 2. The binary representation for $u_{dq}$ is a “1” for Reach 1 and a “0” for Reach 2. The component is relevant only over the months January, February and March (indicated by a “1” in these months, and “0” in other months). The full list of binary values for $c_{qt}$ and $u_{dq}$ are shown in Table 5-2 and Table 5-3.

**Table 5-2: Relevant season of environmental flow elements ($c_{qt}$) (where 1 = July)**

<table>
<thead>
<tr>
<th>Flow Component</th>
<th>Month of year</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1  2  3  4  5 6  7  8  9  10 11 12</td>
</tr>
<tr>
<td>1. High water velocity</td>
<td>0  0  0  0  0  1  1  1  1  1  1  0</td>
</tr>
<tr>
<td>2. Riffle habitat</td>
<td>0  0  0  0  0  1  1  1  1  1  1  0</td>
</tr>
<tr>
<td>3. Shallow water habitat</td>
<td>0  0  0  0  0  1  1  1  0  0  0  0</td>
</tr>
<tr>
<td>4. Wetland inundation</td>
<td>1  1  1  1  0  0  0  0  0  0  0  0</td>
</tr>
<tr>
<td>5. Shear Stress (s/a)</td>
<td>0  0  0  0  0  1  1  1  1  1  1  0</td>
</tr>
<tr>
<td>6. Shear Stress (w/s)</td>
<td>1  1  1  1  1  0  0  0  0  0  0  1</td>
</tr>
<tr>
<td>7. Slow shallow habitat</td>
<td>0  0  0  0  0  1  1  1  0  0  0  0</td>
</tr>
<tr>
<td>8. Deep water habitat (s/a)</td>
<td>0  0  0  0  0  1  1  1  1  1  1  0</td>
</tr>
<tr>
<td>9. Deep water habitat (w/s)</td>
<td>1  1  1  1  1  0  0  0  0  0  0  1</td>
</tr>
</tbody>
</table>
Table 5-3: Environmental flow elements applicable to each reach ($u_{eq}$)

<table>
<thead>
<tr>
<th>Reach</th>
<th>Flow Component</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1</td>
</tr>
<tr>
<td>1 (d/s Eildon)</td>
<td>1</td>
</tr>
<tr>
<td>2 (d/s Goulburn Weir)</td>
<td>0</td>
</tr>
</tbody>
</table>

**Daily to Monthly Conversion**

The environmental response curves have been developed on a daily timestep. To reflect a range of ecologically important hydrological events, most environmental flows studies in Australia make recommendations on a daily scale. As the optimisation model is being run on a monthly timestep, the daily environmental response curves must be converted to a monthly timestep. Although perhaps not ideal to represent environmental requirements on such a coarse timestep, the monthly timestep is considered appropriate for a number of reasons. Firstly, flows in the Goulburn, especially over summer months when minimum recommendations are set, are highly regulated and can thus be managed operationally to meet requirements. Secondly, the regulation also means that daily flows have relatively low variability within months.

The problem of converting daily environmental flow requirements to a monthly timestep has been encountered in previous environmental flows studies as water allocation models are often constructed on a monthly timestep (for example, SKM, 2002). The complexities depend on the flow variability of the system and the types of flow components required. If catchment inflows are highly variable, a direct conversion of daily minimum flow requirement to monthly (multiplying by the number of days) is likely to under estimate the required volume of water. Similarly, if a fresh is required for only a small number of days in a month, it is difficult to ensure that flow is available for those days rather than spread across the month. Neal et al. (2005) document a detailed approach to converting daily recommendations to a monthly series. The method depends on a daily natural flow series and attempts to time environmental events to occur at times when water was naturally available. This approach is particularly suited to environmental flow recommendations that include an “or natural” clause. The difficulty
in applying this method when using environmental response curves is that the minimum flow volume is not a fixed target.

The environmental response curves predominately represent minimum or base flow requirements (with lower and upper limits). As mentioned previously, regulation in the Goulburn system means that daily flows are fairly consistent within each month. Direct conversion from daily to monthly flows is used as an adequate representation of monthly requirements (simply number of days multiplied by daily flow).

The exception to this is wetland inundation, which represents “events” rather than minimum flow requirements. For this situation, the standard method to convert daily response curves to monthly response curves no longer applies. These events need to occur in addition to regular instream flows, with the varying duration and magnitude of events taken into consideration. The daily annual flood frequency series was compared to the monthly flood frequency and used to make the transformation\(^{31}\). A relationship was developed as shown in Figure 5-13. There are two series shown; one shows a relationship between the peak daily and monthly events on an annual basis (for example, comparing the 1976 maximum daily event to the maximum monthly event of the same year), the other shows the daily peak event and monthly peak event for the same ARI. The latter series was used to develop the relationship. Thus, a 15 GL/d event is equivalent to a 204.5 GL/m event, and a 60 GL/d event is equivalent to a 710.1 GL/m event.

\(^{31}\) Note that these series were calculated based on the daily natural flow series downstream of Lake Eildon from 1975 – 2000.
5.6. Model application

The optimisation model was run for a number of scenarios to ascertain which factors most influence the environmental water release pattern from storage. The outcomes of these model runs are presented in the following sections.

5.6.1. Changing the climate and storage conditions

Recall from section 3.5.3, current proposals will entitle the environment of the Goulburn and neighbouring basins to 75 GL so the optimisation model was run with an annual environmental allocation of 75 GL (Food Bowl Modernisation Project Steering Committee, 2007). As this environmental allocation will be provided from water savings projects, (that is, pipelining existing open channels) it will not alter irrigation entitlements. Assuming that every flow component was provided in the previous year \( r = 1 \), the optimisation model was run initially for each combination of year type \( p = 1, 2, 3 \) and storage level \( h = 1, 2, 3 \). Before providing the results for all model runs,
detailed discussion of a single run will be used to explain the results of the optimisation model and their interpretation and explanation given the model structure and assumptions.

As an example, Figure 5-14 shows the optimal monthly release pattern for environmental allocations in an average climate year with a medium storage level. The top sector shows the monthly releases with exogenous releases (such as irrigation and spills) shown in grey and environmental releases shown in black. The figure shows that in an average year with medium storage, releases should occur in November, December, April and June. Below the monthly release pattern, each month’s environmental response curves (for the relevant reach) are shown. The dotted pink lines show the total passing flow and the corresponding risk for each flow element at each time step. To illustrate this, consider velocity requirements for Reach 1 the month of December in Figure 5-14. The total passing flow (including the environmental release and other releases) is 277 GL. Using the environmental response curve (and following the dotted pink lines), a flow of 277 GL results in a risk of 51.5% due to lack of appropriate velocity habitat. The total risk for both reaches, for all flow components and timesteps, is calculated as 13.59% (an unweighted average of all calculated risks to individual flow components).

The figure indicates that in Reach 1 the majority of environmental flow requirements are not met (i.e. risk is greater than 0%). No wetland inundation event has been provided (100% risk). High flows to meet irrigation requirements, leading to higher than natural flow conditions between December and March, are contributing to the total environmental risk (with velocity, riffle habitat and shallow water habitat requirements not fully met). The high flows are exogenous conditions and not possible for the environmental manager to adjust with the current decision limited to release decisions for the environmental allocation. The only flow component where the environmental manager could improve the outcome is wetland inundation, which would require a release of 120 GL/m in October to reduce the risk. This is greater than the current environmental allocation of 75 GL and, therefore, it is not possible for an environmental manager to affect the environmental outcome.
The environmental release decisions must therefore be targeted at improving conditions for Reach 2 (downstream of Goulburn Weir). All flow elements for all relevant months are provided in this reach (Figure 5-14). The environmental releases (November, April and June) would ensure that deep water habitat requirements are adequately met. Shear stress sets an upper limit on flow and, therefore, will not require additional releases from environmental water. Slow shallow habitat requirements are adequately met by existing releases from other water users.

This highlights that the management of environmental risk in the two reaches should be significantly different. The predominant risk for Reach 1 is excess flows during summer months, while Reach 2 requires additional flows to meet environmental needs.

The Total Environmental Risk calculated for each model run (varying year types and storage levels) is shown in Table 5-4. The total level of risk does not vary greatly with the various climate and storage level scenarios, most likely because high summer flows in Reach 1 cause high risk for some flow components. Release decisions for environmental allocations can not address this.
Figure 5-14: Optimal monthly release pattern for environmental entitlements and relevant flow components (Average year, Medium Storage): Downstream of Lake Eildon (Reach 1) and Downstream of Goulburn Weir (reach 4)
Table 5-4: Total Environmental Risk (%) based on optimal release pattern (all components provided in previous year, \( r = 1 \)) (75 GL)

<table>
<thead>
<tr>
<th>Year Type</th>
<th>Storage Level</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Low</td>
<td>Medium</td>
<td>High</td>
</tr>
<tr>
<td>Dry</td>
<td>13.33</td>
<td>13.84</td>
<td>14.84</td>
</tr>
<tr>
<td>Average</td>
<td>11.91</td>
<td>13.59</td>
<td>14.57</td>
</tr>
<tr>
<td>Wet</td>
<td>12.75</td>
<td>14.47</td>
<td>12.26</td>
</tr>
</tbody>
</table>

The monthly release patterns for each model run are shown in Figure 5-15 to Figure 5-17. Most releases are to meet deep water habitat requirements. The variation in release patterns depends on the variation in other storage releases. In average and wet years, with high storage levels, additional water is released in October and September respectively to meet wetland inundation requirements. This is only possible because other river flows are sufficiently high in November to obviate the requirement for an additional release to meet deep water habitat needs in that month. In wet years, when the storage level is high, a large release is made in January. This is counterintuitive because flows in Reach 1 are already high during summer months. Although additional release increases flow in Reach 1, it also increases the area of available riffle habitat (due to the shape of the environmental response curve and the original habitat rating curve) (refer to Figure 4-10c).
Figure 5-15: Optimal Monthly release pattern for environmental entitlements (Dry year) (a) release pattern (b) total passing flow downstream Lake Eildon (c) total passing flow downstream Goulburn Weir
Figure 5-16: Optimal Monthly release pattern for environmental entitlements (Average year) (a) release pattern (b) total passing flow downstream Lake Eildon (c) total passing flow downstream Goulburn Weir
Figure 5-17: Optimal Monthly release pattern for environmental entitlements (Wet year) (a) release pattern (b) total passing flow downstream Lake Eildon (c) total passing flow downstream Goulburn Weir

The model inputs for irrigation releases and spills for each year type and storage level are calculated by averaging a number of years of data from the GSM model (as described in section 5.5.1). Although the categories of year type and storage level have been used to try to isolate some of the variation in irrigation releases and spills, there will still be a range of flows within each category. The medium storage scenario is used as an example
as the bulk of data occurs in this category. The variation in releases for dry, average and wet years with medium storage levels are shown in Figure 5-18 and Figure 5-19.

Figure 5-18: Variation in releases (total of irrigation, spills, compensation and SECV) from Lake Eildon (Medium Storage) (a) Dry year (b) Average year (c) Wet year
Figure 5-19: Variation in releases (total of irrigation, spills, compensation and SECV) from Goulburn Weir (Medium Storage) (a) Dry year (b) Average year (c) Wet year

The optimisation model was run using the actual release data for each year of data available in GSM under each category for a medium storage; dry year (16 years of data), average year (40 years) and wet year (13 years). The modelling results show how the
variation in model inputs for other releases translates to variation in the optimal release pattern for the environmental entitlement. A summary of the results is shown in Figure 5-20.

Figure 5-20: Variation in optimal release pattern (varying model inputs for other storage releases) (a) Dry year (b) Average year (c) Wet year
Figure 5-21 shows how the risk for individual components varies with each of the runs for varying climate inputs in an average year with medium storage. The variation in risk comes mostly from flow components in Reach 1, over the summer months. There is also variation in risk from lack of deep water habitat provision in reach 4. If we refer back to Figure 5-19, the biggest variation in environmental release occurs in November and June, with no releases over December to March. There is no clear link between the variation in release pattern and the variation in risk for individual components. This is perhaps not surprising as the total release is driving the calculated environmental risk, not just environmental releases.

Figure 5-21: Variation in risk of individual components (varying model inputs for other storage releases) (Average year, Medium storage) (a) Velocity habitat (b) Riffle habitat (c) Shallow habitat (d) Deep water habitat
5.6.2. Changing the number of years since each flow component was provided

The greater the interval between provision of flow components, the greater the environmental risk. The effect of increasing the risk as the number of years \( (r = 1...3) \) increases is to change the gradient of the environmental response curves (as illustrated in Figure 4-8). The optimisation model was run, varying the number of years since each flow component was provided (changing \( r \)). This was applied to dry, average and wet years, but only for medium storage levels. Recall that the shape of the environmental response curves will adjust depending on the year type (wet, dry or average), whereas the storage level does not impact the shape of the response curves.

The optimisation model results show that as the number of years between flow provision increases, only a small number of flow components influence the environmental release decision. In dry years, the optimal environmental release pattern changes as the number of years since provision of winter deep water habitat increases (releases moving from April and May to October and July). In average years, the number of years since provision of riffle habitat also changes the release pattern. As the number of years since provision of riffle habitat increases, environmental releases in December reduce. This water will not provide additional benefit at other times of the year, and thus the total environmental release also reduces. Recall that the shape of the riffle habitat curve (Figure 4-10c) has a secondary depression when \( r = 1 \) (flow provided the previous year). When flow has not been provided for 3 years (\( r = 3 \)) this secondary depression is removed from the environmental response curve and there is no longer a benefit in releasing additional flows to improve riffle habitat provision. In wet years, both changes to deep water habitat (summer and winter) and riffle habitat result in changes in the release pattern. As the number of years since provision of riffle habitat increases, releases move away from January, February and March. Increasing the number of years since deep water habitat has been provided shifts flow to April (for summer requirements) and June (for winter requirements).
5.6.3. Changing the environmental allocation

Until this point, the model was run using an Annual Environmental Allocation of 75 GL. The environmental outcome achieved by varying the annual entitlement ($s = 0…500$ GL) is now assessed. This will require adjustments to the irrigation allocation, so the irrigation adjustment constraint (Equation 12) becomes important. As the environmental allocation increases, the derived 12 month irrigation series will be adjusted so that the total annual irrigation delivery is reduced by the same volume as the increase in environmental allocation. This volume will be distributed based on the percentage of irrigation water delivered in each month. For example, if the environmental allocation increases by 50 GL per year, the irrigation releases must reduce by 50 GL. In an average year with medium storage, the January irrigation release is 312 GL, which constitutes 24% of the total annual irrigation release (1306 GL). Therefore, the January irrigation release will be reduced from 312 GL to 300 GL, a reduction of 24% of 50GL. This is further shown in Figure 5-22.

![Figure 5-22: Example of reduction in irrigation supplied due to an increase in environmental allocation (average year type, medium storage level)]
A difficulty arises in differentiating between irrigation releases from Lake Eildon and Goulburn Weir. In some instances, releases from Lake Eildon continue over Goulburn Weir to meet downstream demands. On other occasions, inflows below Lake Eildon are used to meet downstream demands. This highlights the simplification in simply adjusting the irrigation releases up or down based on the environmental allocation. The operation of the system cannot be fully represented with such a simple approach. However, to demonstrate the likely impact on irrigation volumes, it can be assumed that the total annual irrigation volume is the sum of the maximum monthly volume downstream of Lake Eildon and Goulburn Weir. In other words, it is assumed that, where possible, demands downstream of Goulburn Weir are supplied from Lake Eildon.

Another assumption with this approach is that the crop mix will remain constant even though the irrigation allocation has reduced. In reality, as the allocation reduces, it is likely that lower value crops will exit the system and high value crops will play a larger role. As the crop mix changes, the pattern of water required may also change.

Figure 5-23 shows how the total environmental risk varies with changing environmental allocation for each year type and storage level (assuming every flow component was provided in the previous year $r=1$). The gradient of the curve represents the reduction in risk per additional GL of water provided to the environment.
Figure 5-23: Change in Total Environmental Risk as a result of changing environmental allocation
(a) Dry year (b) Average year (c) Wet year
As a general summary of the model results, for all model runs, the initial portion of the environmental allocation is released to minimise the risk by providing deep water habitat. As the annual allocation to the environment increases, and deep water habitat requirements are met, additional releases are made in December and January to provide for riffle habitat. As allocation increases further, the volume of water provided in December progressively increases until sufficient entitlement is available to begin influencing wetland inundation at which point the release pattern changes to provide high October releases. The middle portion of the total risk versus allocation curve (where deep water habitat has been provided but water is insufficient to provide wetland inundation) is usually flatter, indicating that in this range, less environmental benefit is achieved by each additional GL of water (Figure 5-23).

Risk posed by inadequate flow to meet velocity, riffle and shallow water habitat requirements is reduced purely through purchasing entitlements from consumptive users and thus reducing irrigation releases during summer months. However, significant allocations are required before the reduction in risk is substantial (in the order of 200 GL or greater).

In dry years, when no allocation is available, the risk to the environment is similar regardless of the storage level. However, the lower the storage level, the larger the reduction in environmental risk from the same allocation. When storage levels are at low or medium levels, the reduction in risk with each increase in allocation is less significant once deep water habitat is provided, yet there is not adequate water to provide wetland inundation (shown by a flattening of the curve in Figure 5-23a between 100 and 180GL in medium storage conditions and 150 and 250 GL in low storage conditions). The changes in gradient are more pronounced in low storage years. When the storage levels are high, there is a smooth transition between providing flows to meet deep water habitat and wetland inundation, and thus the gradient of the curve is more consistent.

In average years, deep water habitat is provided with an allocation of only 20GL in high storage years, 50GL in medium storage years and 120GL in low storage years. Above these allocations, the gradient of the total risk curve remains relatively flat (Figure 5-23b).
until flow is sufficient to begin wetland inundation. In high years, the risk to wetland inundation can be reduced with as little as 75GL allocation. In low and medium storage years, an allocation of 300GL is required to commence reducing risk from inadequate wetland inundation.

In wet years, when the storage level is high, deep water habitat is already provided most of the time, even with zero environmental allocation. The slope of the total risk curve is relatively flat, indicating that change to risk through providing additional riffle habitat and increasing wetland inundation is small. In low and medium storage years, an allocation of 100 GL and 50 GL (respectively) is required to meet deep water habitat requirements. Again, past this point, risk is changed minimally, even with large volumes of additional water.

### 5.6.4. Removing legislated compensation flow

The Bulk Entitlement requires that Goulburn Murray Water release a compensation flow to ensure legislated minimum flow requirements are met downstream of Lake Eildon, Goulburn Weir and McCoys Bridge.

The optimisation model was used to examine the environmental consequences of removing these minimum flow requirements. The components $e_{\text{comp}}$ and $g_{\text{comp}}$, representing releases to meet minimum flow requirements, were removed from the passing flow equation in the model (refer back to Figure 5-4) and this water was assumed to be held in storage. The model was then run for a range of annual environmental allocations. The results for medium storages in all year types are shown in Figure 5-24. The figure shows that in dry and wet years, for the same given allocation there is a higher environmental risk when the compensation flow is removed. Once the allocation reaches a certain level (200GL in wet years and 120GL in dry years) the environmental outcome is effectively the same, regardless of compensation flow releases. In some instances there is even a small reduction in environmental risk as lower flows pass Reach 1 during summer months. Figure 5-25 shows the environmental risk from lack of deep water habitat provision with zero (0) environmental allocation. The figure shows that removing the compensation flow increases the environmental risk from lack of deep water habitat.
provision. Once the allocation increases above a certain threshold (in a dry year 120GL),
the deep water habitat requirements can be met through the environmental allocation
rather than the minimum compensation. There are only minimal changes to the risk
posed by other flow components.

Figure 5-24: Change in Total Environmental Risk as a result of changing environmental allocation –
comparison of with and without minimum flow release based on BE rules (a) Dry year
(b) Average year (c) Wet year (all with medium storage levels)
Appendix A details the process for converting compensation flows into an environmental entitlement. As shown in Appendix A, there is a large variation in the volume of water released from storage to ensure legislated minimum flows are met. Recall from discussions in section 2.5.5, regulation minimum flow requirements have a higher reliability than entitlements. If this is converted to an entitlement, the entitlement would represent the volume that is available in 97% of years. The equivalent environmental entitlement (if compensation flows are removed) is calculated as 5.48 GL. This effectively means, if minimum flow rules were removed, an additional 5.48 GL could be allocated to the environment without impacting the reliability of supply for consumptive users. Based on the current environmental allocation of 75GL, there would be a very small increase in environmental risk during dry and wet years if the minimum passing flow rules were replaced by an environmental entitlement of 5.5GL.

The Bulk Entitlement also specifies an Environmental Flood be provided given a trigger inflow to Lake Eildon is reached. The optimisation model was run removing this environmental flood release. There was no change to the overall risk in either reach. Environmental flood releases occur in November (refer to Figure 5-7 to Figure 5-12) and are aimed at providing high flows for wetland inundation. However, the volumes
provided are not high enough to meet the full requirements and, therefore, have little impact on total risk.

5.6.5. Adjusting the weighting of each reach and each component

So far, the objective function places equal weighting on both reaches. However, the community may value reductions in environmental risk in one reach more than the other. This may be due to the existing environmental values in the river, access to the river at each location or activities in each reach.

The optimisation model was run for a range of weightings for each reach (varying $w_d$ in Equation 1). The results show that the monthly environmental release decision does not alter unless the value of one reach is different to the other reach by a factor of more than ten to one (Table 5-5). The release pattern changed when the Goulburn reach (Reach 2) is weighted down to between 0 and 0.1, the equivalent of one tenth the value of Reach 1. Reach 1 must be given zero (0) weighting to impact the release pattern. The calculated total risk obviously changes significantly depending on the weighting of Reach 1 (as all risk is contributed by flow elements relating to Reach 1).
Table 5-5: Environmental Release pattern with various weightings for each reach (Average year, Medium storage, 75 GL allocation, r=1). Indicates where changed weightings have caused a change in the release decision.

<table>
<thead>
<tr>
<th>Weighting</th>
<th>Environmental Release (GL/month)</th>
<th>Total Risk</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reach 1</td>
<td>Reach 4</td>
<td></td>
</tr>
<tr>
<td>Jul</td>
<td>Aug</td>
<td>Sep</td>
</tr>
<tr>
<td>0</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>0.05</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>0.1</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>0.2</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>0.4</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>0.5</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>0.6</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>0.7</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>0.8</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>1</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>0</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>0.05</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>0.1</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>0.2</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>0.4</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>0.5</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>0.6</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>0.7</td>
<td>0.0</td>
</tr>
<tr>
<td>1</td>
<td>0.8</td>
<td>0.0</td>
</tr>
</tbody>
</table>

The risk contributed by each flow component is added in the objective function to provide a total risk. As previously discussed, it is likely that the objective function is more complicated that this. It is possible that some flow components will impact on a number of aspects of river ecosystem (for example, velocity may be important for both macrophytes and small bodied fish) while others will only affect a single aspect of the ecosystem. This implies that some flow elements will have a larger impact on the overall ecological condition and should thus be weighted more heavily compared to other flow components.
The optimisation model was run with a range of different weightings for each flow component (varying $W_q$ in Equation 1). The environmental release pattern was only affected if the weightings for riffle habitat and shallow water habitat increased relative to the other flow components. The weighting on riffle habitat and shallow water habitat had to increase to 4 times the weighting of other components to affect the release pattern.

While this provides some insight as to how important the structure of the objective function and the weighting of different flow components are, it does not demonstrate how decisions would change if there were multiplicative effects between the different flow components.

Table 5-6: Environmental Release pattern with various weightings for each flow component
(Average year, Medium storage, 75 GL allocation, $r=1$) Note: weightings for all components not listed = 1.

<table>
<thead>
<tr>
<th>Weighting</th>
<th>Environmental Release (GL/month)</th>
<th>Total Risk</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Jun</td>
<td>Aug</td>
</tr>
<tr>
<td>1 1</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>4 1</td>
<td>10.1</td>
<td>0.0</td>
</tr>
<tr>
<td>8 1</td>
<td>0.1</td>
<td>0.0</td>
</tr>
<tr>
<td>1 4</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>1 8</td>
<td>0.0</td>
<td>0.0</td>
</tr>
</tbody>
</table>

5.6.6. Introducing capacity constraints

A significant proportion of the environmental risk is caused by excess flows in Reach 1 during the summer months. While increasing the environmental allocation causes some decrease in irrigation releases (and thus in summer flows) the environmental manager has limited ability to reduce these excess flows purely through managing environmental allocations.
Recent changes in water policy have led to the unbundling of water rights in the Goulburn catchment. Unbundling separates the traditional water entitlements into water shares, delivery shares and a water-use license (DSE, 2004). Delivery shares have the potential to be rationed or traded to help manage congestion or flow in certain sections of the delivery system (Productivity Commission, 2006). This would allow an environmental manager to purchase delivery shares, effectively setting a capacity constraint on flow down the river in a given month.

The optimisation model was run for a range of environmental allocations, with a range of different capacity constraints \( (cap = 0\ldots350 \text{ GL}) \) placed on irrigation releases downstream of Eildon (for an average year, medium storage). It should be noted that the capacity constraint was only applied to irrigation releases, and not to the total flow in the reach. It was assumed that the capacity constraint downstream of Eildon would not impact irrigation releases downstream of Goulburn Weir.

The results are shown in Figure 5-26. A capacity constraint of 250GL is required before environmental risk begins to reduce. This is the equivalent to a reduction of 100GL delivery to irrigators over the months December to February (Table 5-7). If the capacity constraint is set at 100GL, there is a significant drop in environmental risk (approximately 6%, or almost halving the risk). Figure 5-27 shows the risk for each individual environmental component relevant downstream of Lake Eildon. Velocity and shallow water habitat are most improved through the use of capacity constraints.
Figure 5-26: Change in total environmental risk with changing environmental allocation and capacity constraints (average year, medium storage).

Figure 5-27: Change in environmental risk with changing capacity constraints (75GL allocation, average year, medium storage) for Reach 1 environmental flow components (a) Velocity (b) Riffle Habitat (c) Shallow Habitat and (d) Wetland inundation.
Table 5-7: Volume of Irrigation water not available due to capacity constraint (75GL environmental allocation, average year, medium storage)

<table>
<thead>
<tr>
<th>Month</th>
<th>Capacity Constraint (GL)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>300</td>
</tr>
<tr>
<td>JUL</td>
<td>0.0</td>
</tr>
<tr>
<td>AUG</td>
<td>0.0</td>
</tr>
<tr>
<td>SEP</td>
<td>0.0</td>
</tr>
<tr>
<td>OCT</td>
<td>0.0</td>
</tr>
<tr>
<td>NOV</td>
<td>0.0</td>
</tr>
<tr>
<td>DEC</td>
<td>0.0</td>
</tr>
<tr>
<td>JAN</td>
<td>11.8</td>
</tr>
<tr>
<td>FEB</td>
<td>0.0</td>
</tr>
<tr>
<td>MAR</td>
<td>0.0</td>
</tr>
<tr>
<td>APR</td>
<td>0.0</td>
</tr>
<tr>
<td>MAY</td>
<td>0.0</td>
</tr>
<tr>
<td>JUN</td>
<td>0.0</td>
</tr>
<tr>
<td>TOTAL</td>
<td>11.8</td>
</tr>
</tbody>
</table>

There is currently no data available on the price of capacity shares. Ideally, the price impact on irrigators of setting a constraint would be assessed. This would be based on alternative options for sourcing water such as winter fill dams or construction of alternative delivery methods. The price of the capacity shares (and the corresponding environmental outcome) could be compared to the price of purchasing additional environmental entitlements (and the corresponding environmental outcome). The Goulburn Broken CMA commissioned a study looking at the structural limitations to delivering environmental flows. This included looking at the cost of alternative delivery channels and winter fill dams. The report concluded that these alternatives are too expensive to make capacity constraints a viable option (SKM, 2006b). However in other catchments, depending on the environmental gain, capacity constraints may provide a society welfare increasing outcome.
5.7. **Summary**

This type of linear optimal decision making model can be used to compare the relative significance of the different environmental water requirements on the release decision. In the case of the Goulburn River Basin, two flow components alone drove the decision making process: deep water habitat and wetland inundation. Riffle Habitat was important once deep water habitat had been provided and there was still not an adequate environmental allocation to meet wetland inundation requirements. This would indicate that future investigations of environmental water requirements should focus on better understanding these response curves.

The optimisation model can be run for a range of different environmental allocations to show the reduction in environmental risk per additional unit of water allocated to the environment. For the Goulburn River, the model runs showed the initial portion of the environmental allocation is released to minimise the risk by providing deep water habitat. As the environmental allocation increases, and deep water habitat requirements are met, additional releases are made in December and January to provide for riffle habitat. As allocation increases further, the volume of water provided in December progressively increases until sufficient entitlement is available to begin influencing wetland inundation, at which point the release pattern changes to provide high October releases. The middle portion of the total risk versus allocation curve (where deep water habitat has been provided but water is insufficient to provide wetland inundation) is usually flatter, indicating that in this range, less environmental benefit is achieved by each additional GL of water.

There are currently legislated minimum passing flow requirements downstream of Lake Eildon and Goulburn weir. The model was run with these legislated flows removed to assess their role in reducing environmental risk. Based on the current environmental allocation of 75GL, there would be a very small increase in environmental risk during dry and wet years if the minimum passing flow rules were replaced by an equivalent environmental entitlement of 5.5GL, but in an average year there would be little impact
on environmental risk if the minimum passing flow rules were removed. This increase in risk is due to the different reliability of legislated releases and entitlements.

Changing the weightings of the two reaches or the individual flow components does not impact the release decision unless the weightings are significantly different (4 times the relative weighting for individual components). This would suggest that a simple objective function can capture much of the information, noting that the risks are assumed additive and do not interact (that is, no multiplicative effects).

Reach 1 (downstream of Lake Eildon) carries high volumes of water in summer to meet irrigation requirements. Additional environmental entitlements will not address this issue. An optimization modelling approach can be used (as demonstrated) to investigate the role of capacity constraints. If large capacity constraints are required to improve environmental condition, the value of purchasing these capacity constraints can then be compared to the value of purchasing additional water.

Another important consideration is that the high flows in reach 1 make the total risk very insensitive to changes in environmental allocation. Purchasing environmental allocations has little impact on the overall risk for reach 1 as the issue is excess water. Therefore, for all scenarios, the model is still returning relatively high risks for reach 1. This perhaps limits the ability of the Goulburn River as a study basin to demonstrate the full potential of the model.
6 Optimising environmental releases over multiple years

6.1. Introduction

The single year model, described in Chapter 5, is used initially because the model structure is simpler to derive, making it easier to isolate the elements that drive decision making. Layers of additional realism and complexity can subsequently be added. In this chapter, the single year model is extended to a multi-year model, allowing decisions made in one year to influence environmental demands in later years. The multi-year model is used to ascertain whether decisions to release water in a particular year are influenced by predictions of conditions in future years. It is also used to investigate how decisions would be made about the volume of environmental water held in storage for use in subsequent years (adding a decision variable to the model).

The chapter begins with a description of the model scope and the key differences from the previous single year model (section 6.2). Detail of the model structure, including objective function, decision variables, constraints and inputs, are described in sections 6.3 and 6.4. The differences between the multi year model results and single year model results are analysed in section 6.5.1. The final section of the chapter discusses the modelling results when the environmental manager can decide to carryover environmental water to the following year (section 6.5.1).

6.2. Model Scope

The multi-year optimisation model uses the same concepts as the single year model described in Chapter 5. The model objective is to minimise environmental risk over a five year period (not just a single year) by determining the optimal monthly release pattern. The model inputs are similar to those described for the single year model.

32 Note that the model is based around a water year rather than a calendar year, running from July to June to ensure that each irrigation season is within a single water year.
Therefore, only the points of difference between the multi-year and single year model will be described in detail.

6.2.1. Defining climate, storage and antecedent conditions

In chapter 5, the single year model was run for a range of initial conditions that defined climate, initial storage volumes and the numbers of years since each environmental flow component had been provided. The multi-year model allows a sequence of years to be investigated in a single model run. While still defining initial conditions as inputs to the model, the optimisation model then runs a series of five year sequences that differ in terms of climate conditions (wet, dry and average) and storage volumes (low, medium, high). The environmental response curves adjust to reflect the provision of flow in previous years. Thus, the decision to release water in a given year will impact on future water requirements. A multi-year model reveals the effect of this on the decision making process. It also allows investigation of the additional decision to hold environmental water in storage (carry over water) for use in later years. While other factors could be investigated, the effect of climate and carry over storage are chosen as example of the effectiveness of this approach to decision making. These examples were chosen in particular as they seemed likely to have major influences on decisions.

In the single year model described in Chapter 5, each scenario specified the number of years since provision of an adequate flow component and this determined the shape of each flow component’s environmental response curve. In the multi-year model, the environmental response curve adjusts as an endogenous rather than exogenous variable at each timestep, according to flow provided in the previous timesteps. At each time step in each year, the model decides whether to release flow. Thus, each year’s decision influences the risk in future years and indicates the environmental response curve to be used in model calculations for future timesteps. This is demonstrated in Figure 6-1. The number of years since a given flow component has been provided (defined by \( r \)) changes the shape of the environmental response curve in the following year. In both Figure 6-1a and Figure 6-1b, the initial model input for the first year for a given component is \( r = 1 \). In other words, the flow component was provided in the previous year. In Figure 6-1a,
the environmental releases determined by the model meet the requirements of the flow component (by providing zero risk in the appropriate number of months) and therefore the model uses a binary variable (check) to feed this information to the following timestep. This insures that the environmental response curve remains the same (with \( r = 1 \)) for the following year. However, in Figure 6-1b the flow component is not adequately provided. Therefore, the model output for that year ensures that the environmental response curve in the following year changes to reflect that adequate flow has not been provided (\( r = 2 \)). This process is continued for every year of the model.

Figure 6-1: Schematic of how response curve for a given flow component changes in multi-year model (a) when flow component is provided in year one (b) when flow component is not provided in year one. Recall that \( r \) defines the number of years since adequate flow was provided and check is a binary variable used to feed this information to the following timestep.
6.2.2. Defining 100% risk

Note that while the model detects the risk of an individual component reaching 100% (in other words, that the probability of irreversible damage is high), the demand for this flow component is not removed in the following timesteps. This comes back to the problem of defining an end point. If 100% risk really does represent irreversible damage, there would be no environmental value in continuing protection once 100% had been reached. Therefore, it could be argued that the environmental response curve should be adjusted so that the relevant flow component is not provided for in future timesteps. As noted earlier the definition of 100% risk is not well understood, and therefore, the model simply tracks whether a “failure” has occurred and then continues to run with all environmental response curves included in the following timestep.

6.2.3. Defining environmental allocations

Recall that water users hold an entitlement to water and each year an allocation of water is available based on the volume of the entitlement held. The allocation is calculated based on potential inflows and actual volume available in storage. It is expressed as a percentage of entitlement. In other words, a 100% allocation means that the volume of water available in a given year is equal to the entitlement held.

The single year model was run with a fixed environmental allocation. However, the allocation varies each year for the same environmental entitlement. This is because allocations are calculated each year based on the water available in the system. Therefore, the multi-year model uses environmental entitlement rather than environmental allocation as a model input. The annual allocation (which will vary each year depending on predicted inflows and storage levels as described in section 3.5.3) is then calculated for each year, based on the given environmental entitlement.

6.2.4. Allowing carryover

The multi-year model allows a portion of environmental water to be held in storage for use in the following year. The model specifies a maximum volume that can remain in storage to the following season and allows this water to be carried over for only a single
year. Any spills from Lake Eildon are considered to come from this “carried over” volume of environmental water held in storage. For example, assume at the start of the irrigation year (July) the environment has 20GL of water held in storage, carried over from the previous season. If Lake Eildon spills 15GL in August, carried over environmental water account reduces to 5GL. The new allocation for the current year is not impacted by spills, this only applies to the volume of water carried over.

The model assumes that the operation of the environmental manager will not greatly affect storage volume and spills sequence\textsuperscript{33}. For this reason, the water balances of storages are not explicitly modelled to account for changed operation of the system. If the environmental allocation is used each year, it is unlikely that the timing of spills will be affected. However, the impact of carrying over water on storage volume and spills is also not modelled. It is assumed that if water is carried over, it is already allocated for environmental use and therefore is not considered when calculating available water resources for the following season. Thus, irrigation releases should not be affected and remain unchanged. Given the volumes of water under consideration (when compared to the total storage volume of Eildon and the volume of existing spills) and that carryover can only occur over a single season, this is a reasonable assumption for most circumstances. If environmental allocations increase, then modelling of storage may be required. The purpose of this model is to demonstrate the process of using an optimisation approach. If a model was constructed to provide the actual flow recommendations for the Goulburn River, this assumption would need detailed testing.

The model also does not consider additional storage charges incurred from carrying water over and holding in storage until the following season.

\textsuperscript{33} The full supply volume for Lake Eildon is 3,390 GL. Spills from Lake Eildon range from zero in a dry year, up to 950 GL in a wet year with high storage. These spills tend to occur between July and November. Carryover volumes being considered in the model range from 0 to 160 GL, with some environmental releases occurring in October and November.
6.2.5. Summary

Table 6-1 summarises the differences in key model inputs and assumptions between the single year and multi-year model. The obvious difference is the period modelled (single year versus multi-year), however other obligate changes ensue from this main difference.

Table 6-1: Summary of key differences and assumptions for single year and multi-year model

<table>
<thead>
<tr>
<th>Model Parameter or Assumption</th>
<th>Single Year Model</th>
<th>Multi-year Model</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of years</td>
<td>One</td>
<td>Five (optimisation occurs over five years, not for each year separately)</td>
</tr>
<tr>
<td>Environmental release timestep</td>
<td>Monthly</td>
<td>Monthly</td>
</tr>
<tr>
<td>Available environmental water</td>
<td>Model input is environmental allocation</td>
<td>Model input is environmental entitlement (allocation is then calculated for each of the five years)</td>
</tr>
<tr>
<td>Lake Eildon storage and spills</td>
<td>Not explicitly modelled (based on GSM data)</td>
<td>Not explicitly modelled (based on GSM data)</td>
</tr>
<tr>
<td>Changes to environmental response curve</td>
<td>As model input at start of single year</td>
<td>Modelling constraints used to adjust response curve in each year of the model</td>
</tr>
<tr>
<td>Carryover storage</td>
<td>Not modelled</td>
<td>Included as decision variable</td>
</tr>
<tr>
<td>Data input at access for decision making</td>
<td>Full knowledge of all climate data for every time step</td>
<td>Full knowledge of all climate data for every time step</td>
</tr>
</tbody>
</table>

It should be noted that the model is deterministic in terms of climate and streamflow inputs. That is, the model assumes full knowledge of climate, storage and other releases for all months in all years; the model assumes perfect knowledge. In reality, the EWM would not have access to all this information. A future extension would be to have only probabilistic information available at each stage of the decision timeline.

6.3. Model formulation

The model is an extension of the single year model described in chapter 5. The objective function now minimizes the total annual environmental risk (given a certain sequence of climate conditions, number of years since each flow component was provided in the initial year, environmental entitlement and allowable carryover of environmental allocation) across both environmental reaches, for all flow components over a five year
period (Equation 18). Binary variables specify which environmental flow components are relevant to each reach and at each timestep.

As in the single year model, rather than being linear, environmental response functions consist of a series of linear segments forming a piecewise linear function. As the environmental demand functions are piecewise linear, the objective function and constraints have been written using binary variables to transform the problem into a linear form (Equations 19 to 26). This allows the model to be solved using linear programming.

Constraints require that the total annual environmental release is less than the annual environmental allocation in any given year (Equation 27). This is complicated if carryover is allowed. A constraint is added to ensure that the carryover volume (that is, the environmental allocation not used in a specific year but held in storage until the following year) is less than the maximum allowable volume (Equation 28). The carryover component is the difference between the annual allocation and the actual volume of water released that year (assuming that water is carried over for only one year) (Equation 29). In accounting for carryover storage, storage spills are considered to come from the carryover entitlement. Therefore, a constraint is required to account for carryover volume on a monthly basis (Equations 30).

In the single year model, the annual environmental allocation was entered as a model input. The multi-year model uses the environmental entitlement as an input, and each year, this is converted to an allocation based on the storage level (as described in section 6.2.1). The annual allocation is first calculated as a percentage of the entitlement. This is then multiplied by the environmental entitlement to obtain the volumetric allocation (using Equations 31 and 32).

As described in the single year model, the current environmental allocation is assumed to be 75GL, gained through water savings projects. Increases in annual environmental allocation (for example, by purchasing additional entitlements) must be accounted for by decreased irrigation entitlement. The irrigation adjustment constraint adjusts the volume of irrigation releases to account for any increase in the environmental allocation (Equation 33). The new irrigation release for a particular month is the irrigation release
based on 75GL environmental allocation, minus a percentage reduction, based on the proportion of the annual irrigation occurring in that month.

While most environmental flow elements in the model are the minimum required in all relevant months, the environmental response curve for wetland inundation requires one event to occur each year\(^{34}\), albeit in any one of the relevant months. The model is constrained to consider individual risk contributed by the wetland response curve in only one of the applicable months and automatically chooses the month in which the wetland constraint is best met (in other words, the month when risk is smallest for the wetland component). The wetland constraint adjusts the calculation of Equation 19 to ensure that an inundation event is only required in one of the three months (Equation 34 and 35).

The environmental response curves adjust each year, according to the provision of flow components in the previous years (as previously discussed in section 6.2.1). The adequacy of provision in a particular year is defined by a series of constraints (Equations 36 to 45). A binary variable indicates whether flow was or was not provided in the previous year. Then, for every year, the risk curve for each component is determined using a series of constraints. These set the risk, expressed as a percentage, to be equal to at least the risk curve for the number of years since flow has been provided, minus the number of years flow has been provided (Equation 46).

Figure 6-2 provides further explanation of the constraints for adjusting the environmental response curve. For example, assume the environmental response curve for flow element \((q)\) is linear. The gradient of the curve increases with the number of years since flow was last provided (where \(r_1\) represents the curve when flow was provided in the previous year up to \(r_3\) representing the curve where flow was last provided two years ago). The risk for flow component \((q)\) at time \((t)\) must be at least equal to the risk calculated using \(r_1\). If indeed, flow was not provided in the previous year, then a binary variable \(b\) is set to zero. The second criteria is then added to also make \(r_{qt}\) at least equal to the risk calculated

\(^{34}\) Recall from Section 4.4.2, that a wetland inundation event occurs in every year (wet, dry and average), however the size of the event varies depending on the year type to reflect the recurrence interval of different sized events.
using \( r_2 \), thus ensuring that \( r_{qt} \) is defined by the curve \( r_2 \) rather than \( r_1 \). If, on the other hand, flow was provided in the previous year, this criteria is made redundant by the large negative value \( M^* \cdot b \). The process continues with each criteria, summing the number of years in which flow was provided in previous years.

\[ \sum \sum \sum \sum \sum = \sum \sum \sum \sum \sum = \sum \sum \sum \sum \sum = 5 \times 12 \times 1 = \sum \sum \sum \sum \sum \]

where:

\[ m(\chi_{\text{adj,ost}} + f_{\text{adv,p,o}}) = \sum z_{p,q,\text{q},q,\text{q},\text{q},\text{q},\text{q},\text{q}} \text{indrisk} (b_{p,q,\text{p},\text{p},\text{p},\text{p},\text{p},\text{p},\text{p}},) \]

Objective function

\[ \min \text{totalrisk}_{\text{adj,ost}} = \frac{1}{12} \sum \sum \sum \sum \sum = \frac{1}{12} \sum \sum \sum \sum \sum = \frac{1}{12} \sum \sum \sum \sum \sum = 5 \times 12 \times 1 = \sum \sum \sum \sum \sum \]

Equation 18

Where:

\[ m(\chi_{\text{adj,ost}} + f_{\text{adv,p,o}}) = \sum z_{p,q,\text{q},q,\text{q},\text{q},\text{q},\text{q},\text{q}} \text{indrisk} (b_{p,q,\text{p},\text{p},\text{p},\text{p},\text{p},\text{p},\text{p}},) \]

Equation 19

Figure 6-2: Example of constraint to ensure the environmental response curve changes with the number of years since flow was provided.
Subject to:

Constraints for piecewise linear transformation to linear form

\[(x_{ajgost} + f_{ajgost}) = \sum_{i=1}^{n_{pq}} z_{ipq} b_{ipq} \quad \forall d, q, t\]  \hspace{1cm} \text{Equation 20}

\[z_{ipq} \leq y_{ipq} \quad \forall q, t\]  \hspace{1cm} \text{Equation 21}

\[z_{ipq} \leq y_{ipq} + y_{(i-1)pq} \quad \forall q, t, i \in 2..n_{pq} - 1\]  \hspace{1cm} \text{Equation 22}

\[z_{pq} \leq y_{(n-1)pq} \quad \forall q, t\]  \hspace{1cm} \text{Equation 23}

\[\sum_{i=1}^{n_{pq}} z_{ipq} = 1 \quad \forall q, t\]  \hspace{1cm} \text{Equation 24}

\[\sum_{i=1}^{n_{pq}-1} y_{ipq} = 1 \quad \forall q, t\]  \hspace{1cm} \text{Equation 25}

\[y_{ipq} \in [0,1] \quad \forall q, i, t\]  \hspace{1cm} \text{Equation 26}

Supply constraints

\[\sum_{t=1}^{12} \max(x_{ajgost} - estorage_{at}, 0) \leq s\]  \hspace{1cm} \text{Equation 27}

\[carry_{at} \leq g\]  \hspace{1cm} \text{Equation 28}

\[carry_{at} \leq s - \sum_{t=1}^{12} \max(x_{ajgost} - estorage_{at}, 0)\]  \hspace{1cm} \text{Equation 29}

If \( t = 1 \), \( estorage_{at} = carry_{at-1} \),
else \( estorage_{at} = \max(0, estorage_{at-1} - spills_{ta} - x_{ajgost})\)  \hspace{1cm} \text{Equation 30}

Allocation constraints

\[current = 75 \times allocation_{aj}\]  \hspace{1cm} \text{Equation 31}

\[^{35}\text{Recall 75 GL is the current environmental allocation.}\]
\( sALLOC = s \times allocation_{aj} \) \hspace{1cm} \text{Equation 32}

Irrigation adjustment constraint

\[
eirr_{q_a,h_{aj}} = eirr_{q_a,h_{aj}} - \left( \sum_{t=1 \ldots 12} \max \left( eirr_{q_a,h_{aj}}, girr_{q_a,h_{aj}} \right) \right) \times (sALLOC - \text{current}) \quad \forall a, t
\]

\text{Equation 33}

Wetland constraints

\[
m(x_{af,ost} + f_{adp,h_{aj}}) \geq \left( \sum_{q=1 \ldots 14} z_{q_a,h_{a1q}} indrisk (b_{q_a,h_{a1q}}) \right) - 100 \times pcount_{a14t}
\]

\text{Equation 34}

\[
\sum_{t} pcount_{a14t} = 1
\]

\text{Equation 35}

Constraints to determine if component has been provided

\[
Indrisk_{ipad} = u_{ad} \times c_{q} \times m(x_{af,ost} + f_{adp,h_{aj}})
\]

\text{Equation 36}

For \( q = 1 \), If \( Indrisk_{ipad} = 0 \) \hspace{1cm} \forall t = 6 \ldots 11 \hspace{1cm} \text{then check}_{aq} = 1

\text{else check}_{aq} = 0 \hspace{1cm} \text{Equation 37}

For \( q = 2 \), If Count ( \( Indrisk_{ipad} = 0 \) ) > 4 \hspace{1cm} \forall t = 6 \ldots 11

\hspace{1cm} \text{then check}_{aq} = 1 \hspace{1cm} \text{else check}_{aq} = 0

\text{Equation 38}

For \( q = 3 \), If \( Indrisk_{ipad} = 0 \) \hspace{1cm} \forall t = 6 \ldots 8 \hspace{1cm} \text{then check}_{aq} = 1

\hspace{1cm} \text{else check}_{aq} = 0

\text{Equation 39}

For \( q = 4 \), If Min(\( Indrisk_{ipad} \)) = 0 \hspace{1cm} \forall t = 6 \ldots 9 \hspace{1cm} \text{then check}_{aq} = 1

\hspace{1cm} \text{else check}_{aq} = 0

\text{Equation 40}
For $q = 5$, if $\text{Count}(\text{Indrisk}_{aq} = 0) > 4 \ \forall t = 6\ldots11$,
then $\text{check}_{aq} = 1$, else $\text{check}_{aq} = 0$

Equation 41

For $q = 6$, if $\text{Count}(\text{Indrisk}_{aq} = 0) > 4 \ \forall t = 1\ldots5,12$,
then $\text{check}_{aq} = 1$, else $\text{check}_{aq} = 0$

Equation 42

For $q = 7$, if $\text{Count}(\text{Indrisk}_{aq} = 0) > 2 \ \forall t = 5\ldots7$,
then $\text{check}_{aq} = 1$, else $\text{check}_{aq} = 0$

Equation 43

For $q = 8$, if $\text{Count}(\text{Indrisk}_{aq} = 0) > 3 \ \forall t = 6\ldots11$
then $\text{check}_{aq} = 1$, else $\text{check}_{aq} = 0$

Equation 44

For $q = 9$, if $\text{Count}(\text{Indrisk}_{aq} = 0) > 3 \ \forall t = 1\ldots5,12$
then $\text{check}_{aq} = 1$, else $\text{check}_{aq} = 0$

Equation 45

Constraint to adjust the environmental response curves

$r_{aq} \geq m^v(x_{aqgost} + f_{adp,ja}) \cdot M \sum_{a-a,v,a-l} \text{check}_{aq} \ \forall v$

Equation 46

Where the decision variables are:

$x_{aqgost}$ environmental release for the month ($t$) in year ($a$), given a certain sequence of year types ($j$) defining climate and storage conditions, number of years since each flow component was provided in the initial year ($o$), environmental entitlement ($s$) and allowable carryover of environmental allocation ($g$).

$carry_a$ carryover volume for year $a$ (ie. volume held in storage for use the following year)
And the calculation variables are:

\[ y_{q,p,a,r} \] binary variable to convert piecewise linear to linear form

\[ z_{q,p,a,r} \] variable to convert piecewise linear to linear form

\[ pcount_{a,q} \] a binary variable to count when flow components have been provided.

And the model parameters are:

\[ b_{q,p,a,r} \] the flow at breakpoint \( i \), for component \( q \) and conditions \( p_{aq} \) and \( r_{aq} \)

\[ n_{p,a,r} \] the number of breakpoints for each flow component (\( q \)), for conditions in year \( a \) (\( p_{aq} \)) and risk factor (\( r_{aq} \)).

\[ indrisk_{q,p,a,r} \] the risk at breakpoint \( i \), for component \( q \) and conditions \( p_{aq} \) and \( r_{aq} \)

\[ allocation_{a,j} \] allocation (as percentage of entitlement) for year \( a \) in scenario \( j \)

\[ Check_{a,q} \] a binary check where 1 represents that a flow component (\( q \)) was provided in year \( a \) and 0 represents that the flow component was not provided.

\[ c_{a,t} \] a binary representation of flow components that are relevant in each month (a month is either relevant (1) or irrelevant (0) to a particular flow component).

\[ current \] the current environmental allocation (75GL)

\[ eirr_{t,p,h} \] irrigation release from Lake Eildon for time \( t \), year type \( p_{aq} \) and storage level \( h_{aq} \) (GL)

---

36 Calculation variables are separated from decisions variables as they refer to variables used to undertake calculations in the model, but that do not represent any real life decision.
$estorage_{ia}$  the volume of environmental allocation available in storage as part of carry over storage

$f_{adtph}$ passing flow (based on irrigation releases, spills and legislated minimum flows) at reach (d) for month (t) in year (a) given conditions year type ($p_{aj}$), storage level ($h_{aj}$)

$girr_{ph}$ irrigation release from Goulburn Weir for time $t$, year type $p_{aj}$ and storage level $h_{aj}$ (GL)

$h_{aj}$ storage in year ($a$) based on year sequence $j$

$p_{aj}$ weather / year type in year ($a$) based on year sequence $j$

$r_{aq}$ risk factor for each flow component ($q$) in a given year ($a$), $r_{aq} \in \{1...3\}$ (representing 0 years to 2 years since flow was last provided)

$u_{daq}$ a binary representation of which flow components are relevant for each compliance point.

$w_{d}$ weighting of each reach ($d$)

$W_{q}$ weighting of each component ($q$)

Where the indexed on subscripts are

$a$ time in years, $a \in \{1 ...5\}$

$d$ environmental reach, $d \in \{1 ...2\}$

(representing Reach 1 downstream of Lake Eildon and Reach 2 downstream of Goulburn Weir)

$i$ breakpoint number

$j$ defines sequence of year-type and storage level

200
0. initial risk scenario providing combinations of different years since each flow component was last provided at timestep $a = 1$ (i.e., defines the combination of $r_{aq}$) where $r_{aq}$ is the risk factor for component ($q$) in year ($a$).

$q$ environmental flow element, $q \in \{1 \ldots 9\}$
(representing the flow elements shown in Figure 4-9 to Figure 4-17)

$s$ annual environmental entitlement (in GL),

$t$ time in months, $t \in \{1 \ldots 12\}$

6.4. Model Inputs

Most model inputs are unchanged from the single year model (as described in Chapter 5). However, with a five year sequence used in the multi-year model, additional storage and climate information is required. The multi-year model also requires information on how to calculate allocations based on entitlement volumes. These added model inputs are discussed below.

6.4.1. Storage and climate data

The release pattern for irrigation and spills for each year type (dry, average and wet) and storage level (low, medium and high) remains the same as those inputs described in section 5.5. However, rather than a single year of data, the multi-year model requires a sequence of five years of data.

Model inputs were fabricated sequences of year types and storage volumes. The model was run for different combinations (or patterns) of year types and storage volumes. As the number of combinations of five year sequences is large, a selection was chosen to represent a range of relevant conditions. For example, one model run may have the year types (wet, wet, average, dry, dry) with the storage volumes (high, high, high, medium, medium). There are a large number of possible five year sequences and only a small selection of these combinations was modelled. The sequences were selected arbitrarily (they are shown in Appendix B).
6.4.2. Allocation volume

Model inputs included an environmental entitlement volume that is converted into an annual allocation for each year. The approach to determine allocations in the Goulburn basin was outlined in Chapter 3. Broadly speaking, the allocation is determined by available water in storage (across all storages in the system) and forecasted inflows for the next twelve to twenty four months.

Model inputs have been generalised into the three year types (wet, dry and average) and three storage levels (low, medium and high). This allows individual years within the multi-year model to be compared with the single year model results. Recall from chapter 5 that year types and storage levels are defined by the 20th and 80th percentile ranges using data from the GSM. Using GSM allocation data, a distribution of annual allocations (February allocation levels37) for each combination of year type and storage level was found (Figure 6-3). The figure shows that storage level plays a more significant role in determining allocation than the likely inflow. Storage levels can change allocation from 100% of entitlement (low storage) up to 190% entitlement (high storage) regardless of the climate. This is expected as the February allocation is calculated assuming the 99% exceedance probability inflow to storages. Based on the medium values, in years with high storage the allocation is taken to be 190%38. In medium storage years the allocation is taken to be 170% and in low storage years the allocation is taken to be 100%.

While there seems to be little variation in the allocation for high storage and low storage years, there is a large variation in the allocations for medium storage years. In part this is

37 February allocation levels are commonly used as the key allocation statistic (DSE, 2003)
38 Recall that a seasonal allocation is the volume of water available to a water user expressed as a percentage of their water entitlement. In the Goulburn system, allocations are expected to be at least 100% of entitlement in 97 out of 100 years. It is possible to have allocations that are greater than 100% of entitlements if storage. If the volume of water available is greater than 100% of water right and licence volume and losses, enough water is reserved to meet up to 100% of the next irrigation season's water right and licence volume and loses. The additional water is then made available in the current year.
merely a function of the number of data points used to calculate the distributions (refer to section 5.5.1).

Figure 6-3: Distribution of Annual Allocation for each year type (wet, dry, average) and storage volume (low, medium, high)

6.5. Model Application

The multi-year model is used to investigate two different questions. Firstly, the multi-year model results are compared with the single year model results to determine whether knowledge of future conditions influences release decision. Do our decisions improve with a multi-year model or is adequate information contained in the single year model? Secondly, the multi-year model is used to investigate an additional decision variable; allowing the environmental manager to decide to carryover water to the following year. This could not be investigated with the previous single year model as the value of carryover water in the following year could not be assessed.

The following sections discuss comparisons between the single year model and multi-year model outputs (section 6.5.1) and the influence of carryover decisions (section 6.5.2).
6.5.1. Comparison to single year model

Ensuring results are comparable

Changes or benefits obtained using multi-year model outputs were established by comparisons with equivalent single year model outputs. Recall that in the single year model (described in chapter 5), a range of environmental allocations were used as model inputs. In the multi-year model, an environmental entitlement is specified and this is converted to an allocation based on the year type (refer to section 6.4.2). As a number of years are modelled together, the entitlement must be used as the model input to ensure that the volume of water available to the EWM changes each year.

Recall from section 6.4.2 that allocation varies depending on storage level. Table 6-2 shows the allocation for the different storage levels and year types considered in the multi-year model runs. Note that dry year results are based on a low storage level, average year results on a medium storage level and wet year results on high storage levels, as only a finite number of model runs could be included in the analysis.

Table 6-2: Allocation for different modelling scenarios based on a 75GL entitlement

<table>
<thead>
<tr>
<th>Storage Level</th>
<th>Year Type</th>
<th>Entitlement (ML)</th>
<th>Allocation (%)</th>
<th>Allocation (ML/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low</td>
<td>Dry</td>
<td>75</td>
<td>100</td>
<td>75</td>
</tr>
<tr>
<td>Medium</td>
<td>Average</td>
<td>75</td>
<td>170</td>
<td>127.5</td>
</tr>
<tr>
<td>High</td>
<td>Wet</td>
<td>75</td>
<td>190</td>
<td>142.5</td>
</tr>
</tbody>
</table>

If direct comparisons with the single year model are to be made, the single year model run must match the allocation levels of 75, 127.5 and 142.5 ML/year used in the multi-year model. The single year model was rerun using allocations of 127.5GL (the average year allocation used in the multi-year model for a 75GL entitlement) and 142.5GL (the wet year allocation). Note that results for a 75GL allocation (the dry year allocation in the multi-year model) were obtained in the initial model runs shown in section 5.6.1.

The multi-year model generates large amounts of data and combinations of year types and storage volumes and only a small proportion could realistically be compared. In year
one, all flow components were assumed to have been provided in the previous year \((r = 1\) for all flow components in month \(t = 1\) and year \(a = 1\)). Model runs were then grouped according to the year type in the first year of modelling (dry, average, wet). The monthly release patterns for the first year can then be compared to the monthly release pattern established in the single year model run. This allows two separate questions to be addressed:

1. is the release pattern for a given year type the same, regardless of the climate condition in subsequent years?; and
2. is the release pattern for a given year type the same for the multi-year model and the single year model?

The results for dry, average and wet year types are discussed in the following sections. The dry year results are worked through in steps to explain the process used.

**Dry year results**

Recall that the first question to be addressed is whether the release pattern for a given year is affected by climate conditions in subsequent years. This question was initially addressed by analysing climate sequences where the first year of the five year sequence is dry. For example, the release pattern “DDDDA” represents four dry years followed by an average climate year. Only the release patterns in the first (dry year) of the five years were compared. Does the optimal release pattern in the initial year depend on climate in the subsequent years?

The multi-year model was run for 19 different climate sequences. Each sequence commences with a dry year and was followed by a different combination of dry, average and wet years. The monthly optimal release patterns for the initial dry year are shown in Figure 6-4. When climate sequences generated the same optimal release pattern in the initial year, they were grouped together in the figure.
Figure 6-4: Monthly release pattern for year one (dry year) followed by a variety of year types in subsequent years (D represents a dry year, A represents an average year and W represents a wet year).

Because Figure 6-4 shows a number of different release patterns for the initial dry year, it might suggest that the following climate sequence influences the release decision. The reasons for this variation require further consideration. The findings from the single year model results (chapter 5) would lead to the expectation that provision of deep water habitat may be a major influence on the release decision. Recall also that (from section 4.4.1),

- the response curve for deep water habitat will shift in the second year if adequate deep water habitat was not provided in the initial year; and
the requirement for deep water habitat is that adequate conditions must occur in three of the six winter spring months, and three of the six summer autumn months. To examine the influence of deep water habitat requirements on the release decision, the results from Figure 6-4 were broken into two sets of data: sequences where the release pattern in the initial dry year does not meet deep water habitat requirements and sequences where deep water is met in the initial dry year. In Figure 6-5 the corresponding risk posed by lack of deep water habitat is shown below the monthly release patterns. The following discussion demonstrates how the risk plots separate the release patterns into the two categories. The model runs shown in Figure 6-5a only achieve zero risk in two of the six winter months (refer to Figure 6-5c), despite minimising the overall risk. Therefore, deep water habitat is not provided and the environmental response curve for deep water habitat shifts for the following year (i.e. the risk factor $r = 2$ for year $a = 2$). In contrast, Figure 6-5b shows other runs where deep water habitat is provided in three of the six winter months (refer to Figure 6-5d), meeting the deep water habitat requirements. Consequently, the environmental response curve doesn’t shift (i.e. the risk factor $r = 1$ for year $a = 2$). A key difference between the runs shown in Figure 6-5a and those in Figure 6-5b is that releases in those runs in which deep water habitat is provided (Figure 6-5b) occur across four months rather than spread over five months. Zero risk is ensured in at least three months by concentrating flow releases over fewer months.

The provision or failure to provide deep water habitat explains some of the variation in release patterns observed in Figure 6-4. However, inspection of Figure 6-5a, shows that the release pattern of different runs vary, even though in all cases, deep water habitat has not been provided. Similarly, Figure 6-5b shows variation in release patterns even though in all cases deep water habitat is provided. This variation is not readily explained for the following reason. One would expect the release pattern in year one (dry year) to be the same regardless of the subsequent climate sequence, as long as the response curve for the following year is kept the same. This is because, in the model structure, the binary representation of “provided” or “not provided”, which informs the shape of the response curve, is the only aspect that influences decisions in following years.
Figure 6-5: Comparison of runs where deep water habitat is and is not provided where year one is a Dry year and followed by a variety of year types in subsequent years (D represents a dry year, A represents an average year and W represents a wet year) (a) Release pattern when deep water habitat was provided in the Dry year; (b) Release pattern when deep water habitat not provided in the dry year; (c) Risk caused by lack of deep water habitat for release patterns in (a); and (d) Risk caused by lack of deep water habitat for release patterns in (b). Note if multiple scenarios are listed together, all scenarios listed have the same monthly release pattern.
This unexpected variation is most likely explained by an artefact of the model output. In many instances, only a small percentage difference in the overall risk is generated by a number of different release patterns. The model processing time becomes extended while the model tries to differentiate between a number of very close solutions. Although in some instances, a solution will be found, it is not necessarily the optimal solution. It may be a solution that is considered within an acceptable error margin of the optimal solution. In this case, the release pattern identified by the model may differ from the optimal release pattern. However, in a practical sense, there is actually very little environmental difference between a release pattern resulting in a total risk of, for example, 18.751% or 18.752%. Therefore, one can assume that any one of the generated release patterns would provide a similar outcome. To demonstrate this, consider Figure 6-5b. A larger release in November with a smaller release in June (grey runs) results in adequate deep water habitat provision in November (risk of zero to this flow component) and a risk between 13.2898% and 14.0470% in June due to lack of deep water habitat. On the other hand, a larger release in June (blue runs) results in adequate provision of deep water habitat in June (zero risk), and a risk of 13.8728% in November due to lack of deep water habitat. Given that the objective function calculates the total risk over five years, twelve months and nine flow components, the difference to the objective function resulting from these two release patterns is negligible. It is also likely that these differences are within the error margin of model inputs.

To generalise, the model runs in Figure 6-5a and Figure 6-5b show that when a dry year is followed by a second dry year, it is important that winter deep water habitat is provided in the initial year (the runs in Figure 6-5a are all dry year followed by an average or wet year, and the runs in Figure 6-5b are mostly dry year followed by a second dry year). Lack of deep water habitat contributes a significant risk in some months. As the environmental releases cannot fulfil deep water habitat requirements in every winter month, a change in the environmental response curve will lead to an increased risk if a second dry year follows (as the gradient of the response curve will have steepened, ensuring a higher risk for the same flow level). However, if the dry year is followed by a wet or average year, the winter deep water habitat requirements can be adequately met in all winter months, re-establishing the response curve at the base case ($r = 1$). Therefore,
it is possible to minimise the risk in the dry year and not meet deep water habitat requirements as they can be adequately met in the subsequent year.

However, Figure 6-5 reveals exceptions to this rule: consider DAAAA and DAADD (shown in Figure 6-5b). In average years, reduction in total risk is due only to the shape of the riffle habitat curve (recall discussion in section 5.6.2) as deep water habitat is already provided by other passing flows and does not require an additional environmental release. If riffle habitat is not provided, it steepens to become a trigger flow (either provide adequate flow or risk is 100%). Once the environmental curve has reached a trigger shape, there is no longer scope to reduce risk. In comparison, the risk contributed by changing the response curve for deep water habitat is insignificant. For these model runs, the model was not able to calculate the optimal release pattern.

This highlights an issue with the structure of the model and the difficulty in isolating the optimal release pattern. The monthly release patterns for all runs where deep water habitat is not provided (all runs shown in Figure 6-5a) should be the same. The monthly release pattern for all model runs where deep water habitat is provided (all runs shown in Figure 6-5b) should also be the same. The differences shown in Figure 6-5 are due to calculation difficulties and marginal differences in the objective function calculated. Therefore, the optimal release patterns were found by manually calculating the minimum risk in year one for all model runs where winter deep water habitat was provided (or separately, not provided) and using the corresponding release patterns. Figure 6-6 shows the optimal release pattern for scenarios (with light grey squares representing the release pattern where deep water habitat is provided, and darker grey squares representing the release pattern where deep water habitat is not adequately provided). The vertical black lines show the variation (maximum and minimum) in modelling results across all model runs. For example, in November, the optimal release if deep water habitat is provided for the year is 13 GL/month, whereas the optimal release is 22 GL/month if deep water habitat is not provided in an adequate number of months of the year. Across all model runs, the suggested release ranged between 11 and 30 GL/month.
Figure 6-6: Optimal monthly release pattern for a dry year, low storage in year one (with deep water habitat provided and not provided). Variation in release patterns calculated for different runs is shown as black lines. Note if both dark and light squares are not visible, the optimal release is the same in that month.

From an operational perspective, the release pattern calculated for different model runs only varies in a few months. Therefore in most instances, it would be best to take a conservative approach and meet deep water habitat requirements. This would result in only a small increase risk, but it would ensure that regardless of the climate in the following year, the response curve will not change and thus there will be minimal risk in subsequent years. It also allows a smaller release to be made in November, early in the season so that large releases can occur at the end of the season once the final allocation for the year has been announced.

The model results, so far, have been used to assess whether the optimal release pattern varies depending on the climate in subsequent years. The second question is whether the
derived release patterns are different to the single year model results. While a minimum total risk is achieved in the single year model run, the model is not constrained to ensure that habitat is provided over three months in each season. Therefore, for the single year model run, deep water habitat is not adequate in that year. This means, the single year model is compared to the release pattern where deep water habitat is not provided, as derived by the multi-year model. Figure 6-7 shows that the release patterns are identical. Therefore the only advantage of the multi-year model has been determining a release pattern for conditions where deep water habitat requirements are met. Although the objective function has not changed in the multi-year model, additional constraints ensure that alterations in environmental response curves are based on antecedent conditions. Constraints could be added to the single year model and provide a similar outcome with a model that solves more efficiently.

![Figure 6-7: Comparison of single year model results (dry year) and multi-year model results for a dry year where deep water habitat is not provided.](image-url)
**Average and Wet year results**

The modelled release patterns for runs commencing with average and wet year conditions show similar outcomes. Once again, the multi-year model was run for a number of different climate sequences with average year results based on runs commencing with an average climate year followed by a variety of climate sequences, and wet year results based on model runs commencing with a wet climate year followed by a variety of climate sequences.

When runs begin with an average climate year, first year releases are aimed at providing riffle habitat, suitable velocity and shallow habitat. Deep water habitat is already adequately provided. The same flow components are provided in all model runs (with the same flow components not provided). In other words, in all model runs, the environmental response curves will be the same in the following year and are not affected by the decided release pattern. As the environmental response curves will be the same in the following year in all situations, the release patterns can be directly compared. The optimal release pattern for an average climate year (manually calculated), along with the variation between model runs in releases each month, is shown in Figure 6-8. The grey squares represent the optimal release, while the black lines show the range or variation in release pattern between the different multi-year model runs. The variation in release pattern is once again due to the number of solutions within close range to the optimal solution. The optimal solution was found taking the minimum risk over the year generated by the different release patterns. This results in the same release pattern as generated the release pattern for an average year generated using the single year model.
Figure 6-8: Optimal monthly release pattern for an average year, medium storage in year one (matching the single year model results). Variation in release patterns calculated for different runs is shown as black lines.

Similarly, the results for wet year conditions are shown in Figure 6-9. While there is still some minor variation in the model results, most run results match the single year model output for a wet year. The wet year runs tended to have a much faster modelling run time.

Figure 6-9: Optimal monthly release pattern for a wet year, high storage in year one (matching the single year model results). Variation in release patterns calculated for different runs is shown as black lines.
Summary

Both the single year and multi-year model show that in wet and average conditions, the calculated optimal release pattern is the same. This is regardless of the climate conditions in following years. In dry years, the release pattern depends on the following climate conditions. If the dry year is followed by another dry year, it is important that deep water habitat requirements are met (ensuring the response curve does not change). If the dry year is followed by a wet or an average year, the optimal release pattern is the same as that calculated using the single year model. If constraints are added to the single year model to show the number of months in which flow must occur (the same constraints added to the multi year model), the single year model provides the same information as the multi year model. There is no benefit, therefore, in constructing a multi year model (if carry over is not being investigated).

6.5.2. Allowing carry over storage

The multi-year model allows an additional decision variable: whether to hold environmental water in storage for use in the following year. The model was used to investigate the carry over volume from the first year to the second year using the climate sequences AAAAA, ADAAA, and AWAAA. In each case the corresponding storage level was used (low storage for a dry year, medium storage for an average year and high storage for a wet year). These sequences were chosen to demonstrate the process and isolate the effects of an average, dry and wet conditions. For each climate sequence, the model was run with different environmental entitlement volumes to determine how much water would be held over to the following year and the change in total risk over the five years by allowing carryover.

As previously discussed, deep water habitat is the flow component driving release decisions (with wetland inundation becoming important when large volumes of water are available). Therefore, any decision to carryover water will be based on the need to provide deep water habitat in the following year. The modelling results for AAAAA and AWAAA both showed that no water needs to be carried over from the first year to the second year. This is to be expected because, under average conditions with medium
storage, and wet conditions with high storage, deep water habitat can be provided with even low entitlements. There is no need for water held in storage from the previous year.

When an initial average year is followed by a dry year with low storage (ADAAA), allowing carryover of environmental water results in a lower overall risk. Deep water habitat requirements are not easily met in a dry year. Providing carryover water from the previous year improves the provision of deep water habitat and reduces the overall risk. Figure 6-10 shows the volume of water that should be released in the first year (average year) and the volume that should be carried-over to the second year (dry year) for various environmental allocations for year one. In other words, the figures are comparing how much should be used in year one compared to how much should be carried over to year two. Figure 6-10a and b show the same information in two different formats. Figure 6-10a shows the carryover and release volumes as a percentage of the environmental allocation, while Figure 6-10b shows the absolute values. The figures show that until environmental allocation reaches around 40GL, an EWM should choose to hold all of the entitlement in storage for use in the following dry year. For allocations above this, around 50% of the allocation should be released, with the remainder held in storage.

![Figure 6-10: Volume of carryover from first year (average climate, medium storage) to second year (dry climate, low storage) compared to environmental entitlement volume (climate sequence ADAAA).](image-url)
Figure 6-11 demonstrates that deep water habitat (shown in red) is the only flow component where the environmental allocation (and carryover) for year one (the average year) affects the level of risk. Although not shown here, the same can be said for the following dry year, where release decisions only have a big impact on the risk due to lack of deep water habitat provision.

Figure 6-11: Annual Risk for individual flow components in year one (average year).

As most flow components remain constant, the deep water habitat component from Figure 6-11 is highlighted for more detailed analyses in Figure 6-12. Figure 6-12 shows how the risk caused by lack of deep water habitat changes as the environmental allocation in year one changes. Figure 6-12a shows risk in year one while Figure 6-12b shows risk in the following year. Recall (referring back to Figure 6-10) for allocations up to around 40GL, almost all the environmental water is held in storage for the following year. Therefore, for environmental allocations in year one of up to around 40GL, there is minimal change in risk from winter-spring deepwater habitat provision in year one (Figure 6-12a). However, as the environmental water is carried over for use in the
following year, there is a steep decline in risk caused by winter-spring deep water habitat in the second year (dry year, shown in Figure 6-12b). For allocations in year one of greater than 40GL, some allocation is used in year one (average year) with the remainder carried over to the second year (dry year). This results in reduced risk in both years.

The curves shown in Figure 6-12 move (increasing or decreasing risk for a particular component) depending on the gradient of the environmental response curves. An example can be seen in Figure 6-12b for allocations of between 100 and 130GL. At a 100GL allocation, there is zero risk for summer-autumn and 5% risk for winter-spring. At 130GL, this switches, with close to zero risk for winter-spring and 3% risk for summer-autumn. The gradient of the environmental response curves (or the marginal value) makes it clear that there is more benefit in gaining the 5% decrease in winter spring for only a 3% rise in summer-autumn. Recall the response curves are piecewise linear.

Once allocations in year one reach 100GL, there is no risk from lack of deep water habitat in year one. With allocations of more than 160GL in year one (and a carryover of 75GL), there is no risk from lack of deep water habitat in year two. It is therefore expected that carryover volumes would not exceed 75GL even if the allocation continued to increase.
One would expect carryover storage to be used for the large wetland inundation events. However, in a dry year with low storage the highest monthly passing flow from July to October is 95 GL/month. To ensure zero risk by providing an inundation event, flows of 444 GL/month are required (given a wetland inundation event did not occur the previous year). This would require an environmental release of 349GL which is well above the volumes analysed. Wetland inundation may play a larger roll influencing carryover decisions if passing flows from irrigation and other releases were more significant.

The results shown are based on the pairing of average year conditions and medium storage, dry year conditions and low storage and wet year conditions and high storage. Storage levels play the main role in determining allocation levels for a given year. It is therefore likely that storage levels, along with climate, are influencing the carryover
decision. Obviously if storage levels are low, the risk of spilling is also significantly reduced. If we look at the decision to carryover water with the same climate sequence (ADAAA), but make the storage level high in the dry year (instead of low), only small volumes of water are carried over (in the order of 3 GL) for allocations up to 50GL (past which the optimal decision is not to carry over water). If the storage is high in the dry year (and thus allocations are increased), passing flows are significant enough to provide a large portion of deep water habitat requirements, thus negating the need for carryover.

6.6. Summary

If carry over is not being assessed (and is not a decision option) the single year model can provide the required information. This would require constraints to be added to the single year model to ensure flow components are provided in dry years. This may also apply to other year types if the allocation volume changed. A series of different single year model runs with varying constraints could be assessed if a new river system was being modelled. The advantage of using the single year model is the shorter processing time and ease of isolating the optimal solution.

The multi-year model becomes important if holding environmental water in storage is a decision available to management. The model demonstrates how the carryover volume will change depending on climate and entitlement volumes. Only a small number of scenarios were investigated to demonstrate the process.

While the single year model showed that the variation in stream flow in a given year included in each year type is likely to generate a range of different release patterns (section 5.6.1), this analysis was not repeated using the multi-year model since the above results demonstrate that the outcome should be similar and could be modelled by adjusting constraints in the single year model.
7 Discussion

7.1. Introduction

The Goulburn River has been used as a case study to illustrate how an optimisation approach could be used to make release decisions for environmental water. The approach could be generalised to any river system. Figure 7-1 shows the steps in implementing an optimisation approach to making release decisions for environmental water. The exact method will depend on the data available for the specific river being assessed. In the Goulburn system, for example, the first seven steps (data collection, reach selection, field assessment, identification of assets, objectives, identification of indicators of ecological response and hydraulic modelling) were already provided by existing environmental flows studies. The environmental objectives and type of indicators chosen will then inform the method used to develop environmental response curves, ideally with advice of an expert panel to inform the shape of the response curves. In this project, for the Goulburn River, habitat was used as an indicator of environmental condition and informed the development of the environmental response curves. The objective function for the optimisation model should relate back to the environmental objectives and assets identified in the initial steps.

The approach shown in Figure 7-1 suggests the creation of a lumped optimisation model, which involves developing the model structure, identifying decision variables and using the model to identify decision points and key parameters. This lumped model is a simplified representation of the system, with only key elements represented. The outputs from the lumped model can then be reviewed by the expert panel to check that the model constraints and environmental response curves are causing the model to produce acceptable outcomes. Once the model has been validated by the expert panel, a detailed optimisation model representing the complete system (either deterministic or stochastic) can be produced to identify a set of decision rules for the environmental water manager. The environmental response curves and model constraints should be updated when ever new information becomes available.
This research has focussed on the steps of developing environmental response curves, defining an objective function and creating a lumped optimisation model to identify the drivers and decision points (using the Goulburn River as a case study). In essence, these steps address the first two research questions; How can environmental water requirements be quantified to indicate the marginal value of water to the environment? and What factors influence the decision to release environmental water in a given month?.
The process of developing environmental response curves was demonstrated using the Goulburn River as a case study. The response curves were developed with reference to the environmental flows study already completed for the study area. There is limited information or knowledge of how the ecosystem responds to flow variation, or the resilience of different aspects of the ecosystem. The environmental response curves for the Goulburn River were developed based on the information available at the time. A number of assumptions and extrapolations were made. These response curves would require review as monitoring programs reveal more information to inform the shape of the response curves. Some of the key assumptions in developing environmental response curves and potential future work are discussed in section 7.2.

The case study demonstrates how an optimisation model can be used to determine the relevance of various inputs to the decision to release environmental water. The modelling results demonstrated that, for the Goulburn River, one flow component (deep water habitat) drives the decision to release environmental water from storage at a given time. The modelling also showed that specific climate and storage data have a large impact on the volume of water an EWM would choose to release from storage. The multi year model only provided additional information to the single year model if carryover was introduced as a decision variable. The multi year model was not always able to isolate the optimal release pattern, however this demonstrates that there are a number of release patterns that yield similar environmental risk. While the optimisation model provides a useful tool in understanding factors that influence the decision to release water, this transparency also requires perhaps a larger amount of structured information as input to the decision making process.

The final research question addresses the transparency and flexibility of the decision making approach. It effectively asks, does this optimisation approach actually provide advantages over existing decision processes for environmental water? The overall research aim was to develop a conceptual framework for flexible and transparent prioritising of environmental water releases. Table 7-1 highlights some limitations of existing approaches and explains how the optimisation approach addresses these issues.
The key change is presenting environmental requirements in a way that allows the marginal value to the environment to determine release decisions.

Table 7-1: Issues to overcome in current approaches and improvements from Optimisation Approach

<table>
<thead>
<tr>
<th>Issues to overcome (based on current approaches)</th>
<th>Solution in Optimisation Approach</th>
</tr>
</thead>
<tbody>
<tr>
<td>Binary approach of setting minimum requirements</td>
<td>Environmental needs are represented by environmental response curves that relate environmental outcome to flow, showing the marginal value of water to the environment.</td>
</tr>
<tr>
<td>Transparency of decisions (trade off decisions on how to use a limited supply of water are often subjective and not well documented)</td>
<td>Decisions on how to release environmental water are based on the marginal value of water to each flow component or river reach. While the shape of the response curves will require expert judgement, they are clearly presented. The model then allows assessment of the key variables or inputs driving each decision.</td>
</tr>
<tr>
<td>Adaptability (often environmental flow recommendations are set comparing natural to some current condition. Difficult to then assess impact of changed regime)</td>
<td>Once the environmental response curves and model are structured, any flow scenario can be assessed.</td>
</tr>
<tr>
<td>Potential to update with new knowledge (requires reconvening of expert panel)</td>
<td>The environmental response curves can be updated as new information becomes available. This will also require reconvening of the expert panel.</td>
</tr>
<tr>
<td>Operational decisions (based on long term statistics making it difficult to determine rules for operational decisions)</td>
<td>The model constructed here shows how decisions should change with time and such circumstances as climate. Once the model for a given river has been validated by an expert panel, it can be changed to a stochastic model and used to generate operation level decisions.</td>
</tr>
</tbody>
</table>

### 7.2. Limitations of study

While the optimisation approach provides advantages, there are areas of the method that require further exploration. The case study of the Goulburn River aimed to establish whether the optimisation approach was feasible, but it also highlighted a number of issues requiring further development. These are enunciated in detail in the following section, however will be summarised here.

Developing the environmental response curves requires knowledge of the flow limits that the ecosystem can withstand. The response curves developed in this study were based on the data available at the time. They do not include a robust definition of end points.
(100% risk) or the spatial and temporal timing of environmental damage. The response curves were developed for the key flow components highlighted in the environmental flow study. However, with more data available, it may have been more transparent to link the environmental response curves to species types. This would also allow dependence on multiple flow components to be introduced to the optimisation model. One of the most important simplifications of the existing optimisation model is that it does not allow for interaction between flow components. For example, fish species may require riffle or deep pool habitat as an adult, floodplain inundation for breeding and marginal vegetation areas for juvenile habitat. Ideally, the model would reflect this using the objective function or constraints to show that a certain sequencing of flows or combination is required.

The Goulburn catchment was used as a study basin to demonstrate an optimisation approach and the sort of information in can provide or questions it can address. However, the reach below Lake Eildon suffers from excess summer flows. This contributed a significant portion of the overall environmental risk. The model was then less sensitive to the various management “decisions” available. Another study basin would perhaps have better demonstrated the optimisation approach.

Models are often misunderstood or mistrusted. Models serve a number of purposes. First, they can be used to predict outcomes for a specific case or scenario—such as the consequences of a particular activity by an Environmental Water Manager. However, to be used for this purpose, the model must be tested or calibrated to show that it is does indeed reflect the situation it purports to model. The model produced in this research is a demonstration model to show a process or method. It has not been tested to the extent that it can be used to infer practical decisions in the Goulburn Basin. However, models serve a second purpose of identifying the inputs that will have the greatest impact. This then highlights the regions that require further research to improve the quality of the inputs and thus improve the reliability of decisions. While the information to populate the model may not currently be available, it helps demonstrate which pieces of information are critical to move the process forward. In this context, a model is a very important research tool as it highlights the information required, and the quality of that
information, to be able to make decisions. In this way, models are invaluable as a pure research tool. They can more identify the critical factors that influence the environment, thus prioritizing research, without the need to undertake costly interactive studies to discover by trail error which factors matter most.

Ecologists, amongst others, struggle with the idea of trading off flow components against each other. This is seen very much as an economic approach to environmental management. It would be disappointing if a model such as this optimisation model was seen as a tool for commercial exploitation, rather than as a mechanism for understanding the optimal use of a society’s finite resource. While numerous flow components are important for the health of the environment, ultimately, the health of a river and its use are a compromise. A model sets up a framework where expert knowledge can be presented in a logical structure to understand these tradeoffs. All environmental flow projects must decide which components of the flow regime the environment can cope without. A modelling structure helps to isolate the steps and logic in these decisions. The difficulty is, while each constraint or logic step in the model can be communicated to stakeholders, the final flow series generated by the optimisation model may be more difficult to describe. This is especially the case while the definition of “total risk” is left vague in terms of temporal and spatial impact.

In almost all biological systems, measurement on key indices has proved the most effective tool in management. It is difficult to believe that the river ecology is necessarily more complex than the ecology of the human body – perhaps less well studied. However there are now numerous examples, where for example, renal or cardiac function can be reduced to just a few key measures. Furthermore, measuring these accurately has improved outcomes. That these systems could be so succinctly measured and managed would have seemed a forlorn hope fifty years ago. Managing rivers may well require us to begin the steps of isolating key functions and focussing research on measuring and describing these functions accurately. While the model in this thesis does not purport to do this, it has demonstrated a method that allows this sort of understanding to be developed.
The model is currently a deterministic model. This has suited the aim of demonstrating the optimisation approach. However, in practice, an environmental manager would not be aware of the future climate conditions when making an individual release decision. Therefore, the modelling needs to be develop to determine decisions without the knowledge of future events. This would make it more useful as a management tool.

The following section discusses further work that could improve the current optimisation model.

7.3. Further work

This research has developed a base model to demonstrate that an optimisation approach is feasible, and show the sorts of information required and output possible from the modelling process. There is a range of further work that could be done to extend and refine the optimisation model. Some of the steps are detailed below and used to illustrate and discuss some of the modelling assumptions used in this thesis.

1. Further develop environmental response curves, improving the definition of end points (100% risk) in terms of spatial and temporal environmental damage.

Environmental response curves are integral to an optimisation approach to make release decision for environmental water. As discussed in section 4, environmental response curves reveal the marginal value of providing water to each flow element. Environmental response curves have key components that will assist in making allocation decisions:

- A comparable measure of environmental outcome: in this research, environmental risk was used as a measure of environmental outcome.
- Environmental response is dynamic: environmental water requirements change according to flows provided in previous years and the resilience of the system.

In the Goulburn case study, the environmental response curves were based on the shape of the habitat rating curve and natural provision of habitat. The method adopted to develop environmental response curves will vary depending on the data available for the river under consideration. However, there is a set of information or knowledge that must
be available for an optimisation approach, regardless of the factors used to build the environmental response curves. This set of information includes:

- identification of end points (100% risk);
- identification of zero risk boundary;
- interpolation between these two extremes (here assumed to be linear);
- resilience of the ecological attribute (how many years can it last without water);
- and
- recovery (how many years of adequate flow are required for regeneration after a stress period) of the ecological attribute.

The definition of environmental risk requires further development. The definition of 100% risk implies that irreversible damage will occur but what actually constitutes “irreversible damage” is imprecise. Both spatial and temporal scale of the damage affect the extent of regeneration or recovery. If only a small section of the river is damaged, regeneration from neighbouring reaches may be possible. The temporal scale of the impact is also important. Does the flow result in instant damage (such as high flows uprooting vegetation) or is there a long gradual process? The optimisation model results show that environmental risk caused by lack of appropriate velocity and riffle habitat is high, in some instances, reaching 100%. These limits were set according to knowledge of the ecology in the reach. However, we know that under current conditions, flows would reach levels posing high levels of environmental risk, and yet, there are still environmental values present in the reach.

Similar discussions are also relevant to definition of zero risk. For the Goulburn River case study, as is the case with many environmental flows studies, we adopted the natural flow paradigm, in which it is implicit that natural conditions are favourable and larger deviations from natural pose higher risks. However, Reach 1 (downstream of Lake Eildon) has been affected by changed seasonality and the impacts of regulation since the construction of Lake Eildon in 1929 (G-MW, 2008). It is likely that any environmental values still present in the reach have adapted to these new conditions. Therefore, unless the objective is to return the reach to natural conditions (which would have significant impact on irrigation), the natural reference point is not necessary appropriate for Reach 1.
The shape of the response curves should develop to accommodate new knowledge as it becomes available. The results from the Goulburn optimisation model indicate that when the environmental response curves changed, reflecting the years since flow was provided, the release decision was only affected by the shape of the deep water habitat response curve and the riffle habitat response curve (refer to section 5.6.2). Obviously this will be catchment specific depending on the flow components relevant in a given catchment. However, the optimisation model allows assessment of the sensitivity of decisions to the shape of the environmental response curve. This in turn informs where effort should be placed in refining the environmental response curves.

It is important that an expert panel be involved in both the initial development of the environmental response curves, but also reviewing the initial optimisation model results. The riffle habitat environmental response curve for the Goulburn River case study provides a clear example. While conceptually, it makes sense that the gradient of the environmental response curve changes based on the change in area of available riffle habitat, the riffle habitat response curve has a “kink” where additional riffle habitat becomes available at higher flows (refer to Figure 4-10). The modelling results show that this leads to additional environmental releases in summer months despite passing flows already exceeding natural conditions. It may be important that the riffle habitat is located in the main channel of the river (rather than in side areas as would be the case with high flows). An expert panel would be best placed to review the model outputs and environmental response curve to ensure that these additional summer releases are in fact improving environmental outcomes.

2. **Review the objective function (or constraints) to incorporate interaction between flow components and penalties for allowing high risk to individual flow components.**

The objective function for the Goulburn River case study (in both the single and multi-year models) was set as the average risk for all flow components, across both reaches and all relevant time steps. Unless extreme changes were made to the weightings of the reaches and flow components, there was no impact on the decision of when to release water. This implies that, for this particular river, a simple objective function produces the
same result. This may not be the outcome for other river systems. There are a number of additional considerations that were not included in the objective function:

- **Interaction between flow components** – if two flow components are both aimed at benefiting the same element of the ecosystem (for example, fish) it may be that there is no benefit in providing one flow component without the other (for example, why trigger spawning if there is nowhere for juvenile fish to survive in the following months). In principle, these interactions can be accounted for in either the structure of the objective function or using constraints.

- **Extreme risk** – the current objective function does not penalise for high risk levels in one particular component. By averaging all the flow components, 100% risk to component $A$ with 0% to component $B$ is considered the same as 50% risk to each component $A$ and $B$. Again a more sophisticated functional form of the objective function could allow for non-linearities.

3. **Develop the definition of total risk.**

The two optimisation models constructed for the Goulburn River (the single year model described in section 5 and the multi-year model described in section 6) demonstrate how an optimisation approach could assist an EWM in deciding when and how much to release from storage. The main output from the optimisation model is the monthly environmental release pattern. The “total risk” as defined by the objective function is predominantly used to isolate the optimal release pattern. However, as became apparent in the multi-year modelling (refer to section 6.5.1), the difference is very small between the total risk calculated for a number of different release patterns. This extended the multi-year model run times and in some cases made the optimal solution difficult to isolate. As total risk is difficult to define, it is difficult to know what constitutes a significant difference in overall risk. Even when allocation was increased from 0 to 500GL, the risk in a dry year only reduced from 18 to 4% risk. Without a clear definition of total risk (and the value the community places on a reduction in risk), it is difficult to know how substantial a reduction this is. This is again an issue with the objective function and averaging results across a large number of flow components and timesteps.
4. **Link the optimisation model to the resource allocation model for the catchment**

Climate and storage levels were seen to have a big influence on the environmental release decision. Rather than using average passing flow patterns, it would be more appropriate to use actual data for each year. Ideally this would involve linking the optimisation process to the allocation model.

In the multi-year model, one of the assumptions was that spills from storage would not be affected by release and carryover decisions by the EWM. Linking the optimisation model to a resource allocation model, where storage volumes and spills are modelled in detail, would ensure spills are correctly represented.

5. **Use the model to explore optimal combination of regulated minimum flows and entitlements**

Section 2 provided an overview of environmental flow policy in Australia. The shift in policy from a regulatory approach to provision of environmental entitlements was discussed. In many catchments, a combination of regulated minimum flows, caps on consumptive users and entitlements, act concurrently to provide environmental water.

The Goulburn River has legislated compensation flow requirements and environmental flood releases, along with the proposed (through the food bowl modernization project) 75 GL entitlement. Section 5.6.4 shows the environmental outcome if the legislated compensation flow is removed. The results show that in dry and wet conditions, for the same given entitlement, there is a higher environmental risk when the compensation flow is removed. Once the allocation reaches a certain level (200 GL in wet years and 120 GL in dry years) the environmental outcome is effectively the same, regardless of compensation flow releases.

The optimisation model allows investigation of the combination of environmental entitlements and legislated flows. Creating an additional decision variable to set the optimal minimum legislated flow, allows an optimization model to demonstrate the effectiveness of both approaches and how they should be used together.
6. Expand the model to include decisions on how and when to increase the environmental entitlement.

A clear advantage of the policy change to create environmental entitlements is the ability for an EWM to enter the water market and purchase water on behalf of the environment (as discussed in chapter 2). The plots of total risk versus environmental allocation (shown in section 5.6.3) show how understanding the marginal value of water to the environment can assist by informing decisions on environmental allocation. The plots show the additional gain (or reduction in risk) for each additional unit of allocation. An EWM should choose to purchase water in the market at volumes where the marginal value is high (in other words, the EWM would not purchase water on the flat part of the curve where an addition unit of water has no additional benefit).

Without a clear definition of total risk, or a dollar valuation on total risk, it is not possible to determine the ideal allocation of water between the environment and consumptive users. However, the optimization model could be expanded to include decisions about purchasing water in the market. An additional budget constraint could be added, with decision variables expanded to include permanent and temporary trades and option purchases and calls at each timestep (along the lines of the modeling undertaken by Hollinshead (2006)).

7. Extend the model from a base model to a set of operational decisions for an environmental manager.

The model developed for the Goulburn River system is deterministic. That is, from the first timestep, the decision maker (or model) is aware of the flows and climate for all future timesteps. In the case of an EWM, they would have to make release decisions without the foresight of future flows and climate data. There are a number of different approaches that can be used:

- A deterministic model can be constructed and run for a range of different scenarios. The corresponding release decisions can be combined to form a distribution.
A stochastic model can be constructed, where each decision is made based on a distribution of possible future flows and climate. The optimal release decision will be made at each timestep based on the likely future events.

With either approach, it is important that the deterministic model is constructed as a first step so that the model outputs can be reviewed by an expert panel to ensure that the model is correctly representing the environmental requirements.

8. **Once a detailed model is developed, use existing environmental flows monitoring programs to verify the model outcomes and update model inputs as required.**

One of the difficulties encountered in determining release patterns for environmental flows using a modelling approach was that of verifying or calibrating the model. This issue has been widely discussed in monitoring of environmental flows (for example, Gippel, 2001). Reference data or base data for ecological condition is not always available. Ecological responses may not be instantaneous, and are often delayed. Many external factors also affect instream health (for example, cattle access, and nutrients inflow) making it is difficult to isolate the impact of flow. The *Victorian environmental flows monitoring and assessment program* (Chee et al., 2006) proposed the uses of conceptual models and Bayesian statistics to improve the understanding of ecosystem response.

While it is not possible to immediately verify the optimisation model results, the model outputs can be assessed against long term environmental flows monitoring data (once it is available) and the information from these monitoring programs used in the review process to improve model inputs.
7.4. Summary

This section has summarised how an optimisation model can be constructed to address environmental release decisions and the range of policy questions that the model can incorporate. However, there are still a number of areas that require further research. These include:

1. Further developing environmental response curves, improving the definition of end points (100% risk) in terms of spatial and temporal environmental damage.

2. Reviewing the objective function (or constraints) to incorporate interaction between flow components and penalties for allowing high risk to individual flow components.

3. Developing the definition of total risk.

4. Linking the optimisation model to the resource allocation model for the catchment.

5. Using the model to explore optimal combination of regulated minimum flows and entitlements.

6. Expanding the model to include decisions on how and when to increase the environmental entitlement.

7. Extending the model from a base model to a set of operational decisions for an environmental manager.

8. Once a detailed model is developed, using existing environmental flows monitoring programs to verify the model outcomes and update model inputs as required.
8 Conclusion

The objective of this thesis was to develop a conceptual framework for prioritising environmental water releases in a flexible and transparent manner.

This has been achieved by answering three research questions:

1. How can environmental water requirements be quantified to indicate the marginal value of water to the environment?

2. What factors influence the decision to release environmental water in a given month?

3. Does an optimisation approach provide the flexibility and transparency to make release decisions?

The marginal value of water to the environment is an important input to making informed trade-off decisions about environmental water. Environmental response curves provide a relationship between environmental outcome and flow. While this does not provide information on the value the community places on a particular environmental outcome, it provides enough information to allow trade-off decisions between different flow elements required for the environment. The method used to develop environmental response curves will vary depending on the information available for each river. A method was demonstrated using the Goulburn River, based on habitat rating curves and habitat provision under natural conditions. There are still a number of uncertainties and knowledge gaps in our understanding of the ecological response to flow. The shape of environmental response curves will need to be revised as new information becomes available.

The Goulburn River case study illustrated how an optimisation model can be used to isolate the parameters which have an important role driving the release decision. In the case of the Goulburn River, the majority of release decisions were based around one flow component (deep water habitat). This may be different for other river systems. However, more generally, an optimisation model has the advantage that it provides a mechanism for
key parameters to be isolated. Climate and storage volumes (both of which impact on passing flows) also had a significant impact on release decisions.

Optimisation modelling, and the development of environmental response curves and modelling constraints, provides a transparent means of decision making for environmental water. As the model inputs can all be updated, the process is flexible and adapts to new information as it becomes available. However, complex optimisation models can be difficult to solve. The modelling process should start simple and identify key parameters to limit modelling issues. The structure of the objective function and constraints will also impact on the ease of interpretation of results.
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GLOSSARY

**Annual allocation**  Volume of water allocated to a water entitlement in a given season/year.

**Consumptive Use**  The application of water to a use that typically diverts water from its natural flow and permanently withdraws at least some of the water from the water source.

**Delivery Share Capacity**  A share of an irrigation supply channel capacity (in a regulated system) or a watercourse capacity (in an unregulated system), specified as a percentage share or a volumetric supply rate at a particular time.

**Economically efficient**  An activity is economically efficient if there is no other use where the resources would yield a higher value or net benefit.

**Entitlement**  The volume of water authorised to be used under a licence to take and use water.

**Environmental Demand Curve**  A function that relates flow to environmental outcome, in terms of dollar value.

**Environmental Flow Component or Element**  An element of the flow regime that provides a specific ecological trigger or distinct habitat provision (for example, high flow, low flow, overbank flow).

**Environmental Flow Recommendations**  The flow regime recommended by expert panels under the current State Government method of determining environmental flows.

**Environmental Water Manager (EWM)**  An agency with overall managerial responsibility for achievement of environmental objectives (currently the Catchment Management Authorities in Victoria)

**Environmental Response Curve**  A function that relates flow to environmental outcome, where environmental outcome is described by a scale of 0 to 100% environmental risk.
**Environmental Risk**  Likelihood of adverse environmental risk based on the natural provision of habitat (where 0% represents no risk and 100% risk represents likely irreversible damage).

**Environmental Water Reserve (EWR)**  The formal recognition of the environment as a water user in Victoria. The EWR comprises of caps, conditions on consumptive users, entitlements and water shares.

**Externality**  Occurs when a side effect of a decision by an individual (or business) affects another party’s wellbeing, but that effect is not taken into appropriate account by the decision maker.

**Extraction**  The withdrawal of water from surface water and groundwater sources.

**Market mechanism**  Instrument that encourages behaviour through market price signals rather than through explicit directives.

**Opportunity cost**  The fordone benefits from the next best alternative use of a resource.

**Optimal Environmental Flow Pattern**  Environmental flow pattern provided once the marginal value to the environment is considered (based on a given allocation). This may or may not be equal to the Environmental Flow Recommendations.

**Option contract**  A contract that gives the right, but not the obligation, to purchase or sell a good at a specified price within a specified period of time.

**Over-allocation**  The allocation of water rights to consumptive users is above a sustainable level.

**Passing flow**  The total flow past a given point including water released for consumptive users, spills, natural flows and other releases.

**Permanent trade**  The trade in water entitlements (or share) on a permanent basis.

**Regulated river or stream**  Rivers or streams with flow controlled through the use of weirs, locks and dams.
**Return flow** There is a difference between the amount of water extracted for consumptive use (the gross water usage) and the amount of water actually used in crop production (the net water usage). The return flow refers to the volume of water that re-enters the river system through seepage (often the difference between gross and net water usage).

**Storage capacity share** A market-compatible water sharing system that defines storage access in terms of a share of a reservoir capacity, and inflows and outflows (which include deductions for evaporation and seepage losses).

**Temporary trade** The trade of seasonal allocations (or dividends), with the entitlement remaining with the original owner.

**Unbundling** Historic water entitlements bundled water, land, water use, delivery and works approvals. Unbundling refers to the separation of historic water entitlements into separate entitlements.

**Water-use efficiency** The physical relationship between water required and output (including system losses, irrigation techniques etc).
Appendix A  Goulburn Simulation Model

The Goulburn Simulation Model (GSM) represents the operating systems for the Goulburn, Campaspe and Loddon River Basins. This appendix includes a detailed outline of the model components relevant to this research. It then describes the derivation of the optimisation model inputs (for this research) based on the GSM data.

A.1. Data Availability

Table A-1 provides an assessment of the data availability for the Goulburn catchment. There is a large volume of data available through the GSM. This data is on a monthly timestep and available over 114 years of record.

Table A-1:  Data availability for the Goulburn Catchment

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Source</th>
<th>Scale</th>
<th>Start Date</th>
<th>End date</th>
<th>Continuity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temporary traded volume (exchange)</td>
<td>Water</td>
<td>Zonal</td>
<td>1998</td>
<td>Current</td>
<td>Discontinuous, Weekly</td>
</tr>
<tr>
<td>All (successful and unsuccessful) sell and buy bids (exchange)</td>
<td>Water</td>
<td>Zonal</td>
<td>1998</td>
<td>Current</td>
<td>Discontinuous, Weekly</td>
</tr>
<tr>
<td>Temporary and Permanent Traded Volume (All trades)</td>
<td>GMW</td>
<td>Districts</td>
<td>1994</td>
<td>Current</td>
<td>Annual</td>
</tr>
<tr>
<td>Irrigation supply allocation changes announcements</td>
<td>GMW</td>
<td>Systems</td>
<td>1993</td>
<td>Current</td>
<td>Monthly</td>
</tr>
<tr>
<td>Irrigation deliveries</td>
<td>GMW</td>
<td>Systems</td>
<td>1994</td>
<td>Current</td>
<td>Weekly</td>
</tr>
<tr>
<td>Rainfall and Evaporation</td>
<td>GSM</td>
<td>GSM storages</td>
<td></td>
<td></td>
<td>Monthly</td>
</tr>
<tr>
<td>Irrigation and river flows</td>
<td>GSM</td>
<td>GSM links</td>
<td></td>
<td></td>
<td>Monthly</td>
</tr>
<tr>
<td>Crop areas</td>
<td>ABS</td>
<td>Districts</td>
<td>1993</td>
<td>1997</td>
<td>Discontinuous, Annual</td>
</tr>
<tr>
<td>Streamflow</td>
<td>Theiss</td>
<td>405203 (Goulburn River at Eildon)</td>
<td>1916</td>
<td>Current</td>
<td>Daily</td>
</tr>
</tbody>
</table>

---

39 Water Move conducts water exchanges for all water trading zones in Victoria where trading rules have been defined. Goulburn Murray Water (GMW) is the main water authority in the Goulburn Region. The Australian Bureau of Statistics (ABS) collects data on land use. Theiss is responsible for streamflow monitoring across Victoria.

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<table>
<thead>
<tr>
<th>Parameter</th>
<th>Source39</th>
<th>Scale</th>
<th>Start Date</th>
<th>End date</th>
<th>Continuity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Streamflow</td>
<td>Theiss</td>
<td>405232 (Goulburn River at McCoy Bridge)</td>
<td>1965</td>
<td>Current</td>
<td>Daily</td>
</tr>
<tr>
<td>Streamflow</td>
<td>Theiss</td>
<td>405200 (Goulburn River at Murchison)</td>
<td>1881</td>
<td>Current</td>
<td>Daily</td>
</tr>
<tr>
<td>Streamflow</td>
<td>Theiss</td>
<td>405201 (Goulburn River at Tarwool)</td>
<td>1908</td>
<td>Current</td>
<td>Daily</td>
</tr>
<tr>
<td>Streamflow</td>
<td>Theiss</td>
<td>405204 (Goulburn River at Shepparton)</td>
<td>1921</td>
<td>Current</td>
<td>Daily</td>
</tr>
</tbody>
</table>

**A.2. Details of the Goulburn River as modelled in the GSM**

A detailed schematic of how the Goulburn River is represented in the GSM is shown in Figure A-1. The GSM is constructed using REALM (Resource Allocation Model) and uses a series of arcs (representing pipes or river reaches) and nodes (representing junctions) to model the system. Each arc has a specified penalty that describes the priority of providing flow down that particular arc over other arcs in the system (where a large negative number represents a high penalty if flow is not provided). Arc capacity constraints are used to set “rules” as to when certain arcs should flow. The capacity constraints can be set either as fixed (for example, a capacity of 100 would limit the flow to 100 ML/m in the given arc) or variable where the capacity is linked to conditions in other arcs or storages (for example, the capacity may be a factor multiplied by the flow in another arc).

Table A-3 lists details of the arcs relevant to this research project. The labelling convention (arc number and name) used in the GSM has been followed. A description of each arc is provided, along with the capacity and penalty associated with each arc. The GSM obviously extends to include detailed modelling of the Loddon and Campaspe basins, however these areas are not relevant to this research. The GSM includes a series of “accounting” arcs to determine allocation levels: these are not detailed here. For further information on these elements of the GSM, refer to the User Manual (DSE, 2003).

A number of arcs are used to determine releases from Lake Eildon. These include arcs to define flood pre releases (5), state electricity releases (8 and 9), environmental flood
releases (234, 235, 236), irrigation and compensation flow releases (1) and spills (7). An arc (10) is included to ensure that the combined releases do not cause downstream flooding. The compensation flow (or minimum flow requirements) downstream of Lake Eildon are defined by arcs 239 and 246. These arcs ensure that the flow passing this reach is at least equal to the minimum flow requirement. Any excess flow travels through a third arc (3). The minimum flow arcs ensure that the combined passing flow is adequate; they do not calculate specifically the release from Eildon to meet minimum flow requirements. If a release is required from Eildon specifically to meet a minimum flow, it is released using the same arc that controls irrigation releases (1).

The majority of water released from Eildon is diverted at Goulburn Weir. Flow passing downstream of Goulburn Weir is defined using three arcs. While an attempt is made to divert the majority of water at Goulburn Weir, in reality there are some passing flows and arc 18 allows for these “non ideal” diversions. Similarly, when an environmental flood is released from Eildon, not all of this water will be harvested at Goulburn Weir and thus some will flow downstream (defined by arc 241). A third arc (18) allows for additional releases to meet downstream irrigation demands and minimum flow requirements. The compensation flow (or minimum flow requirement) is defined by arc 245, with any excess flow in the river travelling down arc 19.

The compensation flow requirement downstream of McCoys bridge is defined by arc 244, with additional flows in the river travelling through the parallel arc 171. As is the case with Lake Eildon and Goulburn Weir minimum flows, these arcs do not define specific environmental releases but are used to force a minimum flow (comprised of irrigation and specific environmental releases) to occur.
Figure A-1: Schematic of relevant section of the GSM
Table A-2: Details of relevant arcs in the GSM

<table>
<thead>
<tr>
<th>Arc #</th>
<th>Arc Name</th>
<th>Description</th>
<th>Capacity</th>
<th>Penalty</th>
</tr>
</thead>
<tbody>
<tr>
<td>-</td>
<td>EILDON INFLOW</td>
<td>Inflow to Lake Eildon</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>EILDON REL #1</td>
<td>Controls releases from Eildon at Dead Storage. Includes Irrigation and compensation flow releases.</td>
<td>Variable capacity arc based on Eildon storage volume.</td>
<td>0</td>
</tr>
<tr>
<td>2</td>
<td>EILDON REL #2</td>
<td>Ensures that releases from Eildon don’t cause downstream flooding (maximum 12000 ML/D).</td>
<td>Variable capacity based on downstream natural inflows.</td>
<td>0</td>
</tr>
<tr>
<td>7</td>
<td>EILDON SPILLS</td>
<td>Spills from Lake Eildon</td>
<td>Fixed Capacity</td>
<td>100</td>
</tr>
<tr>
<td>5</td>
<td>EILDON FLOOD PRE RELEASE</td>
<td>Forces flood pre release based on target curves for Lake Eildon.</td>
<td>Variable based on Eildon storage levels and Eildon target curve</td>
<td>-4000000</td>
</tr>
<tr>
<td>8</td>
<td>SECV REL#1</td>
<td>Release from Eildon to meet hydropower demands</td>
<td>Variable based on SECV entitlement</td>
<td>-4100000</td>
</tr>
<tr>
<td>9</td>
<td>SECV REL#2</td>
<td>Controls hydropower releases at dead storage level</td>
<td>Variable based on Eildon Storage</td>
<td>10</td>
</tr>
<tr>
<td>234</td>
<td>EIL #1 ENV FLOOD</td>
<td>Release for environmental flood, occurs if target inflow exceeded.</td>
<td>Variable based on trigger inflow to Eildon for Environmental Floods</td>
<td>-5000000</td>
</tr>
<tr>
<td>235</td>
<td>EIL #2 ENV FLOOD</td>
<td>Prevents environmental flood release occurring if spill has already occurred in Oct or Sep</td>
<td>Variable based on previous months events</td>
<td>0</td>
</tr>
<tr>
<td>236</td>
<td>EIL #3 ENV FLOOD</td>
<td>Adjusts flood release to take account of other releases occurring in Nov</td>
<td>Variable capacity arc based on other releases from Eildon.</td>
<td>0</td>
</tr>
<tr>
<td>10</td>
<td>EILDON NO FLOOD</td>
<td>Ensures that total release from Eildon will not cause flooding</td>
<td>Variable based on downstream inflows and Eildon releases</td>
<td>0</td>
</tr>
<tr>
<td>3</td>
<td>EIL TO TRA</td>
<td>Instream flows from Eildon to Trawool (to meet d/s demands, excludes minimum compensation flow)</td>
<td>Fixed Capacity</td>
<td>10</td>
</tr>
<tr>
<td>246</td>
<td>MIN D/S EILDON</td>
<td>Ensures a compensation flow of 120 ML/d downstream of Eildon</td>
<td>Fixed Capacity (changes each month depending on no. days – days x 120)</td>
<td>-53000000</td>
</tr>
<tr>
<td>239</td>
<td>X MIN D/S EILDON</td>
<td>Increases min flow to 250 ML/d if trigger inflow to Eildon exceeded</td>
<td>Variable Capacity based on trigger inflow to Eildon</td>
<td>-53000000</td>
</tr>
<tr>
<td>18</td>
<td>GW TO GR INF SPILLS</td>
<td>Forces spills at Goulburn Weir to allow for non ideal</td>
<td>Variable based on natural inflows at Trawool and</td>
<td>-4100000</td>
</tr>
<tr>
<td>Arc #</td>
<td>Arc Name</td>
<td>Description</td>
<td>Capacity</td>
<td>Penalty</td>
</tr>
<tr>
<td>-------</td>
<td>---------------------------</td>
<td>------------------------------------------------------------------------------</td>
<td>---------------------------------------------------------</td>
<td>-------------</td>
</tr>
<tr>
<td></td>
<td><em>diversion of natural inflows below Lake Eildon</em></td>
<td></td>
<td>Goulburn Weir.</td>
<td></td>
</tr>
<tr>
<td>241</td>
<td>EIL #4 ENV FLOOD</td>
<td>Forces part of Environmental flood release from Lake Eildon to spill – excess of diversion capacity</td>
<td>Variable capacity based on the environmental flood release from Lake Eildon</td>
<td>-5000000</td>
</tr>
<tr>
<td>17</td>
<td>GW TO GOULB R</td>
<td>Additional release to Goulburn River to meet downstream demands and compensation flow requirements</td>
<td>Fixed Capacity</td>
<td>10</td>
</tr>
<tr>
<td>245</td>
<td>MIN D/S GWEIR</td>
<td>Ensures minimum passing flow of 250 ML/d</td>
<td>Fixed Capacity (changes each month depending on no. days – days x 250)</td>
<td>-5300000</td>
</tr>
<tr>
<td>19</td>
<td>GR TO MURCHISON</td>
<td>Additional flow (in excess of minimum flow) in Goulburn River</td>
<td>Fixed Capacity</td>
<td>10</td>
</tr>
<tr>
<td>244</td>
<td>MIN D/S MCCOYS</td>
<td>Ensures minimum compensation flow downstream of McCoys Bridge of 350 ML/d from Nov – Jun and 400 ML/d from Jul - Oct</td>
<td>Fixed Capacity (changes each month depending on no. days)</td>
<td>-5300000</td>
</tr>
<tr>
<td>171</td>
<td>D/S MCCOYS</td>
<td>Flow instream in excess of compensation flow</td>
<td>Fixed capacity</td>
<td>0</td>
</tr>
</tbody>
</table>

**A.3. Determining Optimisation Model Inputs**

The optimisation model requires the inputs to determine passing flows downstream of Lake Eildon and downstream of Goulburn Weir. The arcs used to determine each component of the passing flow are shown in Table A-3.

**Table A-3: GSM arcs used to calculate optimisation model inputs**

<table>
<thead>
<tr>
<th>Flow Element</th>
<th>Relevant GSM arc</th>
</tr>
</thead>
<tbody>
<tr>
<td>SECV releases</td>
<td>8</td>
</tr>
<tr>
<td>Lake Eildon spills and flood releases</td>
<td>7, 5</td>
</tr>
<tr>
<td>Lake Eildon irrigation releases</td>
<td>1</td>
</tr>
<tr>
<td>Lake Eildon minimum flow releases</td>
<td>1, 234</td>
</tr>
<tr>
<td>Goulburn Weir spills</td>
<td>241, 18</td>
</tr>
<tr>
<td>Goulburn Weir minimum flow</td>
<td>17</td>
</tr>
</tbody>
</table>
The current GSM model incorporates compensation releases in arcs used to meet other downstream water requirements. While there are arcs included to ensure compensation flows are indeed met (for example, arcs 245, 246, 239 and 244), these arcs are used to ensure that all passing flows together exceed the minimum flow rather than to calculate the additional volume of water released from storage purely to meet minimum flow requirements. The issue is further complicated as the minimum flow compliance points are in sequence with different demands, inflows and losses occurring along the system. It is therefore difficult to ascertain which compliance point is driving the need for an additional release from storage.

The optimisation model requires irrigation releases and compensation releases to be separated for two reasons:

- If the environmental allocation is increased, the irrigation allocation must be adjusted. If the compensation release is included with the irrigation releases, it will also be factored. This requires the monthly pattern of minimum flow releases.
- When considering removing the minimum passing flow requirements and transferring this parcel of water into an environmental entitlement, the annual entitlement volume needs to be determined. This requires identifying environmental releases separately to irrigation releases. An annual volume (rather than a monthly pattern) is adequate.

Isolating the compensation flow releases from the irrigation releases required making adjustments to the existing GSM model. The existing minimum flow arcs (245, 246, 239 and 244) were set to zero capacity (effectively “turned off”). A series of accounting arcs were created to determine the additional release required to meet the minimum flow requirements. An additional arc was added leaving Lake Eildon, transporting environmental releases to a terminal node (refer to Figure A-2 and Table A-4). The reliability of supply to consumptive users was compared to the original GSM model to ensure that removing minimum flow releases does not impact on consumptive users (Figure A-3). Under the current system operation, releases made from Lake Eildon to satisfy minimum flow requirements can then be diverted by other users downstream.
Converting these releases to an entitlement changes the parcel of water to an excludable volume of water unavailable for other users. The volume of the Environmental Entitlement needs to reflect this, ensuring that the reliability of supply to other users is not affected.
Figure A-2: Schematic showing changes to the GSM model to isolate compensation flow release
<table>
<thead>
<tr>
<th>Arc #</th>
<th>Arc Name</th>
<th>Description</th>
<th>Capacity</th>
<th>Penalty</th>
</tr>
</thead>
<tbody>
<tr>
<td>244</td>
<td>MIN D/S</td>
<td>TURNED OFF</td>
<td>Fixed Capacity (SET TO ZERO)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MCCOYS</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>246</td>
<td>MIN D/S</td>
<td>TURNED OFF</td>
<td>Fixed Capacity (SET TO ZERO)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>EILDON</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>239</td>
<td>X MIN D/S</td>
<td>TURNED OFF</td>
<td>Fixed Capacity (SET TO ZERO)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>EILDON</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>245</td>
<td>MIN D/S</td>
<td>TURNED OFF</td>
<td>Fixed Capacity (SET TO ZERO)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>GWEIR</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>735</td>
<td>ENV TOT</td>
<td>Total compensation release from Lake Eildon (Lake Eildon to Terminal Node)</td>
<td>Variable Capacity set as the maximum of the capacities of arcs 607, 608, 609, and then limited by arc 2 (to ensure release doesn’t cause flooding)</td>
<td>-5300000</td>
</tr>
<tr>
<td></td>
<td>COMP REL</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>736</td>
<td>MIN REC</td>
<td>Accounting arc to define minimum flow requirement downstream of Eildon of 120 ML/d (based on previous arc 246)</td>
<td>Fixed Capacity (changes each month depending on no. days – days x 120)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>D/S EIL</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>737</td>
<td>MIN REC</td>
<td>Accounting arc to define minimum flow requirement downstream of Goulburn Weir of 250 ML/d (based on previous arc 245)</td>
<td>Fixed Capacity (changes each month depending on no. days – days x 250)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>D/S GW</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>738</td>
<td>MIN REC</td>
<td>Accounting arc to define minimum flow requirement downstream of McCoyrs Bridge (based on previous arc 244)</td>
<td>Fixed Capacity (changes each month depending on no. days)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>D/S MCCOYS</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>739</td>
<td>XMIN REC</td>
<td>Accounting arc to define minimum flow requirement downstream of Eildon (based on previous arc 239)</td>
<td>Variable Capacity based on trigger inflow to Eildon</td>
<td></td>
</tr>
<tr>
<td></td>
<td>D/S EIL</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>740</td>
<td>ENV D/S</td>
<td>Accounting arc to define the additional env release required to meet flow requirements downstream of Lake Eildon</td>
<td>Variable Capacity based on the sum of capacity of arcs 603 and 606, minus arc 3 flow (other passing flows) (Minimum set to zero)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>EILDON</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>741</td>
<td>ENV D/S</td>
<td>Accounting arc to define the additional env release required to meet flow requirements downstream of Goulburn Weir</td>
<td>Variable Capacity based on the capacity of arc 604, minus arc 19 flow (other passing flows) (Minimum set to zero)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>GWEIR</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>742</td>
<td>ENV D/S</td>
<td>Accounting arc to define the additional env release required to meet flow requirements downstream of McCoyrs Bridge</td>
<td>Variable Capacity based on the capacity of arc 605, minus arc 171 flow (other passing flows) (Minimum set to zero)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MCCOYS</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Figure A-3: Comparison of allocations between original GSM model and model with minimum flow releases removed – based on February allocations for Rodney Irrigation District.

The volume of water supplied to the environmental flow arc can be used to define an annual entitlement that is equivalent to the volume of water used historically to meet minimum flow requirements. The conversion to an entitlement is based on an allocation that is met in 97% of years\textsuperscript{40}. If minimum passing flows are converted to an entitlement, the annual entitlement is thus 5480 ML (Figure A-4).

\textsuperscript{40} High security allocations in the region are historically around a 97% reliability (Department of Sustainability and Environment, 2007)
Figure A-4: Volume of water delivered to meet minimum passing flow requirements

An annual entitlement volume equivalent to the volume of water required to meet minimum flow requirements has been determined. The difficulty arises now in understanding the monthly release pattern of this water. Recall that the compensation releases and irrigation releases currently travel through the same modelling arc in GSM. To separate these releases, a monthly release pattern for the compensation releases is required. Again difficulty arises due to the non excludable nature of compensation flows. A release may be made from Lake Eildon specifically to meet a minimum flow requirement, however this water can then harvested downstream and used elsewhere in the system for consumptive purposes. Natural inflows also provide a large portion of water available in the catchment. If the natural inflows are first allocated to minimum flow requirements, only a small additional release may be required to meet minimum flow requirements while a larger release would be required to meet irrigation requirements. However, if this natural inflow is first allocated to consumptive users, a large release from storage would be required to meet the minimum flow requirements with only a smaller portion released to meet irrigation demands. Thus, determining the releases specifically for the purposes of fulfilling minimum flow requirements becomes complicated.
The approach adopted is as follows:

- The monthly flow pattern in arc 735 (in the edited version of GSM shown in Figure A-2) determines the additional volume of water required at each timestep to meet all minimum flow requirements.
- Arc 1 in the original GSM model (the models current operations) shows the combined irrigation and compensation releases from Lake Eildon.
- The minimum of these two series was taken to be the monthly compensation release pattern from Lake Eildon.

This process is further demonstrated in Figure A-5 (using the year 1892). In the months June through to November, minimum flow requirements are satisfied by other releases (be they irrigation releases, SECV releases or spills). In the months December to March, an additional volume is required to meet minimum flow requirements. This volume is well within the volume that was released from Eildon to meet irrigation and compensation flow requirements at that timestep. It is therefore assumed that the additional volume of water was all provided from Lake Eildon releases. However in April and May, the volume of water released from Lake Eildon is less than the required additional volume to meet minimum flow requirements. Therefore, it is assumed the release from Eildon is aimed at meeting minimum flow requirements, with the additional volume of water required provided by downstream natural inflows.
Figure A-5: Comparison of volume of additional water required to meet minimum flow requirements (arc 735 edited GSM) and the actual releases from Lake Eildon (arc 1 original GSM).
Appendix B  Climate sequences used in the multi-year modelling

Table B-1 shows the climate sequences used in the multi-year modelling. A dry year is represented by “D”, average year by “A” and a wet year by “W”. The storage conditions are modelled as low for dry years, medium for average years and high for wet years. As the combination of possible climate and storage sequences is large, only a small representative sample was modelled.

Table  B-1:  Climate sequences used in the Multi-year modelling

<table>
<thead>
<tr>
<th>Dry first year</th>
<th>Average first year</th>
<th>Wet first year</th>
</tr>
</thead>
<tbody>
<tr>
<td>DDDDD</td>
<td>AAAAAA</td>
<td>WWWWWW</td>
</tr>
<tr>
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