EXPLORING THE USE OF WATER MARKETS FOR IMPROVED ENVIRONMENTAL OUTCOMES

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IV. Abbreviations, Units and Terminology

List of Abbreviations

ABARE-BRS	Australian Bureau of Agricultural and Resource Economics and the Bureau of Rural Sciences
ABARES	Australian Bureau of Agricultural Economics and Science
ABS	Australian Bureau of Statistics
ACCC	Australian Competition and Consumer Commission
ACT	Australian Capital Territory
BOM	Bureau of Meteorology
CBWTP	Columbia Basin Water Transaction Program
CER	Clean Energy Regulator
CEWH	Commonwealth Environmental Water Holder
CEWO	Commonwealth Environmental Water Office
CGE	Computable general equilibrium model
CICES	Common International Classification of Ecosystem Services
CLLMM	Coorong, Lower Lakes and Murray Mouth
COAG	Council of Australian Governments
CRDWT	Colorado River Delta Water Trust
CSIRO	Commonwealth Scientific Research and Industrial Research Organisation
DEWHA	Department of Environment, Water, Heritage and the Arts
DEWNR	Department of Environment, Water and Natural Resources
DoAWR	Department of Agriculture and Water Resources
DoE	Department of the Environment
DoEE	Department of the Environment and Energy
EPA	Environmental Protection Agency
ES	Ecosystem services
EU ETS	European Union Emissions Trading Scheme
EWH	Environmental water holder
EWTF	Environmental water trading framework
GVAP	Gross value of agricultural product
GVIAP	Gross value of irrigated agricultural product
GWP	Global Water Partnership
HEM	Hydro-economic model
HSI	Habitat suitability index
IPCC	Inter-governmental Panel on Climate Change
IWG SC-CO$_2$	International Working Group on the Social Cost of Carbon
IWRM	Integrated water resource management
MAE	Millennium Ecosystem Assessment
MDBA	Murray-Darling Basin Authority
MDB	Murray-Darling Basin
MDB BWF	Murray-Darling Basin Balanced Water Fund
MDBC	Murray-Darling Basin Commission
MDBMC	Murray-Darling Basin Ministerial Council
NGO	Non-government organisation
nMDB	Northern Murray-Darling Basin
NPWS	National Plan for Water Security
NSW	New South Wales
NWC	National Water Commission
NWI	National Water Initiative
OECD	Organisation for Economic Co-operation and Development

OEH	Office of Environment and Heritage
OWT	Oregon Water Trust
PC	Productivity Commission
PE	Partial equilibrium
PO	Pareto optimality
PPO	Potential Pareto optimality
QLE	Qualified Local Entity
RTB	Restoring the Balance
RtB	Restoring the Balance
SA	South Australia
SC-CO_2	The social cost of carbon
SDL	Sustainable Diversion Limit
SEACI	South Eastern Australian Climate Initiative
sMDB	Southern Murray-Darling Basin
SRA	Sustainable Rivers Audit
SRWUIP	Sustainable Rural Water Use and Infrastructure Program
TC	Transaction cost
TEEB	The Economics of Ecosystems and Biodiversity
TLM	The Living Murray
TLM	The Living Murray
USDA	United States Department of Agriculture
USED	Upper South East Drainage
US-EPA	United States Environmental Protection Agency
US-FWS	United States Fish and Wildlife Service
WFF	Water for the Future
WGCS	Wentworth Group of Concerned Scientists
WTO	World Trade Organisation
WUA	Water user association

List of Units Used

$AU	Australian dollar[1]	(~0.75 $US)
AF	Acre foot	(1.23×10^6 litres)
EC	Electro conductivity	($1 \mu S/cm$)
GL	Giga litre	(1×10^9 litres)
Ha	Hectare	(1×10^{-2} km^2)
ML	Mega litre	(1×10^6 litres)
tC	Tonnes of carbon	(1×1000 kgC)
tCO_{2e}	Tonnes of carbon dioxide equivalent	(1×1000 kgCO_{2e})

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[1] Unless otherwise indicated all dollar values are in Australian dollars.

Useful Terminology

2800GL *Basin Plan* hydrological scenario	June 2009 hydrological conditions adjusted for an additional 2800GL environmental water in the MDB.
Allocation water trade (see also: water allocation)	The temporary (e.g. annual) sale of water ascribed to a water entitlement.
Appropriation	The abstraction and beneficial use of water from a water resource system. The appropriation date (see: *prior appropriation*) is set from the first date of appropriation of the water right.
Bankfull discharge	The flow volume which connects the river channel to the floodplain. Flow greater than bankfull discharge results in floodplain inundation.
Baseline hydrological scenario	June 2009 hydrological and development conditions in the MDB.
Catchment water balance	A mathematical representation of all catchment inflows, losses, storages and outflows.
Commonwealth Environmental Water Holder (CEWH)	A federal agency established by the *Water Act* (*Cwth* 2007) responsible for the management of environmental water acquired by the Australian federal government. The water rights managed by CEWH is 'held' environmental water (see: *environmental water*)
Consumptive water-use	The abstraction of water which results in the permanent removal or diminishment of water from its source. The volume of water 'taken up' is the consumptive water-use volume (e.g. domestic water-use).
Council of Australian Governments (COAG)	A group consisting of the Prime Minister of Australia, the First Ministers of Australian states and territories, and the President of the Australian Local Government Association. COAG is charged with managing matters of national significance that need to be co-ordinated across states and territories (e.g. the Basin Plan).
Decision horizon (also: *planning horizon*)	The number of years considered in a forecast of future events and relevant information when making a decision (see also: *rolling horizon*).
Demand-based water management	Water management strategies that seek to change the consumptive demand patterns for freshwater through the use of voluntary measures (e.g. education programs), rules and regulation (e.g. water restrictions), and economic instruments (e.g. water markets and water pricing).
Economic water scarcity	Water scarcity that occurs when a population does not have the necessary monetary or human capital to access volumes of water adequate to meet consumptive demand.
Ecosystem services	The services provided by the natural environment, which support the survival and well-being of human populations.
Entitlement water trade	The permanent sale of a water entitlement or water right (see also: *water entitlement*).
Environmental asset	Naturally occurring ecosystems or biomes that provides environmental services or functions. (Note that this definition is adopted from the OECD (2005) *Handbook of National Accounting* and differs from the ABS definition, which requires an asset to have an identifiable owner who derives economic benefit from holding or using the environmental asset (ABS, 2010a).
Environmental steward	An individual or entity whom engages in the sustainable use and protection of the natural environment and its functions.
Environmental water	Water designated to maintain, protect or restore ecological character of freshwater ecosystems. In the MDB, environmental water is either *rules-based* (volume prescribed by water-sharing plans) or *held* (acquired by an environmental water holder from consumptive water users).

Environmental water holder	An entity, government or otherwise, with a mandate to manage acquired (or *'held'*) water rights/entitlement for environmental benefit
Environmental water product (also: *environmental water transfer tool*)	A water right/entitlement, or aspect thereof, that is acquired on the water market and used for environmental use (see also: *water product*).
Environmental water transaction (also: environmental water transfer)	A market style transaction in which an environmental water product is transferred from consumptive use to environmental use.
Federation drought	A hydrological drought affecting much of Australia from 1897 to 1903.
Flow-dependent ecosystem	Natural ecosystems dependent upon the provision of freshwater flows to support and maintain ecological function.
Groundwater	Fresh water stored below the surface of the Earth in the soil and in acquirers.
High security water entitlement	A water entitlement which reliably yields full allocation volumes in approximately 90-95 years out of 100 in the MDB, with little variation between years except during extreme drought conditions.
Hydro-economic model	A mathematical model integrating hydrological and economic principles.
Hydrological catchment (also: *drainage basin*)	The geographic extent for which all surface water runoff converges to a single low-elevation point. In the MDB, a 'catchment' often refers to one of the 22 sub-catchments of the wider MDB.
Hydrological indicator site	The 18 environmental assets in the MDB for which environmental watering requirements were quantified for the Basin Plan.
Long-term average annual yield	The expected long-term average yield from a water entitlement over a 100 year period.
Low/general security water entitlement	A water entitlement which yields a variable allocation depending on water availability. General security water entitlements provide a LTAAY of 42-81% and low security yields between 24-35% in the MDB.
Millennium drought	A prolonged hydrological drought affecting the majority of south-eastern Australia. Prolonged periods of dry conditions were experienced from late 1996 to mid-2010 (BOMb, 2015), and the height of the Millennium Drought was experienced from 2001 to 2009.
Non-consumptive water-use	The use of water which does not result in the permanent removal or diminishment of water from its source (e.g. in-situ environmental watering)
Non-government organisation	An organisation, typically non-profit, which is independent from state, federal or international governments.
Normative framework	A framework or method to help analyse normative trade-offs.
Over-allocated (also: *over-appropriated*)	The state of a water resource system when the volume of water rights/entitlements held by consumptive and non-consumptive water users exceeds the long-term average volume of water available for use.
Physical water scarcity	Water scarcity that occurs when water resource development is approaching or has exceeded sustainable diversion limits, such that there is not enough water to meet all demands.
Prior appropriation doctrine	The dominant water law applied to the western United States, characterised by "*first in time, first in right*" abstraction priorities and beneficial use clauses.
Ramsar Convention	An international treaty identifying internationally important wetlands developed in Ramsar, Iran.
Regulated river	A river subject to major water resource development such that downstream flows are regulated by major upstream water storage infrastructure.

Resilience	The ability for a system to resist or recover from perturbation to its original long-term state.
River Red Gum (also: *Eucalyptus camaldulensis*)	A large, long-lived Eucalyptus tree species indigenous to Australia. It is commonly found along watercourses and is reliant upon regular flooding in ecologically desirable volumes and frequencies.
Rolling horizon	In a rolling horizon decision-making approach, the decision maker makes the most immediate decision (e.g. decisions in the first period) based on knowledge of conditions in the present period and an uncertain (e.g. probabilistic or stochastic) forecast of the remaining years in the decision horizon. In the next period, the second period decisions become the most immediate and the decision horizon is pushed out further into the future, and is thus 'rolled over'.
Supply-based water management	Water management strategies that seek to increase and more effectively manage the supply of freshwater for consumptive use, including measures such as water treatment plants (e.g. desalinisation), water transport and delivery systems (e.g. pipes, channels), infrastructure construction and upgrades (e.g. dams and weirs).
Surface-water	Fresh water stored in the surface of the Earth in rivers, streams, lakes and reservoirs.
Sustainable diversion limit (SDL)	An upper level of water-use that limits the amount of water that can be used for consumptive purposes in the MDB.
The Commonwealth	The Federal Commonwealth Government of Australia
The Murray-Darling Basin Plan (also: *the Basin Plan*)	A co-ordinated approach between state, territory and federal governments to manage water resources in the MDB. The *Basin Plan* was passed into law in 2012 and the initial sustainable diversion limits will come into full effect in 2019.
The Paris Agreement	A global agreement on climate change mitigation actions developed during the 21st session of the Conference of the Parties to the United Nations Framework Convention on Climate Change held in Paris, France. Australia ratified the Paris Agreement in 2016 and it entered into force in November of that year.
Water allocation	The volume of water credited to a water entitlement able to be used in a season, dependent on the total water available in the system and the security of the water entitlement.
Water entitlement (also: *water access entitlement*)	The right to an ongoing share of a total amount of water available in a water resource system. Water entitlements yield annual water allocations. Internationally and commonly referred to as a *water right*.
Water product	A water right, or aspect thereof, that is sold on the water market. Water products are characterised by the type (e.g. groundwater, surface-water), the duration of the water right exchange (e.g. annual, permanent, 5 year lease), and any other contractual characteristics (e.g. dry-year trigger arrangement). (See also: *environmental water product*).
Water user association (also: *water user board*)	A group of water users, such as irrigators, who combine financial, technical and social resources to manage and maintain a water supply system.
Water year (also: *water accounting year*)	The 12 months period commencing each year on the 1st of July.
Without development hydrological scenario	Modelled hydrological conditions without any water resource development and major water abstraction in the MDB.

V.   Abstract

Water resource development has historically proceeded with little consideration for the environment. In the Murray-Darling Basin (MDB) Australia, water resource development has resulted in considerable ecological degradation and a diminishment of flow-dependent ecosystem services (ES). In response, MDB water policy has undergone considerable reform in the past decade, culminating in a commitment to reallocate water from consumptive use back to the environment. This thesis examines potential further use of water markets, and issues associated with this, to provide greater and more efficient environmental flows.

The main question investigated in this thesis was the potential for an environmental water holder (EWH) to use the water allocation market to reallocate water to the environment for improved ecological condition and ES generation. To answer this question, an interdisciplinary mixed-methods approach was employed involving: a) the development of a hydro-economic model that simulates the annual trade decisions of a forward-looking EWH in a MDB sub-catchment; b) 49 qualitative face-to-face interviews across the US and Australia with stakeholders from industry, non-profit and government agencies regarding the role of non-government environmental water holders (NGO EWHs); and c) quantitative survey analysis of 1,000 southern MDB irrigator preferences in 2015-2016 for the sale of environmental water.

Key findings of this thesis show that trading water allocations for the environment can have positive ES benefits by improving floodplain inundation. Under particular hydrological and fiscal conditions, the increase in floodplain carbon storage may be of sufficient market value to offset the cost of environmental water allocation purchases. This indicates a potential carbon-water trading strategy which may provide a novel revenue stream for self-financing EWHs. It was shown that NGO EWHs play a unique role in environmental water reallocation through the provision of flexible and multi-functional water trade arrangements. Results also highlight the importance of social capital in facilitating successful environmental water trades. Lastly, results demonstrate that southern MDB irrigators show a clear preference for the local management of water resources, and in particular NSW and Victorian irrigators rank the federal government among their least preferred buyers of water entitlements. Southern MDB irrigators also demonstrate a clear preference for the use of water allocation trade for the environment.

Key recommendations based on these results include: a) the further judicious expansion of market-based reallocation policies, particularly water allocation trade, in place of continued infrastructure-based reallocation; b) the use of carbon credit generation and sale to cost-effectively finance annual environmental water reallocation; and c) the need to encourage an increased partnership of federal agencies with local and NGO EWHs, in order to increase irrigator participation and sustain local values in environmental water management.

VI. Thesis Declaration Statement

I certify that this work contains no material which has been accepted for the award of any other degree or diploma in my name, in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text. In addition, I certify that no part of this work will, in the future, be used in a submission in my name, for any other degree or diploma in any university or other tertiary institution without the prior approval of the University of Adelaide and where applicable, any partner institution responsible for the joint-award of this degree. I give consent to this copy of my thesis when deposited in the University Library, being made available for loan and photocopying, subject to the provisions of the Copyright Act 1968. I acknowledge that copyright of published works contained within this thesis resides with the copyright holder(s) of those works. I also give permission for the digital version of my thesis to be made available on the web, via the University's digital research repository, the Library Search and also through web search engines, unless permission has been granted by the University to restrict access for a period of time.

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Claire Settre

September, 2017

VII.   Acknowledgements

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VIII.   Publications and Presentations from this Thesis

Academic Journal Articles (peer-reviewed)

Settre, C. & Wheeler, S. (2017) A century of interventions in a Ramsar wetland – the case of the Coorong, Lower Lakes and Murray Mouth, *Australasian Journal of Environmental Management,* 24(2), pp. 1-21.

Settre, C., Connor, J. & Wheeler, S. (2017) Reviewing the treatment of uncertainty in hydro-economic modelling in the Murray-Darling Basin. *Water Economics and Policy,* 3(3), pp.35, DOI: 10.1142/S238264X16500429

Book Chapters (peer-reviewed)

Settre, C. & Wheeler, S. (2016) Environmental water governance in Australia: the movement from regulation and engineering to economic-based instruments. In: V. Ramiah and G. Gregoriou (eds), *Handbook of Environmental and Sustainable Finance*, New York: Elsevier, pp. 67-91.

Settre, C. & Wheeler, S. (2017) Rebalancing the system: acquiring water and trade. In: A. Horn, A. Webb, M. Stewardson, B. Richter, B., M. Acreman (eds), *Water for the Environment: From Policy and Science to Implementation and Management.* London: Elsevier, pp. 399-419.

Current Working Papers

Settre, C., Connor, J. & Wheeler, S. (2017) Can the generation and sale of carbon credits in natively vegetated floodplains offset the costs of annual environmental water purchases?

Settre, C., Connor, J. & Wheeler, S. (2017) Strategic water and carbon market opportunities for a self-financed environment water holder to improve floodplain forest carbon storage.

Settre, C. (2017) Localism, community engagement and 40 Cups of coffee: the role of non-government organisations as stewards of environmental water in the western USA and potential insights for the Murray-Darling Basin, Australia.

Settre, C., Connor, J. & Wheeler, S. (2017) Estimating the economic value of environmental water with ecosystem service supply modelling.

Conference Papers and Seminars

Settre, C. (2017) Water without borders: the who, what and why of international water research. *The University of California, Riverside, Extension Centre Research Series,* 17th February 2017, Riverside, USA.

Settre, C. (2017) Realising floodplain benefits of environmental water reallocation with improved ecosystem services accounting. *The University of California, Riverside, School of Public Policy Seminar Series.* 24th October 2017, Riverside, USA.

Settre, C., Connor, J. & Wheeler, S. (2016) Ecosystem service dynamics in the Murray-Darling Basin: managing water for improved carbon stocks. *NCCARF and CSIRO Climate Adaptation Conference,* 5th – 7th of July 2016, Adelaide, Australia.

Settre, C., Connor, J., Wheeler, S. (2016) Ecosystem service dynamics in the Murrumbidgee catchment, Australia. *Natural Resource Management Science Conference*, 13th -15th April 2016, Adelaide, Australia.

Settre, C., Connor, J., Wheeler, S. (2016) Improving ecosystem services through trade in the Murrumbidgee catchment, Australia, *60th National Australian and Agricultural Resource Economics Society Conference*, 2nd-5th February, 2016. Canberra, Australia.

Settre, C. (2015) Informing environmental water management using ecosystem service values in the Murrumbidgee catchment, Australia. *The University of Adelaide Centre of Global Food Studies Seminar Series,* Adelaide, Australia.

Settre, C. (2015) A century of water resource management in a Ramsar wetland – a case study of the Coorong, Lower Lakes and Murray Mouth, Australia, *59th National Australian Agricultural Resource Economics Society Conference*, 10th-13th February 2015, Rotorua, New Zealand.

Settre, C. (2014) Australia's environmental water governance: from state to market? *Global Conference on Managing in Recovering Markets*, 8th-9th November 2014, Adelaide, Australia.

IX.   Additional Publications during Candidature

Academic Journal Articles (peer-reviewed)

Loch, A., Wheeler, S., **Settre, C**. (*forthcoming*) Private transaction costs of water trade in the Murray-Darling Basin. *Ecological Economics.*

Book Chapters (peer-reviewed)

Palomo-Hierro, S., **Settre, C.,** Wheeler, S., Loch, A. (2016) The Australian and Californian water market experience: an example to follow? In: J. Gomez-Limon and J. Calatrava (eds.), *Water Markets in Spain: Present and Future,* Cajamar Caja Rural, Spain, pp. 315-345.

CHAPTER 1: Introduction

Water resource development has historically proceeded with little consideration for environmental watering requirements. In cases where water is scarce, this has resulted in considerable damage to flow-dependent environments and the services they provide. To improve environmental conditions and to increase the provision of ecosystem services, it is necessary to increase the volume and frequency of water available to the environment. When water rights are fully allocated, this must be achieved through the reallocation of water from consumptive water users to the environment. In a number of semi-arid regions around the world, market-based water policy has been adopted to facilitate the transfer of water from irrigation to the environment. In the Murray-Darling Basin (MDB) in Australia, water market institutions are in an ongoing state of development and expansion. This provides an opportunity to investigate how water markets may be used to improve environmental condition and ecosystem service provision. In particular, this thesis examines how environmental water holders (EWHs) can use water markets to improve environmental conditions for net social benefits. This research is carried out in the context of the MDB and select case studies from the western US.

This chapter provides an introduction to water resources in the MDB and the policy problems faced; a literature review highlighting the gaps and subsequent research questions that are to be addressed in this thesis; and an explanation of the research methodology employed. The introduction concludes with a summary of the overall thesis structure and the publications produced.

1.1 Water Resources in the MDB

1.1.1 MDB Water Resources

The MDB is Australia's largest and most regulated river system. It spans 1,061,469 km^2 and covers parts of four states and one territory and is located in the south-east of Australia. The MDB is separated into two regions: a) the northern basin (nMDB); and b) the southern basin (sMDB). The sMDB is highly regulated and contains the majority irrigated farm land in the MDB (MDBA, 2015) as well as nationally and internationally significant environmental assets (DoE, 2013). The sMDB is also hydrologically connected, which allows water trade to occur between irrigation communities and the environment in South Australia, Victoria and New South Wales (Wheeler et al., 2012). The sMDB is the focus of this thesis. Comparatively, the nMDB is less regulated, has higher rainfall and runoff variability, and has lower levels of water abstraction compared to the sMDB (BOM, 2014c).

Within the MDB rainfall is low and highly variable compared to other parts of Australia (ABS, 2008b). The vast majority of rainfall is evapotranspired, leaving roughly 4% as runoff available to be used for consumptive and *in situ* use purposes (ABARES, 2008b). Within the MDB, surface-water supplies a number of water users including irrigation, potable and domestic supplies, the water supply industry (e.g. losses in water transport), as well as recreational/tourism uses, cultural water supply, and

environmental water uses. Irrigated agriculture is the highest consumer of water in the MDB and accounted for 82% of water consumption in 2004-2005 (ABS, 2008a). Water resources, particularly in the sMDB, have been highly developed to provide reliable water supply for irrigation and potable water demands (MDBA, 2014b). Water resource development was also accompanied by over-allocation of water entitlements in the sMDB; the combined effect leading to considerable ecological damage to floodplains and wetlands, due to alternations of the natural flow regime (Kingsford, 2000).

In addition to water scarcity, the MDB has historically been subject to high variability including periods of extreme drought and flooding. The most recent is the Millennium Drought which resulted in reduced inflows of approximately 40% of the long-term average in the MDB (ABS, 2007) and persisted in the sMDB from 1996 to 2010 (BOM, 2015). Reduction in total overall water availability had an impact on irrigation productive capacity, however the reduction of economic returns to irrigation was less than the reduction in water availability due to drought (Kirby et al., 2014b) due to farmer adaptation such as; fallowing, input substitution, deficit irrigation, land use change, technological upgrades and water trading (Qureshi et al., 2013c; Qureshi and Whitten, 2014; Kirby et al., 2014b). From an environmental perspective, during the period 2004-2007, it was found overall that MDB environmental assets, including bird and fish species, vegetation and wetland areas, were in very poor to moderate conditions (Davies et al., 2010).

During the Millennium Drought the severity of over-allocation and the impacts ecological water resource development was exposed by the drastic and rapid reduction in water availability. The Millennium Drought had particular implications for policy makers due to its severity and longevity (Adamson, 2015). The subsequent need for a rapid policy response to manage the decline in the MDB resources spurred one of the most significant reform processes in Australian water policy history.

In the future drought is projected to become more frequent and severe in the MDB (Chiew et al., 2011) and the severity of the Millennium Drought may be thought of as a window into the future of what can be expected under climate change conditions in the MDB (Grafton et al., 2014). Looking forward, an 11% median reduction in average annual surface water runoff is expected across the MDB (CSIRO, 2008a). Larger declines in surface water availability are expected to occur in the sMDB, with an average annual reduction in runoff of up to 15% in the southernmost catchments of the sMDB by 2030 (CSIRO 2008). Reduced surface water availability is expected to be accompanied by increased salinity (Beare and Heaney, 2002); water quality deterioration (e.g. algae blooms) (MDBA, 2016c); and heightened variability of supply (Connor et al., 2012). Reductions in water availability will naturally affect the volumetric supplies available for irrigation and environmental use. This future increase in scarcity pressure and variability highlights the need for ongoing policy development to establish flexible arrangements in managing competition between consumptive water users and the environment.

1.1.2   Water Reform and the Murray-Darling Basin Plan

Water policy in the MDB has undergone significant and ongoing water reforms at the national level, to address over-allocation of water entitlements and establish a balance between consumptive and non-consumptive water users. This has involved the transition from predominately engineering and regulation based management approaches motivated by national development objectives, to a more pluralistic governance approach involving the government, market and civil sector.

Arguably the most significant aspect of the recent reform process was the *National Water Initiative* (2004) and the *Water Act* (2007) (Wheeler, 2014). Importantly, the passage of the *Water Act* (2007) created a series of federal-level institutions responsible for developing, enacting and monitoring the Basin Plan, which was designed to achieve water resource sustainability in the Basin. Among these institutions was the Murray-Darling Basin Authority (MDBA - formerly the Murray-Darling Basin Commission (MDBC)) charged with the development of the Basin Plan. In addition, the *Water Act* (2007) also created the Commonwealth Environmental Water Office (CEWO) and the Commonwealth Environmental Water Holder (CEWH), responsible for acquiring and managing environmental water holdings for positive ecological outcomes, respectively.

In 2012, the Basin Plan was passed into law after multiple attempts to disallow it (Loch et al., 2014b). The Basin Plan has since been the overarching framework guiding the reallocation from consumptive uses to the environment. The central aim of the Basin Plan is *"to promote the use and management of the Basin water resources in a way that optimises economic, social and environmental outcomes"* (*Water Act* 2007, Part 2, Subdivision B, 20(2)). A necessary step to achieve this aim was the specification of Basin-wide long-term average sustainable diversion limits (SDLs). Bringing water-use in line with the SDLs was to be achieved by reallocating an initial target of 2,750GL of private consumptive water to non-consumptive, plus an additional 450GL of water acquired through efficiency or supply mechanisms (DoAWR, 2017), thus giving a total of 3,200GL to be returned to the environment. At its core, the Basin Plan is a common property approach to rectifying the over-allocation of water rights in the Basin (Adamson, 2015). The newly acquired water is held in public trust by the CEWO and managed by the CEWH to achieve optimal environmental outcomes in line with the *Basin Plan* objectives. At present, the MDB is in the implementation phase of the Basin Plan, with the initial 2,750GL to be acquired by 2019 and the additional 450GL to be acquired by 2024 (DoAWR, 2017).

1.1.3   Current Policy Platform

The *Basin Plan* stipulates two primary mechanisms to acquire the 2,750GL of water for the environment. One is a market-based approach called *Restoring the Balance (RtB),* which relies heavily on the purchase of water entitlements (Wheeler et al., 2013a). For a given budget, purchasing water entitlements is the most efficient means for the Australian government to acquire water for the environment (Grafton, 2010). Entitlement purchases began before the passage of the *Basin Plan*; on the

basis that acquiring water for the environment would be a no regrets solution. The second is an infrastructure-based approach, called the *Sustainable Rural Water-Use Infrastructure Program (SRWUIP)*, which involves investments in on- and off-farm water-use efficiency.

Policy emphasis and budget expenditure between the two water recovery approaches has shifted over time since the passage of the *Water Act* in 2007 (MDBA, 2016a). In 2015 legislation was introduced resulting in no further entitlements to be acquired for the environment (Grafton, 2016) following a bill introduced to parliament capping water entitlement purchases at 1,500GL across the MDB (Hunt et al., 2015). As of April 2017, progress towards the federal recovery targets was estimated to be approximately 2,501GL of the initial 2,750GL target. This has been achieved with an expenditure of $5.3 billion, which is around half of the $8.9 billion committed by the federal government (Grafton, 2016). At present, the current policy platform emphasises the commitment to acquiring the remaining volume of the water recovery target through on- and off-farm irrigation water-use efficiency investments.

However, there are a number of issues associated with the current policy platform. In particular, the *SRWUIP* has been heavily criticised for: being an inefficient method of acquiring water for the environment; creating on-farm technological lock-in; higher water and land use at the farm level (the Jevons Paradox, such that increasing efficiency increases consumption); higher irrigator risk and reduced adaptation ability during drought; reduced runoff and seepage to groundwater; reduced supply availability, reliability and quality for downstream and environmental water users; reduced flow dependent ecosystem services; and a lack of clarity around how saved water is converted to environmental entitlements (PC, 2010; Grafton, 2010; Qureshi et al., 2011; Adamson and Loch, 2014; Loch and Adamson, 2015; Grafton, 2016; Grafton and Williams, 2017; Adamson et al., 2017).

A possible alternative to the use of infrastructure-based reallocation methods involves an increased role for water demand-management and the judicious use of water markets and trade to acquire water for the environment, beyond just buying water entitlements[2]. A step in this direction has been taken by the CEWO and CEWH through the development and adoption of *the Environmental Water Trading Framework* (EWTF) (CEWO, 2016). The EWTF allows the CEWH to buy, sell, carry-over or use water allocations derived from environmental entitlements each year, to achieve optimal environmental outcomes in line with *Basin Plan* objectives and subject to CEWH trade conditions stipulated in the *Water Act* (2007). The challenges and potential negative implications of reallocating water though further infrastructure-based methods, coupled with the existing use of the water market by the CEWH

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[2] A water entitlement is a perpetual or ongoing water right to exclusive access to a share of water from a specified consumptive pool as defined in the relevant water plan. Water entitlements yield annual water allocations, which are the specific volume of water allocated to the water entitlement in a given season, defined according to the rules established in the relevant water plan. Water entitlements, allocations and trade are discussed further in the Section 1.1.4 and in further detail in Chapters Two and Three.

to manage water allocations, demonstrates the need for further research and consideration of additional market-based reallocation mechanisms, to acquire and manage water for positive environmental outcomes through trade.

1.1.4   MDB Water Markets

Water markets increase the allocative efficiency of scarce water resources, by facilitating the flexible allocation of water from low to high value use through trade (Howe et al., 1896; Rosegrant and Binswanger, 1994; Wheeler et al., 2014b; Grafton et al., 2015). Water markets have been implemented in a few regions around the world since the 1970s (Chong and Sunding, 2006) in various degrees of formality (Bjornlund, 2004). However, the implementation of water markets remains underwhelming internationally and the actual functions and impacts of markets are frequently misunderstood (Grafton et al., 2015). Australia's MDB provides an example of one of the most active water markets in the world (Grafton et al., 2011a) and has considerable explanatory value for understanding how water markets function.

In Australia, two water markets operate in parallel: the market for entitlements (permanent water) and the market for allocations (temporary water). Allocations and entitlements are defined in the *National Water Initiative* by COAG (2004, p. 30) as:

> Water (access) entitlement: *a perpetual or ongoing right to exclusive access to a share of water from a specified consumptive pool as defined in the relevant water plan.*

> Water allocation: *the specific volume of water allocated to the water entitlement in a given season, defined according to rules established in the relevant water plan*.

Water markets have long been an integral part of the water policy landscape and are now a central water management tool (COAG, 2004; Bjornlund, 2006). Water markets first rose to prominence in the MDB in the 1980s to deal with the growing water scarcity issue and trade was mainly localised within irrigation districts (Wheeler et al., 2014b). However, water trade was not common until further reform at the beginning of the 1990s and early 2000s, which saw the introduction of a cap on extractions in 1997 and the unbundling of water rights from land (NWC, 2004), thus allowing water entitlements and allocations to be bought and sold separately from land (Crase et al., 2015). Combined with strategic water planning and regulation, water markets in the MDB have had a number of positive impacts, specifically they have helped to: deliver positive environmental benefits, enable irrigators' adaptation to climate change, increase gross value added to farming activities; and achieve social goals (Grafton et al., 2015).

Both entitlement and allocation trade occur predominately in the sMDB. For example, in 2015-2016, 89% of surface water allocation trading and 42% of entitlement trading occurred in the sMDB (ABARES, 2017). Water trade has grown significantly over time and in 2010 it was estimated that the

majority of irrigators in the Basin have engaged in water trade in one way or another, specifically: 86% of irrigators in NSW; 77% in Victoria; and 63% in South Australia (Wheeler et al., 2014b). Among irrigators, water allocation trading is the most common form of trading and is less complex than entitlement trading. Allocation trading is often used by irrigators to mitigate their supply risk, by purchasing water when it is most needed (Zuo et al., 2014). Entitlement trade has been less prominent and entitlement sales are often a strategic decision such as farm exit or restructures (Bjornlund, 2006; Wheeler et al., 2012). As such, entitlement sales increased considerably since the federal government began acquiring entitlements.

Irrespective of the current policy emphasis on irrigation infrastructure efficiency, the *RtB* has been the government's primary market mechanism in recovering water for the environment. Given the significant challenges and risks faced by investment in on-farm efficiency, especially considering the likelihood of more frequent and severe droughts in the future, it is likely that market-based water recovery will once again become a central method of reallocating water between consumptive users and the environment within the MDB. As a result of the highly active water market in the MDB and the likelihood of further use of the water market to acquire water for the environment, the MDB is therefore a suitable case study for the exploration of using water markets to achieve positive environmental outcomes.

1.2 Literature Review: Water Markets and Environmental Outcomes

The following section provides a brief overview of key literature pertaining to the use of water markets in achieving positive environmental outcomes in the MDB, in order to highlight the current gap in the literature and contextualise the research questions.

1.2.1 Modelling Water Trade and the Environment

A considerable number of hydro-economic models have simulated water trading in the MDB (e.g. Hall et al., 1994; Heaney and Beare, 2001; Bell and Blias, 2002; Heaney et al., 2004; Qureshi et al., 2009; Jiang and Grafton, 2012; see Chapter 5 for extensive review). Overall, water trading studies have disproportionately focused on the impacts of water trade and policy change on the irrigation sector, and very few hydro-economic modelling studies have focused explicitly on potential environmental benefits from trade. From the qualitative and empirical literature, it is known that annual environmental water trade can increase irrigator engagement in reallocation programs and may improve environmental outcomes (Wheeler et al., 2013a).

Few models have sought to examine the environmental water trade directly. Key findings from existing modelling studies suggest that:

a) An EWH could implement a strategy of buying allocations when they are most needed by the environment and selling these allocations back to irrigators when they are least needed, at no overall net costs excluding administration (Kirby et al., 2006);

b) Dynamically reallocating water between consumptive users and the environment through the market can reduce environmental drought costs through the creation of pulse flows (Grafton et al., 2011b);

c) Annual allocation trade for the environment can help to minimise ecological degradation by increasing the frequency of moderate floods (Connor et al., 2013); and

d) Net benefits are greater when the CEWH actively trades water for the environment (Ancev, 2014).

However, due to the complexity of ecological modelling, large data requirements and integration challenges, the models mentioned above do not provide adequate representation of the environmental benefit achieved through annual environmental water trade. For example, no estimate of environmental impact is provided by Kirby et al. (2006). Grafton et al. (2011b) and Connor et al. (2013) use conceptual representations of drought costs and ecological degradation, respectively. However, these conceptual responses do not capture the full dynamics of the ecosystem, nor the monetary value of the ecological change. Ancev (2014) extends the previous modelling literature by considering a range of five environmental benefit functions, thus incorporating a higher level of ecological representation than previously considered. Importantly, Ancev (2014) provides an estimate of the economic value of changes in environmental condition. However, the biophysical-economic linkages are based on weak evidence linking environmental value with climactic conditions in the MDB.

The Modelling Gap

From the state of the current modelling literature two important gaps can be identified. First, there is limited understanding and inclusion of environmental dynamics, together with how they might change response to alterations in the hydrological regime caused by an annually trading EWH. Second, there is very little understanding of the economic value of these changes and how subsequent environmental benefits may compare to the cost of the water purchases required to achieve them. Where estimates do exist, the results are based on dated valuation data and tenuous biophysical-economic linkages.

As a result, there is a clear lack of understanding regarding the dynamic trade-offs faced by an actively trading environmental water holder, and how their actions may be optimised to achieve maximum environmental benefit through strategic engagement in the water allocation market. Addressing this gap

is of particular importance, given the new ability of the CEWH to actively participate in the water market and that it must act tactically to maximise ecological outcomes and minimise public expenditure.

1.2.2 Understanding Non-Government Environmental Water Holders

The models discussed above simulate the actions of an annually trading EWH with a mandate to buy and sell water for environmental benefit. However, this overlooks the non-trivial question of who, or which institutions, may act as this environmental water holder. One increasingly pertinent option is the involvement of non-government organization (NGO) environmental water holders, such as water trusts, as stewards of environmental water. Surprisingly, there is little literature examining current use of the water market to achieve environmental outcomes or how NGOs interact with the irrigation community.

Of the existing literature, some central findings regarding the role and impact of NGO EWHs demonstrate:

a) There is scope for the application the land trust model to the management of water resources to facilitate voluntary, compensated market based transfers of water to a non-government environmental steward (Neuman and Chapmann, 1999; Neuman, 2004);

b) Subsidiary style water management can provide flexible and innovative avenues for water market-based environmental water transfers, but faces limitations with regard to upstream-downstream trade-offs, financing water purchases and accountability (King, 2004; Garrick et al., 2012);

c) Non-government engagement in water management can aid in the development of inward- and outward-looking social capital in water resource systems, including establishing vertical linkages with government agencies (Hoogesteger, 2013b); and

d) Social influences, such as social trust and the maintenance of local values in water reform, are important factors influencing irrigators' acceptance and participation in water reallocation programs (Wheeler et al., 2017).

Of the studies detailed above, both Neuman and Chapman (1999) and Neuman (2004) provide a single case study that has considerable explanatory value but limited in application outside of the Oregon context. Further, both papers are written from the perspective of EWT practitioners within exiting water trusts (the Oregon Water Trust (OWT)) and read more as a narrative than a critical evaluation. Similarly, while King (2004) identifies a shift from the land trust approach, the processes by which water trusts use the water market remain largely undocumented. Further, literature examining water trusts outside of the Pacific North West context is not evident, although there is likely applicability of this approach to new cases, such as the MDB (Garrick et al., 2012; Lane-Miller et al., 2013). From that perspective, it is also largely unknown how NGO EWHs would operate in the MDB or of irrigators' willingness to engage with them further.

The EWH Gap

From the key literature detailed above, two important gaps are identified. First pertains to the current role of NGO EWHs in the western US. From the literature it is evident that studies of NGO EWHs (e.g. water trusts) has focused disproportionately on case studies in the Pacific North West, with limited attention to the growing number of water trusts in the more arid southern states, where they face a new set of challenges. It is also found that little academic understanding of the full scope in which NGO EWHs use water markets to achieve their environmental objectives, and in particular, the role of social capital and the impact of social influences on NGO facilitated environmental water transfers. Addressing this gap is particularly pertinent in current political landscape in the US, where in the absence of federal environmental leadership, non-government environmental stewards could play an increasing role in conservation through market environmentalism.

The second gaps pertains to the application of the NGO EWH approach to other context, such as the MDB, where water trust activity is in its relative infancy but there are indications of an increasing scope for private or non-profit ownership of environmental water rights. To date, a limited number of studies have examined the potential for subsidiary-style NGO EWH in the MDB[3], and in particular there is no current literature examining MDB irrigator preferences for their preferred buyer of water entitlements for the environment. Understanding irrigator preferences for the sale of environmental water and the potential role of NGO EWHs could provide an insight into potential avenues to increase irrigator participation in market-based water reallocation in the MDB.

1.3   Research Objectives and Questions

The aim of this thesis is to explore the potential benefits of using the water market to achieve positive environmental outcomes. This thesis therefore seeks to investigate how an EWH may use and expand market-based approaches of acquiring water to most effectively achieve net social benefits, using water markets to improve environmental conditions. Specifically, this thesis will address gaps in understanding pertaining to modelling an environmental response to annual water allocation trading, as well as developing a more detailed understanding of how non-government environmental water holders use water markets to achieve environmental objectives. To achieve these objectives and to fill the gaps in current understanding, a series of research questions are formulated. Overall, the research questions are designed to ask *why* environmental water markets are needed; *how* they may be used for most benefit; and by *whom*.

The first research question was designed to explore how water policy has progressed in the MDB from predominately supply-based approaches to a more pluralistic demand-based approach to water management. In particular, this question is developed to explore the limitations of predominantly supply-based approaches to water management and assess the need for a market-based approach in the

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[3] A notable exception is Garrick et al. (2012)

MDB, along with how water markets can be used as vehicles of reallocation from consumptive users to the environment. To understand MDB water policy progress and the current state of markets as reallocation mechanisms, the following questions were posed:

Research Question 1.1: How has water policy in the MDB evolved over time and what role do water markets currently play in the MDB?

Research Question 1.2: What are the limitations associated with a focus on supply-based management of water scarcity and what alternatives may be adopted?

As identified in the literature review, there are few studies which focus explicitly on enumerating the potential environmental benefits from trade or properly account for the complex dynamics of environmental systems. This question is applied to the Murrumbidgee case study, which provides an example of a hydrologically connected vegetated floodplain that may be positively impacted by improvements in the overbank flooding regime caused by trade. To investigate the potential for water markets to generate positive environmental outcomes through water allocation trade, with a particular interest in the carbon storage benefit, the following questions were posed:

Research Question 2.1: Can annual water allocation purchases made by an environmental water holder improve the condition of vegetated floodplain ecosystems?

Research Question 2.2: What is the value (if any) of the subsequent improvement in floodplain carbon storage?

If it is shown that annual water allocation purchases through the water market can generate a positive ecological and economic impact by improving the hydrological regime, which maintains the health of the floodplain ecosystem and causes subsequent improvements in carbon storage, then the following research questions are pertinent:

Research Question 3.1: Can the market value of improved floodplain carbon stocks offset the cost of annual water allocation purchases required to generate the inundation?

Research Question 3.2: What are the policy implications of the above findings from the perspectives of a self-financing environmental water holder?

The following research questions differ from the question of how water markets are used to achieve benefit, and instead focus on the question of whom or which institutions currently use the water market to achieve environmental outcomes. As highlighted previously, little is known about the role of NGOs and their use of the market. The questions are addressed through an examination of water trusts in the western US, where the involvement of NGOs are comparatively active compared to the MDB. This thesis is also interested in distilling these key findings from research undertaken in the western US to apply to the MDB. To explore this idea the following questions were posed:

Research Question 4.1: How do NGOs, such as water trusts, use the water market to achieve their environmental objectives in the western US?

Research Question 4.2: What insights for the MDB can be identified through various NGO EWH actions and strategies?

Having explored how NGOs use the water market to achieve positive environmental outcomes, it is useful to understand irrigators' willingness to be further involved with NGO EWHs, such as water trusts. This question particularly focuses on irrigators' preferences, rather than other water users, because irrigators' are the largest owners of consumptive water rights in the sMDB. To understand irrigators' preferences for engaging with NGO EWHs, or otherwise, the following questions were posed:

Research Question 5.1: Who would MDB irrigators prefer to sell their water to if given the choice between local, state, federal governments and non-government environmental organizations?

Research Question 5.2: What implications do the findings from the above research question have for how water buy-back programs can be structured?

1.4   Thesis Methodology and Research Design

To answer the research questions posed above and to address the gaps identified in the literature, a mixed-methods inter-disciplinary research approach was used in this thesis. Mixed-methods research (MMR) is a formal research tradition which involves mixing quantitative and qualitative techniques, approaches and concepts into a single study (Creswell, 1998; Creswell, 2003). The use of MMR as a formal research methodology is motivated by a need to pragmatically fit together the insights gained by both quantitative and qualitative philosophical approaches to address complex or interacting research problems (Teddlie and Tashakkori, 2013; Creswell, 2003; Johnson and Onwuegbuzie, 2004). MMR has arisen as the third methodological movement in the social sciences (Teddlie and Tashakkori, 2013) and is based on the assumption that when addressing complex research questions, qualitative and quantitative styles are most valuable when combined (Neuman, 2000). MMR is particularly well suited to water resource research because of the duality of legitimate perspectives pertaining to water-use and management, as well as the complex characteristics of water as a common resource and an economic commodity.

The MMR approach used in this thesis combines two primary methodologies: an integrated hydro-economic simulation model (quantitative) and a series of semi-structured interviews with environmental water stakeholders supported by survey analysis (qualitative). These two primary methodologies are contextualised by a range of literature reviews, policy analyses and case studies. The overall research design is depicted in Figure 1.1 and the key aspects of each approach are further discussed below.

Figure 1.1 Research Methodology and Design

	Qualitative
Methods	Policy analysis, literature review, case study
Outputs	Chapter 2: Water resources and policy reform in the MDB / Chapter 3: Rebalancing the system through trade / Chapter 4: A century of intervention in the CLLMM
	Quantitative
Methods	Literature review, hydro-economic modelling
Outputs	Chapter 5: Hydro-economic modelling literature review / Chapter 6: Modelling theory, data and methods / Chapter 7: Carbon and water market opportunities
	Qualitative/Quantitative
Methods	Qualitative Interviews, survey analysis
Outputs	Chapter 8: The role of NGO EWHs and applications to the MDB

Source: Own Figure.

1.4.1 Quantitative Methodology: Hydro-economic Simulation Modelling

A highly integrated dynamic hydro-economic model (HEM) was developed to simulate the decisions of an EWH with a mandate to improve floodplain health through the trade of annual water allocations. The model was developed to investigate the value of carbon credits stored in native floodplain vegetation, which can be augmented by changes to the overbank flooding regime caused by an annually trading EWH. In particular, the methodology applied sought to determine if there was a possibility for a self-financing EWH to offset the cost of water allocation purchases through the generation and sale of carbon credits stored in natively vegetated and hydrologically connected floodplain wetland.

To simulate this context, a series of five sub-models were developed and integrated into a single HEM. These are: a) a catchment water balance; b) a water price model; c) a floodplain carbon storage dynamics model; d) a carbon valuation model; and e) a dynamic water reallocation algorithm. Methodologically, the HEM is a dynamic simulation model which simulates the reallocation of annual water allocations

to the environment. Using this approach, the model identifies volumes to be reallocated to the environment in years which additional water allocations can generate carbon credits (as a result of forest biomass growth) of market value equal or in excess of the cost of the supplementary water purchases. The HEM follows and significantly extends the methodological approach established by Connor et al. (2013). A major methodological contribution to this approach is the development of the detailed four-state model of carbon stock decay and growth dynamics and valuation which is used to inform EWH reallocation decision-making. The model is coded in the General Algebraic Modelling System (GAMS) and is applied to the case study of the Lower Murrumbidgee wetlands. An overview of the HEM is shown diagrammatically in Figure 1.2.

Figure 1.2 Quantitative Methodology: Hydro-economic Simulation Model
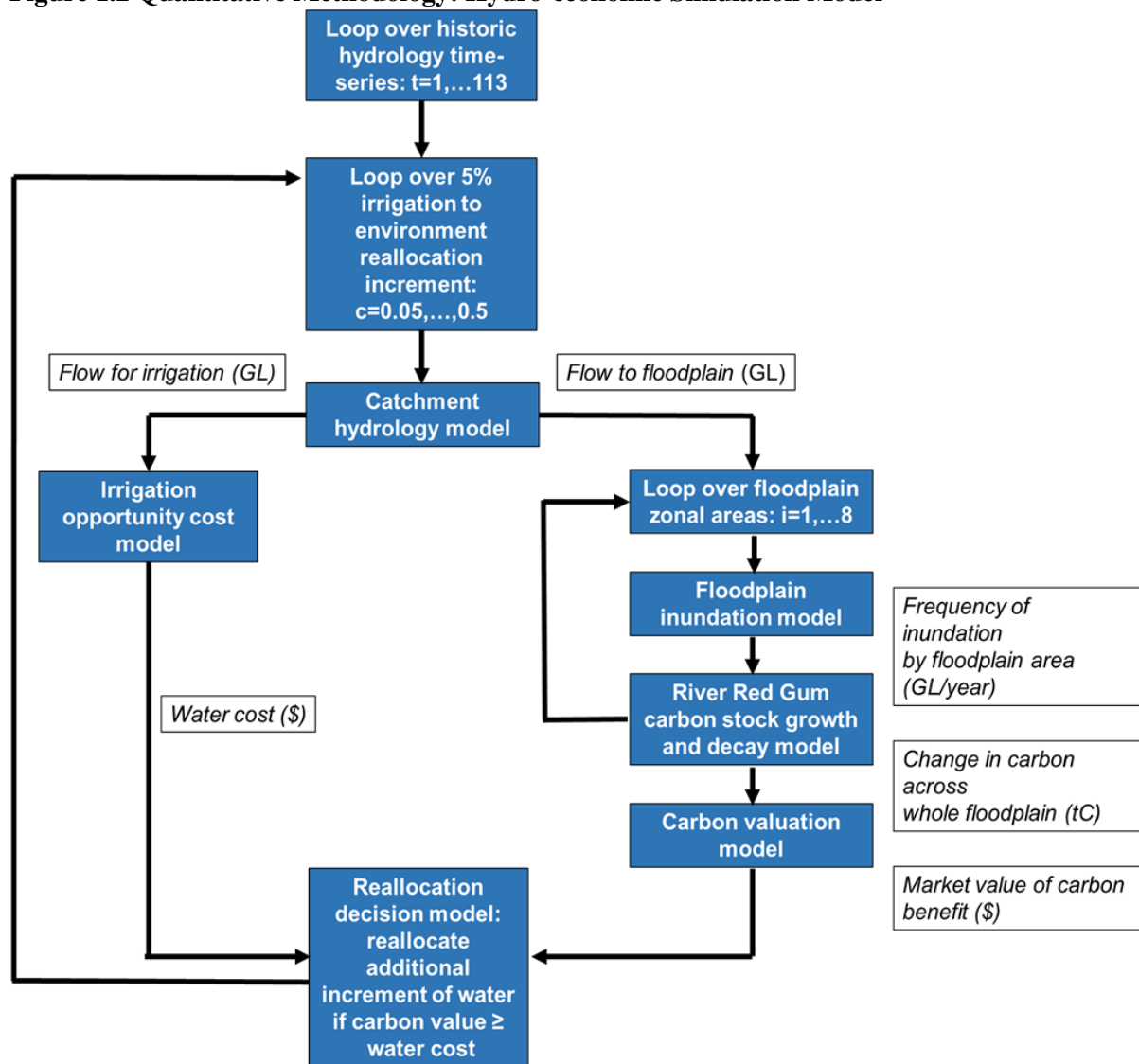
Source: Own Figure.

1.4.2 Qualitative Methodology: Semi-Structured Stakeholder Interviews and Survey Analysis

Forty-nine qualitative, semi-structured and face-to-face interviews were conducted with environmental water resource stakeholders in the western US. In particular, participants were purposively sampled from the non-government sector to understand the role played by non-government organizations as stewards of environmental water, and their use of the water market and water purchase strategies to acquire rights. Because this approach sought to fill the gap of understanding NGO engagement in the environmental water market, the selection criteria was developed around participant knowledge and experience rather than sampling from one particular basin. Additional interviews were also conducted with environmental water transfer (EWT) practitioners from the state level.

The interviews were fully recorded and transcribed. Interview transcripts were thematically analysed using NVivo, which is a computer-aided qualitative data analysis software program. The interview data was analysed through the lens of subsidiary theory, which describes the principle that actions should be taken at the lowest level of governance at which a particular objective can be adequately achieved (Bermann 1994). Informed by an extensive literature review, the thematic analysis specifically examined NGO acquisition strategies and the use of innovative environmental water transfer tools, the role of social capital in facilitating successful transfers, self-financing arrangements, as well as challenges faced by NGOs in the water market, including financial sustainability, institutional inertia and difficulty in demonstrating benefits.

Because the thesis is primarily concerned with the case of the MDB, insights from the qualitative interviews in the US were distilled and examined in the MDB case study. Drawing insights from international cases in water management builds on the academic tradition of comparative water policy assessments (Wescoat, 2005; Wescoat, 2009) and in particular the comparative discussions between the Australian and western US cases (Garrick and Bark, 2011; Grafton et al., 2012; Garrick, 2015).

In particular, this aspect of the methodology sought to understand the preference of MDB irrigators towards sales to non-government environmental organizations, such as the water trusts active in the western US, relative to their preference of water sales to the federal and state governments. To understand this, this question was asked as part of a large-scale telephone survey of 1,000 MDB irrigators. The survey results were analysed using basic statistical methods to examine irrigator preferences. An overview of the qualitative approach is demonstrated in Figure 1.3.

Figure 1.3 Qualitative Methodology: Stakeholder Interviews and Survey Analysis

Qualitative Interviews	Survey Analysis
Western USA	*MDB Australia*

Explore how NGO EWHs use the water market for environmental outcomes. → Develop applicable insights for the sMDB context. → Understand sMDB irrigator receptiveness to NGO EWHs as buyers of environmental water.
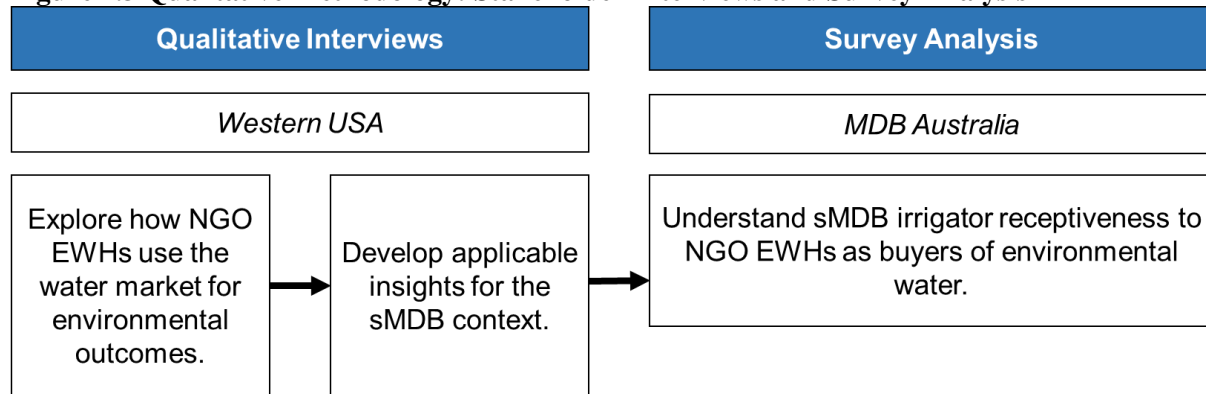
<u>Source:</u> **Own Figure.**

1.5   Thesis Structure

The background, methods and results of this thesis are organised into nine chapters containing published and unpublished work. Following the introduction, Chapter 2 provides an expanded version of the published work:

> **Settre, C.** & Wheeler, S. (2016) Environmental water governance in Australia: the movement from regulation and engineering to economic-based instruments. In: V. Ramiah and G. Gregoriou (eds), *Handbook of Environmental and Sustainable Finance*, New York: Elsevier, pp. 67-91.

This chapter examines the transition of water policy in the MDB from a predominately engineering based management approach to a pluralistic water management paradigm incorporating state, market and civil sector. Importantly, it details the reform process that has led to the widespread use of water markets as a tool to manage water scarcity and concludes with the current state of policy in the MDB. Chapter 2 is intended to provide background to the MDB case study and establish a policy foundation for subsequent analysis.

Chapter 3 presents an expanded version of the published work:

> **Settre, C.** & Wheeler, S. (2017) Rebalancing the system: acquiring water and trade. In: A. Horn, A. Webb, M. Stewardson, B. Richter, B., M. Acreman (eds), *Water for the Environment: From Policy and Science to Implementation and Management*. London: Elsevier, pp. 399-419.

Research in this chapter is also included from:

> Loch, A., Wheeler, S., **Settre, C**. (*forthcoming*) Private transaction costs of water trade in the Murray-Darling Basin. *Ecological Economics.*

This chapter reviews the current state of water markets in the MDB and the use of markets as a tool to reallocate water to the environment. The rationale of market-based reallocation over infrastructure-

based conservation is presented. Two international case studies of environmental water markets are presented before extensive exploration of the MDB case. Advancements and challenges of water entitlement recovery for the environment are presented. Chapter 3 therefore provides a detailed overview of water markets in the MDB and their use as vehicles of reallocation, plus highlights the need for further research and policy development.

Chapter 4 presents the published work:

> **Settre, C.** & Wheeler, S. (2017). A century of interventions in a Ramsar wetland – the case of the Coorong, Lower Lakes and Murray Mouth, *Australasian Journal of Environmental Management*, 24(2), pp. 1-21.

This chapter continues the examination of MDB water policy, with a particular focus on the historical use and current limitations of supply-based approaches to managing water scarcity and ecological degradation caused by over-allocation and drought. This chapter focuses on the case of the Coorong, Lower Lakes, and Murray Mouth (CLLMM), which is located at the terminus of the MDB and is an internationally important and nationally significant wetland complex. This chapter is intended to provide a detailed exploration of MDB policy through the examination of a single case study.

Chapter 5 presents an altered and expanded version of the published work:

> **Settre, C.,** Connor, J. & Wheeler, S. (2017) Reviewing the treatment of uncertainty in hydro-economic modelling in the Murray-Darling Basin. *Water Economics and Policy,* 3(3), pp.35, DOI: 10.1142/S238264X16500429

This chapter provides a systematic review of the state of hydro-economic modelling in the MDB to identify gaps in the literature and possible avenues for advancement. It specifically highlights the need for a greater emphasis on ecologically focused modelling when considering the benefits and costs of reallocation, as well as greater representation of ecological dynamics and more robust attention to the treatment of uncertainty. This chapter is intended to provide a context for the methods and results of the hydro-economic model presented in the subsequent chapters. This publication is also included in Chapter 6.

Chapter 6 presents the supporting theory and methodology used to develop the hydro-economic dynamic simulation model. This chapter details the data used, assumptions, modelling approach and final model specifications. The novelty of this modelling approach and methodological contribution is also detailed therein. This chapter demonstrates how the model is developed and provides context to understanding how the subsequent results were obtained.

Chapter 7 presents two working papers:

Settre, C., Connor, J. & Wheeler, S. (2017) Can the generation and sale of carbon credits in natively vegetated floodplains offset the costs of annual environmental water purchases?

Settre, C., Connor, J. & Wheeler, S. (2017) Strategic water and carbon market opportunities for a self-financed environment water holder to improve floodplain forest carbon storage.

This chapter presents the results of the hydro-economic model. The results indicate the ability for an annually trading EWH to contribute positive environmental outcomes through the purchase of annual water allocations and suggests there may be a persuasive business case for a self-financed EWH to offset the cost of water allocation purchases, through the strategic generation and sale of carbon credits stored in hydrologically connected vegetated floodplains. The hydrological and economic conditions under which this strategy may be viable are explained. The reallocation strategy determined using the HEM results are also compared to previous model results, to demonstrate the usefulness and value of including more representative models of environmental dynamics.

Chapter 8 presents the working paper:

Settre, C. (2017) Localism, community engagement and 40 Cups of coffee: the role of non-government organisations as stewards of environmental water in the western USA and potential insights for the Murray-Darling Basin, Australia.

This paper presents the results of the qualitative interviews undertaken in the western US and MDB survey analysis to answer Research Questions Four and Five. This chapter provides a brief overview of the methodology used in these approaches followed by the results of the qualitative analysis. It is shown that non-governmental organizations use and expand existing water markets to achieve environmental outcomes in unique and under-analysed ways. A series of insights from the interviews are presented which are relevant to the MDB. Survey results of MDB irrigators are also presented, demonstrating that MDB irrigators have clear preferences for water sales, which are broadly in line with the principles of localism.

The thesis concludes in Chapter 9 which summarises the thesis findings, contributions to the literature, limitations and future work. Importantly, a series of policy recommendations and implications are presented based on the key thesis. The thesis is followed by a series of appendices which present additional results and information. In particular, Appendix A extends the future work identified in Chapter 9 with regards to the inclusion of additional ecosystem service values in the hydro-economic model.

CHAPTER 2: Water Resources and Policy Reform in the Murray-Darling Basin

This chapter is an expanded version of a book chapter published in the *Handbook of Environmental and Sustainable Finance* (2016) which investigated the changing paradigms evident in Australian water management. The original chapter was written as an overview of water reform policy in the MDB to demonstrate the transition from a predominately supply-led water management approach to a mature water economy, in which water markets are employed as a central water management tool. The published book chapter is expanded to include a summary of the state of global freshwater water resources and water scarcity, as well as a detailed description of the MDB hydrology, climate, water and land uses, and a more detailed examination of the water reform process. The expanded version of this chapter also includes a greater discussion of the methods and current progress of acquiring water to meet the *Basin Plan* objectives.

Overall, this chapter is intended to introduce the MDB physical and political characteristics and situate the MDB water scarcity challenges within an international context, as well as providing an overview of the history which has led to the current policy status in the MDB. This chapter is therefore important for providing context for the thesis findings and subsequent policy recommendations.

Publication Details

Publication Title	Environmental water governance in Australia: the movement from regulation and engineering to economic-based instruments
Publication Status	☒Published ☐Accepted for publication ☐Submitted for publication ☐Unpublished and unsubmitted work written in manuscript style
Publication Details	Settre, C. & Wheeler, S. (2016) Environmental water governance in Australia: the movement from regulation and engineering to economic-based instruments. In: V. Ramiah and G. Gregoriou (eds), *Handbook of Environmental and Sustainable Finance,* New York: Elsevier, pp. 67-91.

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Name of Principal Author (Candidate)	Claire Settre		
Contribution to the Paper	Formation and idea of paper, literature review, analysis, writing and editing.		
Overall percentage (%)	80%		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	28/8/2017

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

- The candidate's stated contribution to the publication is accurate (as detailed above);

- permission is granted for the candidate in include the publication in the thesis; and

- the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Sarah Wheeler		
Contribution to the Paper	Content guidance, discussion and editing.		
Signature		Date	21/8/17

2.1 Introduction

Many arid and semi-arid regions around the world are faced with the challenge of allocating scarce water between competing demands. Tensions between consumptive and environmental demands are expected to increase as per capita water availability decreases owing to climate change, population growth and economic development (Grafton et al., 2013). Meeting water user demands has historically been managed through top-down control of water resources, including investments in large scale water supply infrastructure (Bakker, 2005). However, supply-side management coupled with generous water rights allocations for consumptive use has rarely given due account for environmental requirements of flow-dependent ecosystems (Quiggin, 2004). This has resulted in considerable degradation of freshwater environmental assets and the services they provide (Abromovitz, 1996; Kingsford, 2000; Dudgeon et al., 2006; Davidson, 2014).

Currently and looking forward, the challenges of achieving efficient and equitable allocation of scarce resources (Howe, 2000) are compounded by the non-uniform change in water supply patterns due to climate change (IPCC, 2014), as well as complex institutional reform required to formally recognise the environment as a legitimate and beneficial water user (Garrick et al., 2011). In light of these obstacles, it is clear that traditional methods of water governance are not wholly sufficient to manage scarce water resources in an era of growing demands, shrinking supplies and increasing uncertainty.

The MDB in Australia represents a unique case study, which has undergone a transition from command-and-control management to a more pluralistic governance structure that incorporates the state, market, and civil sector. Within this transition, the gradual inclusion of economic-based instruments is evident, to a point where the water market is now a water management tool employed by irrigators, governments and environmental water holders (EWHs) within the Basin (Grafton and Horne, 2014; Wheeler et al., 2014b). The process of MDB reform exhibits qualities of adaptive market-based governance, such that the institutional arrangements have evolved to satisfy the needs of the community in an environment of change (Hatfield-Dodds and Nelson, 2007). Overall, MDB policy reform illustrates the transition from being predominantly based on regulation and engineering solutions, to a movement toward economic-based instruments as water governance management tools.

This chapter first introduces the global state of water scarcity in order to situate the MDB example within an international context. A detailed overview of the MDB physical and hydrological conditions are explained before exploring the progress of MDB institutional reform, specifically focussing on reform aspects which have led to the development of water markets and the recognition of environmental flows within the Basin.

2.2 The Characteristics of Water

2.2.1   Water as an Economic Good

Water is commonly spoken of as a single homogenous resource (Hanemann, 2006). However, water and its value is characterised by a number of physical attributes including its location, timing, reliability, quality, and volume. These attributes have implications for how water is managed and governed in society. Water can also be thought of and characterised as an economic good. Water as an economic good is institutionalised internationally by the *Dublin Principles* (1992) which states that: w*ater has an economic value in all its competing uses and should be recognised as an economic good (Guiding Dublin Principle #3)* (GWP, 1996, p.1)*.*

Key characteristics of water as an economic good include: a) water as a private and public good; b) the mobility of water; c) the cost of water; d) the price of water; e) the essentialness of water; f) the heterogeneity of water; g) the importance of marginal value not average value of water; and h) the multiple benefits of water (Hanemann, 2006). While each characteristic may not be particular to water (i.e. it may also apply to land, air or food), the combination of economic and physical attributes applicable to water make it a unique and complex commodity (Savenije, 2002).

Water is essential to all aspects of human endeavours and is vital for sustaining human life and the health of flow dependent ecosystems. The essentialness of freshwater is defined economically by the fact that no amount of any other resource can compensate for having zero amount of water (Hanemann, 2006). That is to say, water is non-substitutable in most of its uses. In addition, freshwater is a scarce and finite (but renewable) resource, such that the total amount available for use is capped and cannot readily be increased. The combination of the essentialness and finite nature of freshwater, place its management at the forefront of importance in economics and policy.

At the centre of the economic conception of water is its dual character as both a private and public good. For example, water can be classified as a private good (e.g domestic drinking supply) or as a public good from which shared benefit is derived (e.g. environmental watering). The conception of water as a public good implies water users do not compete for use (non-rivalrous) and use by one user does not prohibit use by another user (non-excludable) (Grafton and Wheeler, 2015). On the other hand, water as a private good implies competition between water users (rivalrous) who can be excluded from use by other users (excludable). The characteristics of water as a public good can result in the undervaluing and undersupply of water (Grafton and Wheeler, 2015).

An additional characteristic of water is that it is a flow/flux resource which is both temporally and spatially variable in terms of volume (Hanemann, 2006). Water quality may also vary due to the type and location of the resource (e.g. groundwater or surface water) or due to anthropogenic factors such as pollution and use. The mobility of water as a flow resource means that, unless water is captured and stored, it will escape (Savenije, 2002). The varied distribution and mobility of water necessitates its

storage and transport, to secure supply and meet consumptive human supplies in different times and places.

However, an additional characteristic of water is that it is bulky, capital-intensive and expensive to transport relative to its weight (Savenije, 2002; Hanemann, 2006), thus making the provision of water a costly undertaking relative to the provision of other resources. The price of water is a subject of ongoing debate in many academic and government circles (Savenije, 2002). The price of water typically represents the cost of supply provision rather than the scarcity value of water (Hanemann 2006). For this brief discussion, it is sufficient to say that water should be priced such that the marginal net benefit is equal across the competing alternative uses of water (Grafton and Wheeler, 2015), although there are a number of alternative approaches to water pricing being implemented internationally with varying degrees of success.

2.2.2 The Value of Water

Water is unique in the types of services it provides to society (Booker et al., 2012) and a special characteristic of water is the multiple and sequential benefits which can be obtained from its use. The concept of 'value' awarded to water and the environment differs between research traditions (Brown, 1984) and between different community groups with different world views (Syme and Hattfield-Doods, 2007). When considering freshwater value, ecological sciences are often concerned with the intrinsic value of an ecosystem existence. This is exemplified in the philosophy of *deep ecology* (Naess and Sessions, 1984) which identifies the inherent value of the environment independent of the economic benefits generated through its use. Conversely, the economic approach takes an anthropocentric perspective that focuses on the utility of the environment for individuals and society. The difference between these two approaches can be thought of as *intrinsic* and *instrumental* concepts of value (Tietenberg and Lewis, 2014). From an economic perspective, the total economic value (TEV) of the environment or its sub-components (e.g. freshwater) is given by the sum of all use and non-use values (Krutilla, 1967). The TEV of water is shown in Figure 2.1 and is discussed below.

Figure 2.1 The Total Economic Value of Water

Total Economic Value

Use Values — Non-use values

| Direct Use value | Indirect Use Value | Option Value | Quasi-option Value | Bequest Value | Existence Value |

Outputs & services that can be consumed directly

Functional benefits enjoyed indirectly

Future direct and indirect use

Expected new information from avoiding irreversible losses from:

Value of leaving use and non-use values to future generations

Value from knowledge of continued existence

Extractive: water for regional communities, pastoral and mining
Non-extractive: Tourism and recreation, aesthetic

Positive: Research and education, indirect support for regional communities
Negative: Greenhouse gas emissions released with water extraction

-Species
-Habitat
-Ecosystems
-Groundwater reserves

- Icon
- Species
- Habitat
- 'Way of life' connected to traditional uses
- Groundwater reserves

- Icon
-Species
- Habitat
- Ecosystems
- Groundwater reserves

Source: Rolfe (2008), p. 18.

Use values are derived from direct, indirect or consumption use. Direct use values describe the values that can be obtained by the consumption of environmental goods and services, such as food, fuel and freshwater. Indirect use values are derived from functional benefit that are indirectly used, such as the carbon sequestration and storage service provided by trees. Non-use values, also 'passive use' value (Randall 1993), describe the value attributed to an environmental asset despite never using it. One aspect of non-use values is existence value which defines an individual's willingness to pay (or avoid charge) for the continued existence of a resource, despite no intention of using it (Madaraga and McConnell, 1987). This is the value associated with the knowledge that an environmental asset is protected from degradation (Walsh et al., 1984). Another aspect of non-use value is bequest value, which is the value derived from the knowledge that future generations will be endowed with resources and a healthy environment (Walsh et al., 1984). Existence and bequest values are sometimes considered altruistic values, describing the value gained from knowing other people have access to a resource (Pascual et al., 2010). In between use and non-use values are option and quasi-option values, which represent the value assigned to the possibility of future use and non-use of a resource, or use that may occur if new information is made available regarding the use of the environment (Rolfe, 2008).

2.3 The State of Global Water Resources

Water is essential for all forms of life and a vital resource for social progress and economic development. However, freshwater is spatially and temporally distributed around the globe, in ways that are politically and physically challenging, to supply water for life and development (Hanemann, 2006). The following sections discuss the global distribution of freshwater scarcity, types of water user groups, the future supply of freshwater, and the relationship between global water resource development and freshwater ecosystem degradation.

2.3.1   Global Water Distribution and Scarcity

Freshwater makes up a fractional amount (~3%) of global water volumes on Earth (Shiklomanov, 1993). Approximately one-third of freshwater is stored underground (Gleick, 1996) and the majority (~68%) is stored in glaciers and ice-sheets (Shiklomanov, 1993), albeit melting at a significant rate (IPCC, 2014). The remaining freshwater is stored in rivers, lakes and swamps (~1.2%). In addition to accessible groundwater supplies, this volume represents the total volume of freshwater available to society and the natural environment.

Global water distribution and supply availability is governed by the hydrological cycle and is modulated by natural climactic and anthropogenic drivers. These drivers include annual and decadal climactic patterns (e.g. the seasons, El Nino), stochastic variability in rainfall and runoff, changes in supply due to climate change, and human activities including large-scale water storage and transport (e.g. inter-basin transfers). As a result, water availability is unevenly spatially and temporally distributed across the globe in a way which can result in a mismatch between the variability of supply and the relatively constant (although growing) nature of consumptive demands (Hanemann, 2006). Increasing water demands have directly resulted in reduced outflows and placed considerable pressure on freshwater resources in a number of basins around the world (Grafton et al., 2013). When the demand for water extractions exceeds the supply or availability of freshwater, water scarcity is realised.

Water scarcity is a multi-faceted issue, but can be broadly thought of under two definitions: physical (absolute) water scarcity and economic water scarcity. Physical water scarcity occurs where the available freshwater supplies are not of sufficient quantity or quality to meet the aggregate freshwater demand within the region (FAO, 2015). Economic or social water scarcity occurs when institutional and financial capital, or unbalanced power relations and related inequalities, limits the access to freshwater even though supply may exceed demand (Falkenmark et al., 2007). Water scarcity is relative (e.g. relative to demand, such that dry regions are not necessarily scarce if demand is low) and naturally dynamic based on the temporal nature of supply (FAO, 2015). Mekonnen and Hoekstra (2016) found that four billion people live in conditions of extreme water scarcity for at least one month per consecutive year, and that half a billion people live in a constant state of water scarcity. Recently, freshwater scarcity has been increasingly understood as a systematic risk to the global community. For

example, water crises were listed as the world's top global risk by impact in 2015 (WEF, 2015), and third highest risk in 2016 – superseded by failure of climate change mitigation and adaptation (which is also water related) and weapons of mass destruction (WEF, 2016).

Figure 2.2 shows the average annual distribution of water scarcity across the globe from 1996 to 2005, measured as a ratio of the net water withdrawals to the total freshwater availability (also referred to as 'blue water' scarcity). High water scarcity appears in areas of high population density and/or high irrigation water demand and low resource availability (Mekonnen and Hoekstra, 2016). Figure 2.2 captures the extreme drought conditions experienced in Australia from 1997 to 2010, which is discussed in Section 2.6.2 and a number of thesis chapters. The figure also highlights the extreme scarcity faced by the majority of western and central plains of the US, which provides the case study utilised in Chapters 3 and 8.

Figure 2.2 Average Annual Blue Water Scarcity from 1996-2005
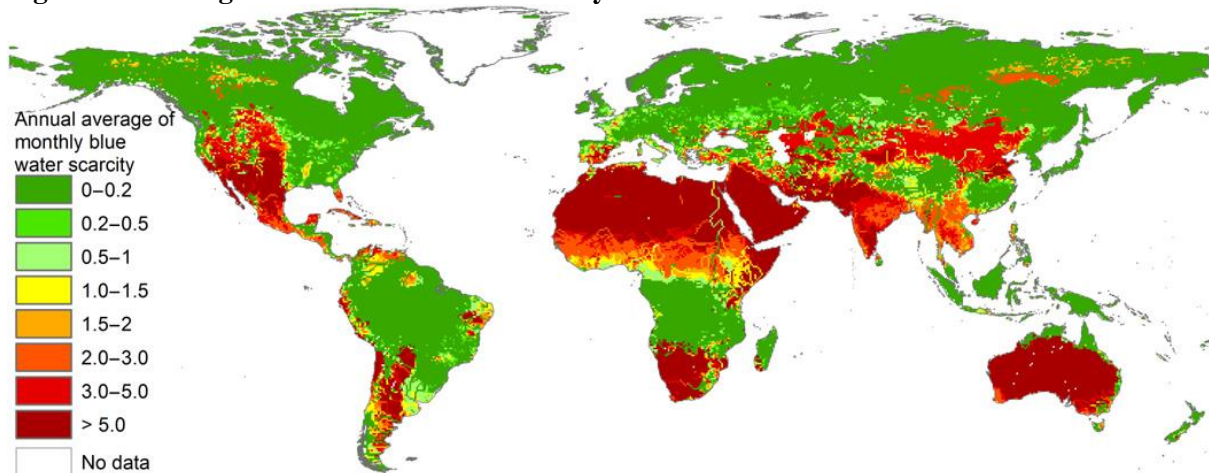
<u>Source:</u> **Mekonnen and Hoekstra (2016), p. 3.**

2.3.2   Water Users

Freshwater is the only resource which links all sectors of human industry and the natural environment. Because water is used in all aspects of human development, there are a number of types of water end-users whose agendas may not necessarily be compatible. In water scarce regions, consumptive and non-consumptive water users compete at the trans-boundary, sectorial and individual level for the use of scarce water resources (Ziolkowska and Peterson, 2016).

Water users can be classified as: consumptive human use, non-consumptive human use; and environmental use. Consumptive water-use refers to the use of water which results in the diminishment (either quality or quantity) of the resource though processes such as evaporation, evapotranspiration, transformation and pollution. Consumptive water-use rarely results in the full diminishment of the volume abstracted, but rather augments the location and timing of water availability (e.g. agriculture is a consumptive use, but some water is returned to the river through runoff or return flows). Non-

consumptive water-use are activities which do not result in the transformation or diminishment of the resource and instead make use of the water *in-situ.*

The environment can be classified as a water user when legislation recognises the legitimacy and beneficial nature of environmental water-use (Garrick et al., 2011). Environmental water-use describes the water used by the environment to maintain ecological integrity and resilience. Depending on the use, environmental water-use can be consumptive (e.g. water evapo-transpired through a floodplain forest after flooding) or non-consumptive (e.g. in-situ channel use to maintain fish habitat).

In these above descriptions, non-water users (e.g. conservation or amenity stakeholders) are not classified as water users, but may hold non-use values such as existence, bequest and options values (Rolfe, 2008) for water resources. Non-use stakeholders are described more fully in Chapter 4, which looks, in part, at the types of classification of wetland stakeholders at the terminus wetland of the MDB. In many cases of water users, there may be overlap between the type of end-uses, specifically environmental use, non-consumptive human uses and non-use values.

2.3.3 The Future Supply and Demand of Freshwater

Global climate change affects a number of aspects of the hydrological cycle, including the volume, variability, quality, and location of freshwater supply, as well as the extremity of water-related natural disasters (e.g. floods and droughts) (Kundzewicz, 2008). These impacts directly affect the volume of fresh surface water available for human and ecosystem use (van Vliet et al., 2013). Evidence suggests that climate change results in a general intensification of the hydrological cycle, including higher rates of evaporation due to temperature increases in some areas, and higher rates of precipitation and flooding events in other areas (Kundzewicz, 2008). In arid and semi-arid regions, climate change is likely to result in a reduction in rainfall and freshwater availability (Ragab and Prudhomme, 2002; Sivakumar and Brunini, 2005). For example, Ragab and Prudhomme (2002) show that very arid regions including North Africa, Egypt, Saudi Arabia, Syria, Jordan and Israel are the hardest hit by reductions in rainfall and may experience a decrease in mean historic rainfall volumes of up to 20-25% in 2050.

As well as reductions in supply, patterns of water demands are also likely to change in the future. Vörösmarty et al. (2000) argue that rising water demands outweigh the impact of climate change on future water scarcity. The demand for water resources is increasing due to increases in global populations, projected to increase to 8.3 billion in 2030 (UNESCO, 2012). Occurring parallel to an increase in population is an increase in the standard of living, resulting in correspondingly higher levels of life expectancy and changes in dietary patterns (Easterlin, 2000) – effectively resulting in the overall consumption of resources in higher intensity for a longer period.

Overall, changes in the supply and demand of water resources can induce rapid and uncertain changes in the food, water, energy and environment nexus. The management of the current complexities and

future challenges of water resource management therefore requires the development of robust and flexible institutions which ensure water is used to generate the maximum net social benefit though high value use.

2.4 Water Resource Development and Ecological Degradation

The development of water resources throughout the 20th century has enabled the rapid expansion of agricultural and industrial society and has benefited billions of people through the supply of potable water (Gleick, 2003). However, water resource development for irrigation and urban supply has typically proceeded with little to no consideration of environmental water requirements and has been implemented at the cost of considerable adverse ecological impacts (Gleick, 2003) such as the degradation and loss of freshwater biomes (Abromovitz 1996; Dudgeon et al., 2006; Davidson, 2014)

Water resource development changes the magnitude and variability of the natural flooding/drying regimes due to river channel construction (e.g. dams, weirs) and high abstractive uses of water (e.g. irrigation) (Kingsford, 2000). River flows of adequate volume and variability are the primary drivers of ecological processes that maintain the health and resilience of rivers, wetlands and floodplains (Junk et al., 1989; Poff et al., 1997; Overton et al., 2009). As a result of altered flow characteristics of the natural flow regime in developed basins, the health of flow dependent ecosystems have suffered severe, widespread and ongoing degradation on a global scale (Poff et al., 1997; Kingsford, 2000; Davidson, 2014). In particular, development of rivers has damaged the health and resilience of floodplains and wetlands which have been subject to ongoing conversion and reduced resilience. For example, it is estimated that the world has lost between 64-71% of total wetland expanse since 1900AD due to water resource development, land-use change and anthropogenic drivers (Davidson, 2014).

Historically, the protection of flow-dependent environments (in cases where protective laws exist) has focused upon the provision of a base-flow to maintain a minimum volume of water in the river (Poff et al., 1997). However, base-flow volumes are often below the volumes required to maintain adequate ecological function generated by flood variability and inundation (Junk et al., 1989; Poff et al., 1997). In recent decades, improved scientific understanding of environmental watering requirements, environmental awareness of the local and global community and the recognition of the environment as a legitimate and beneficial water user (Garrick et al., 2011) (in addition to a number of other factors discussed in detail in Section 2.7.4) has led to the improved legislation and water management practices which account for environmental watering needs. These practices are specifically evident in case studies within Australia's MDB and in some states of the western US, which are discussed further throughout this thesis. Further discussion of the ecological impact of water resource development, the current ecological condition, and the impact of environmental water markets in the MDB is provided in Chapter 3.

2.5 Water Resource Management Strategies

Broadly speaking, water institutions around the world have exhibited clear common trajectories (Saleth and Dinar, 2000; Meinzen-Dick, 2007). This common trajectory centres on the transition from a command-and-control of water resources involving major reliance on supply measures, towards decentralised management approaches incorporating economic-based instruments as demand-focused allocation tools (Saleth and Dinar, 2000). Examples of supply- and demand- management strategies are provided below.

Supply strategies: a) water infrastructure and network systems for water management and transport; b) technological solutions, such as improved water and land management approaches and efficiency improvement measures.

Traditionally, governments have tended to rely on supply strategies to address water policy issues. This approach has been (and continues to be implemented in some countries with rapidly growing urban populations, such as China) implemented to provide subsidised water provision to urban centres and agricultural communities (Hearne, 1998). In a supply-based water management paradigm, water infrastructure is typically owned and operated by the state, who also specify legitimate water users, their water allocation, and the charge for water (Thobanl, 1997). While the development of supply infrastructure has benefited millions of people through the provision of potable water (Gleick, 2003), it has also contributed to the limited flexibility of water allocation mechanisms due to the physical, technical and economic characteristics of large-scale state-owned water supply infrastructure (Rosegrant et al., 1995).

However, the increasing cost of supply provision (e.g. inexpensive locations for dams mostly exhausted), combined with fiscal government limitations, society's changing attitudes in environmental protection and the need for an effective response to water scarcity issues (Bjornlund, 2003) have prompted the transition to alternative or pluralistic paradigms of water management. Since the 1980s, there is a growing recognition that economic instruments should be used to regulate the access and use of water resources. Examples of demand-based management include:

Demand strategies: a) voluntary measures (e.g. change through education); b) regulation and planning requirements (e.g. restrictions on fertilizers and chemicals applied near watercourses, and water meters); c) economic instruments (e.g. water markets, prices, taxes, and subsidies for low-income households' water bills).

Demand-focused water management incorporates the use of economic-based instruments as additional tools to (re)allocate water and is a central pillar of integrated water resource management (IWRM). Demand-based policies typically involve water pricing (e.g. full cost recovery), user-tariffs, as well as water markets for irrigation water and environmental water. Demand-focused policies were widely

advocated by international organizations such as the World Bank and the United Nations resulting in numerous market-based reforms undertaken in the late 20th century (Bjornlund and McKay, 2002). Three central approaches to demand-based management are described in more depth in Table 2.1. In some cases, the lack of success of voluntary and regulation approaches in water management has meant there has been an increased emphasis given to economic instruments over time.

Table 2.1 Demand Management Tools for Freshwater Resources

Demand Instrument	Description	Example
Regulations and Sanctions (command and control)	This approach relies on restrictions of water-use or behaviour	Norms and standards for water quality (e.g. drinking water quality, ambient water quality for recreational water bodies, industrial discharges); performance-based standards; charges to water rights; reducing water allocations; restrictions or bans on activities which have an impact on water resources (e.g. polluting activities in catchment areas, bans on phosphorus detergents); abstraction and discharge permits; water rights; land use regulation and zoning (e.g. buffer zone requirements for pesticides application).
Voluntary and Education Instruments	Providing information to water users about the costs and benefits of their actions to encourage voluntary behavioural change	Metering of water usage; eco-labelling and certification (e.g. for agriculture, water-saving household appliances); voluntary agreements between businesses and government for water efficiency; introduction of carry-over for water holders to store water over multi-periods; water donation schemes; promotion of awareness raising and training in ecological farming practices or improved irrigation technologies; stakeholder initiatives and co-operative arrangements seeking to improve water systems, e.g. between irrigators and water utilities; planning tools (e.g. integrated river basin management plans).
Economic Instruments	Assumes that water consumption or production can be influenced by water charges, subsidies change water demand for irrigation by providing greater incentives for other (less water intensive) production, tradable rights can be achieved through institutional water rights being created	Charges (e.g. abstraction, pollution); user tariffs (e.g. for water services); payment for watershed services (e.g. for protection of catchment upstream); water pricing; reform of environmentally harmful subsidies (e.g. production-linked agricultural support, energy subsidies for pumping water); water markets; buy-back of water entitlements; buy-back of water allocations; subsidies (e.g. public investment in infrastructure, social pricing of water); tradable water allocations (temporary water) and entitlements (permanent water), options and quotas; insurance schemes (Grafton and Wheeler, 2015).

Source: Settre and Wheeler (2016), p. 69.

2.6 Water Resources in the Murray-Darling Basin, Australia

2.6.1 MDB Physical Characteristics Overview

The MDB is Australia's largest and most regulated river system and covers approximately 14% (1,055,600 km^2) of mainland Australia (ABARE-BRS, 2010). The MDB is located in the south-east of Australia and covers all or part of four states and one territory of Australia: NSW, SA, Vic, QLD, ACT, as shown in Figure 2.3. The Basin contains three of the nation's longest rivers: the Darling (2,740km), the Murray (2,530km) and the Murrumbidgee (1,690km), and contains 440,000km of river length in total (BOM, 2014c). Due to the vastness of area, the MDB represents a varied landscape, ranging from alpine regions in the Australian Alps, highly variable ephemeral river systems in the north, and extensive arid and semi-arid plains in the south (BOM, 2014c).

The MDB is separated into two sections; a) the northern Basin, and b) the southern connected Basin. The southern MDB (sMDB) basin is highly regulated and contains the majority of MDB irrigated agriculture (NWC, 2011). The sMDB is hydrologically connected which allows water trade to occur between irrigation communities and the environment in South Australia, Victoria and New South Wales (Wheeler et al., 2012). The northern basin is less regulated, has higher rainfall and runoff variability and has lower levels of abstraction compared to the southern Basin (BOM, 2014b).

Within the northern and southern Basins, sub-catchments or river valleys form the boundaries for local level water planning and management. The *Murray-Darling Basin Plan* (discussed further in Section 2.7.4) delineates the MDB into 29 Sustainable Diversion Limit (SDL) surface-water resource units, which are the regional scale at which *Basin Plan* objectives are implemented.

Figure 2.3 The Murray-Darling Basin, Australia

2.6.2   Murray-Darling Basin Climate and Hydrology

Rainfall and Runoff

Surface water hydrology in the MDB, as everywhere else on planet Earth, is primarily determined by the climate (Kundzewicz, 2008). The climate in Australia is among the most variable in the world and is influenced by large-scale world climactic drivers including the El Nino Southern Oscillation, Indian Ocean Dipole and the Southern Annular Mode[4] (Neave et al., 2015). These large-scale climactic drivers are responsible for the high annual and decadal changes in the Australian and MDB climate, including recurrent droughts and floods (Baines, 1998). In particular, bush fires and droughts in south-eastern Australia are largely a result of the El Nino Southern Oscillation (Baines 1998). Rainfall in the MDB

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[4] Also known as the 'Cerberus' of Australian climate.

is low and variable compared to other areas, as shown in Figure 2.4. The MDB also contains part of South Australia, which is the driest state in the country (Pigram, 2007).

Figure 2.4 Total Rainfall Volume (2005-2006) in Australia
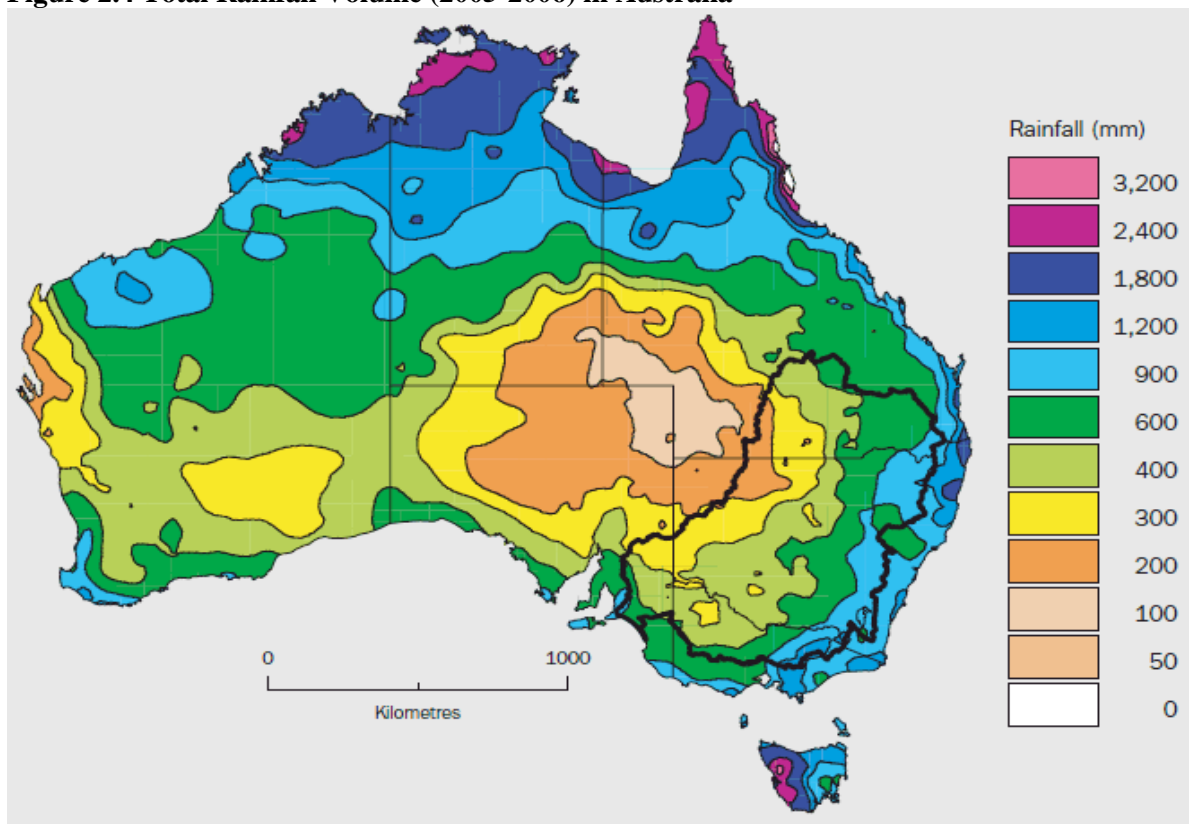
Source: ABS (2010). Available:
http://www.abs.gov.au/AUSSTATS/abs@.nsf/Lookup/1301.0Chapter3042009%E2%80%9310,
[Accessed 22nd June, 2017]
Notes: Black outline indicates MDB.

The total long-term average rainfall in the MDB is 530,618GL (ABS, 2008a). Of this amount, 4% becomes runoff and the remaining is evapotranspired (94%) or contributes to deep drainage (2%) (ABS, 2008b). This is shown in by the long-term water balance components of the MDB and whole-of-Australia in Table 2.2.

Table 2.2 Long-term Water Balance Components in the MDB and Australia

Water Balance Component	Volume (GL)	
	MDB	**Whole-of-Australia**
Rainfall	530,618	2,704,913
Evapotranspiration	497,290	3,291,649
Run-off	23,609	349,431
Deep drainage	9,719	63,833

Source: ABS (2010). Available:
http://www.abs.gov.au/AUSSTATS/abs@.nsf/Lookup/1301.0Chapter3042009%
E2%80%9310, [Accessed 22nd June, 2017]

Additional, but relatively negligible, inflows to the MDB include limited spring snowmelt from the Alpine regions. Transfers into the MDB region also occur from the Snowy Mountain Hydro-eclectic scheme, which transfers water into the Murrumbidgee and Murray rivers, at a prescribed volume of

670GL (BOM, 2014b). Accounting for all sources, the long-term average inflow in the MDB is 11,200GL/year (ABS, 2007). MDB seasonal rainfall for water year 2013-14 and long-term mean monthly flow are shown in Figure 2.5.

Figure 2.5 Mean Monthly Flow and Rainfall in the MDB for 2013-14
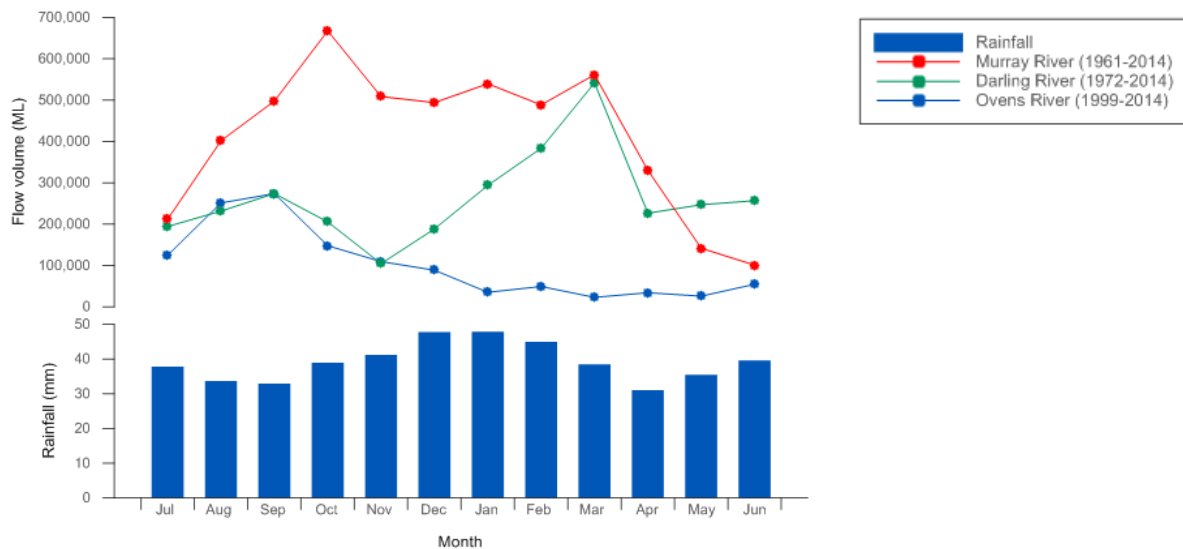
MDB Water-use

The long-term average rate of diversion for consumptive and environmental management purposes in the MDB is 42% of total surface water runoff (MDBA, 2013). Surface water in the MDB is abstracted for consumptive uses including agriculture, potable and domestic supply, industry, and by the water supply industry (e.g. losses in water transport). Figure 2.6 shows the overall proportions of use by each sector, both in the MDB and Australia. Irrigated agriculture is the highest consumer of water in the MDB and accounts for 82% of water consumption in 2004-2005 (ABS, 2008b). Within the agricultural sector, overall water-use and irrigation application rates are heterogeneous between irrigation farms due to land use characteristics as well as water-use efficiency and technology. Agricultural water-use characteristics have also changed considerably through time owing to changing water policy; including purchases of water entitlements and irrigation efficiency upgrades (see Section 2.7.4), reduced water availability due to drought, and a reduction in the number of irrigation businesses in the MDB over time.

Figure 2.6 Water-use in the MDB and Whole of Australia in 2004-05

Murray-Darling Basin Water Use 2004-05

13% 1% 2%

2%

82%

- Agriculture
- Households
- Lost in Delivery Systems
- Manufacturing
- Other Industries

Whole of Australia Water Use 2004-05

10%

3%

11%

11%

65%

- Agriculture
- Households
- Lost in Delivery Systems
- Manufacturing
- Other Industries
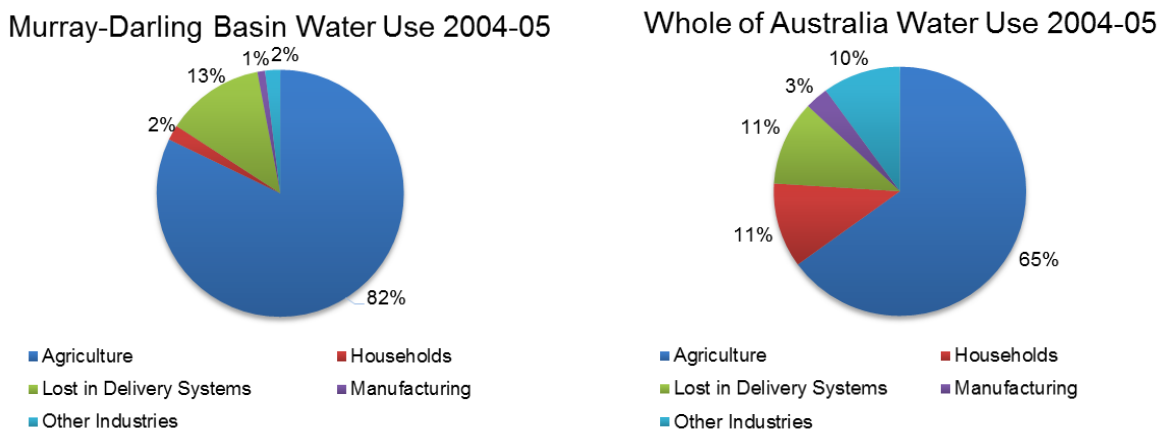
Source: Own Figure. Data sourced from ABS (2008b)

Importantly, surface water demand in the MDB is counter-cyclical, such that highest water demand (i.e. irrigation season) occurs in the period of the year where surface water would be naturally low in the arid and semi-arid regions of the basin. Unlike other nations, which rely heavily on groundwater to meet agricultural demand, groundwater accounts for a small proportion (~14%) of consumptive water demand in the MDB (ABS, 2008b). However, Haneash et al. (2016) found that groundwater use is increasing in cases where irrigators have sold water surface water entitlements and substituted irrigation water-use with groundwater pumping.

Drought

The MDB has historically been subject to extreme drought and flooding, the most significant over the last century, including the Federation Drought (1895-2003), the mid-Century Drought (1937-1947), and the most recent Millennium Drought (1997-2010). The Millennium Drought represents the driest period in the last 110 years of climate record (Timbal, 2009) and is the most severe drought since European settlement in Australia. Although the MDB has been previously affected by severe drought, the Millennium Drought is different due to the following (SEACI, 2010):

- The length of rainfall deficiency surpasses any previous on record (Figure 2.7);
- No isolated wet years within the drought period;
- Seasonal changes within individual drought years; and
- Geographically confined and intensified in south-eastern Australia.

Figure 2.7 Rainfall Deficits across Australia from January 1997 to December 2009

Rainfall Decile Ranges

	Highest on Record
10	Very Much Above Average
8-9	Above Average
4-7	Average
2-3	Below Average
1	Very Much Below Average
	Lowest on Record
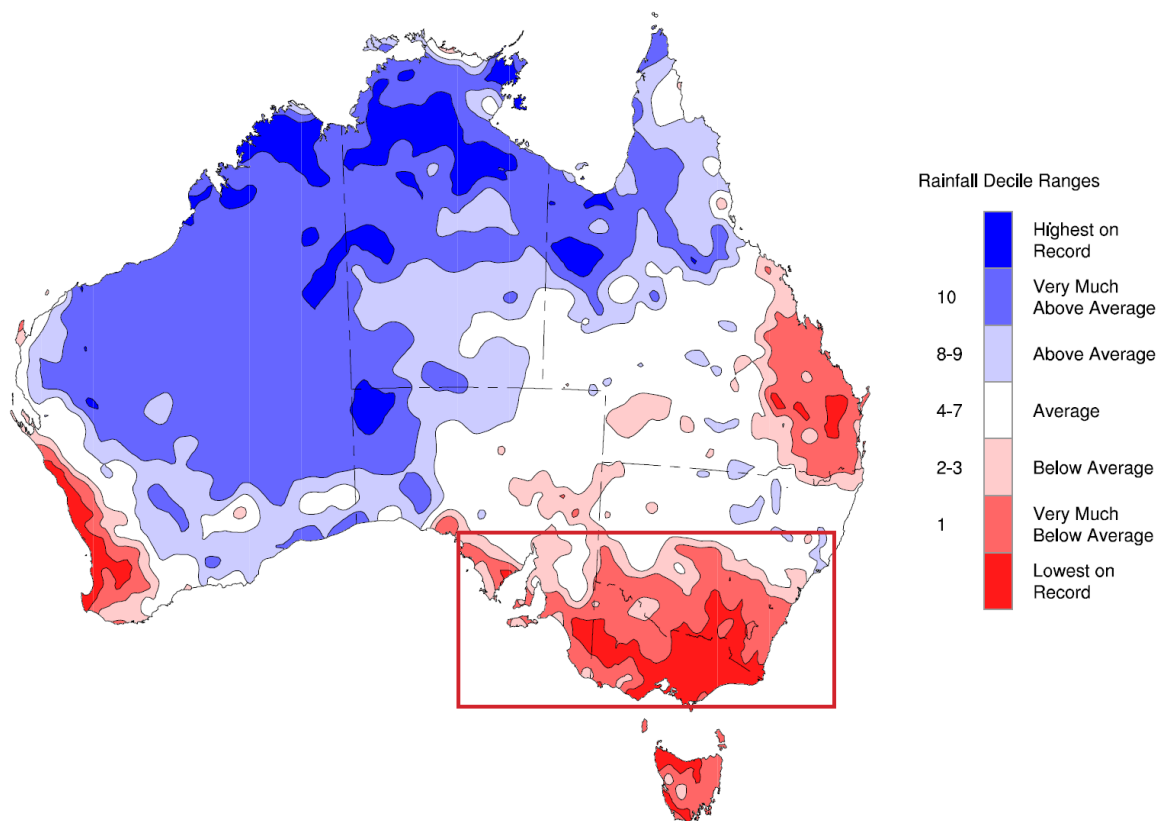
Source: SEACI (2010), p. 6.

Estimates of the severity of the Millennium Drought differ by reporting body and geographic scope. It is generally accepted that during the drought, inflows reduced to approximately 40% of the long-term average in the MDB (ABS, 2007; MDBA, 2011). The impact of the Millennium Drought on annual flows at the Murray Mouth highlights its severity, as demonstrated in Figure 2.8.

During the Millennium Drought the severity of over-allocation and water resource development was exposed by the drastic and rapid reductions in water availability. The Millennium Drought had particular implications for policy makers and planning in the Basin, due to the previously unrecorded severity and longevity (Adamson, 2015). The need for a strategic and rapid policy response to manage the decline in the MDB resources spurred one of the most significant reform processes in Australian water policy history, namely the *National Water Initiative* (2004), and the *Water Act* (2007), which led to the development and adoption of the *Basin Plan* in 2012 (Wheeler, 2014). This reform process is detailed further in Section 2.6.5.

Figure 2.8 Average Flows at the Murray Mouth from 1980 to 2010
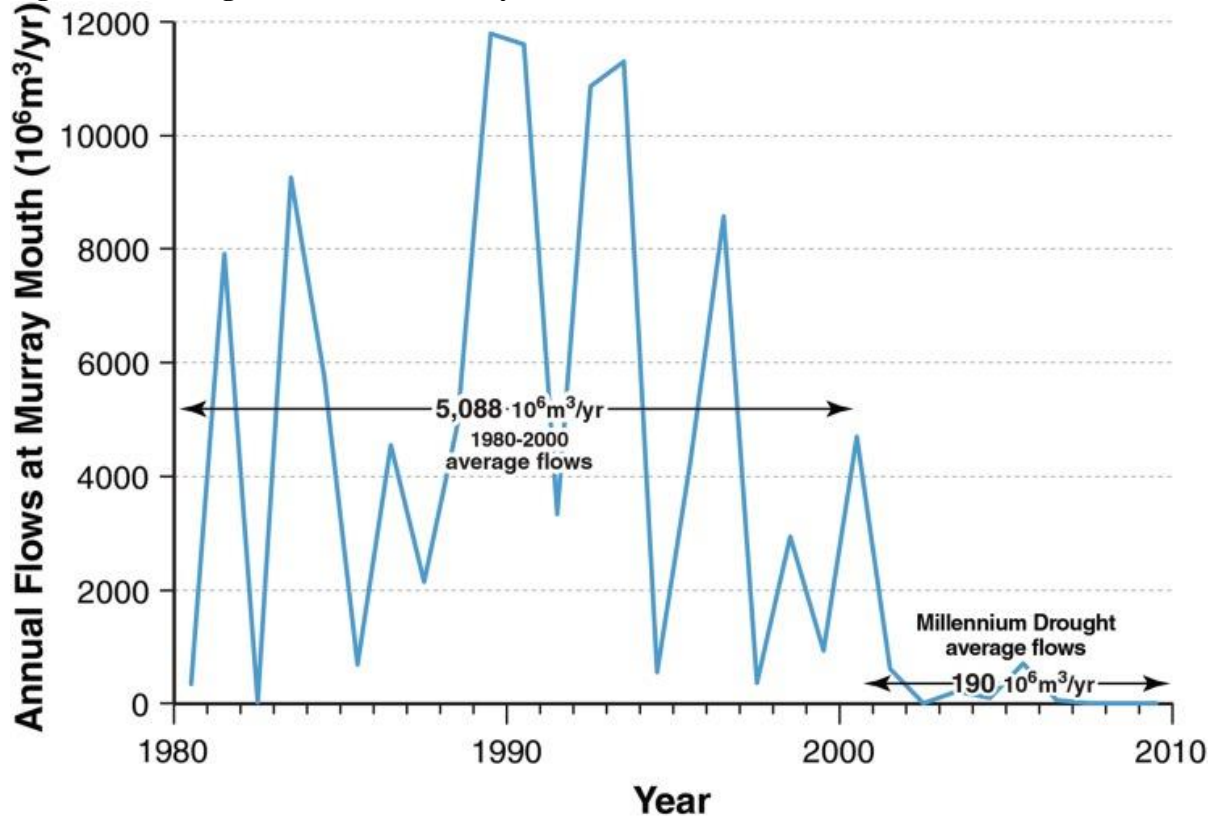
Source: Grafton et al. (2014), p. 1085.

The impacts of the Millennium Drought on MDB resources can be examined through two lenses: the changes in irritated agriculture and production values, as well as changes to the natural MDB ecosystems. From an irrigation point of view, reduction in total overall water availability had significant impact on the volume of water available for production. This resulted in transformation of farming production and management, incurring a reduction in the volume of water used, reduction in farming output, farm investment, and farm related jobs (Wittwer and Griffith, 2011). However, while the reduction in total overall water availability had an impact on irrigation productive capacity, the reduction of economic returns to irrigation was less than the reduction in water availability (Kirby et al., 2014b) due to farmer adaptation such as fallowing, input substitution, deficit irrigation, land use change, technological upgrades and water trading (Qureshi et al., 2013c; Qureshi and Whitten, 2014; Kirby et al., 2014b). As can be seen in Figure 2.9, decreases in water allocations are particularly evident in the lowest water availability year in the sMDB (2006-2007), where announced allocations were near zero for some low/general entitlement holders. In QLD, allocations increased in 2006 following the break of the drought in the northern basin, while drought persisted in the sMDB until the 2010-2011 high inflow years (MDBA, 2011; Figure 2.9).

Figure 2.9 Total Irrigation Water Diversions and Allocations by State During the Millennium Drought
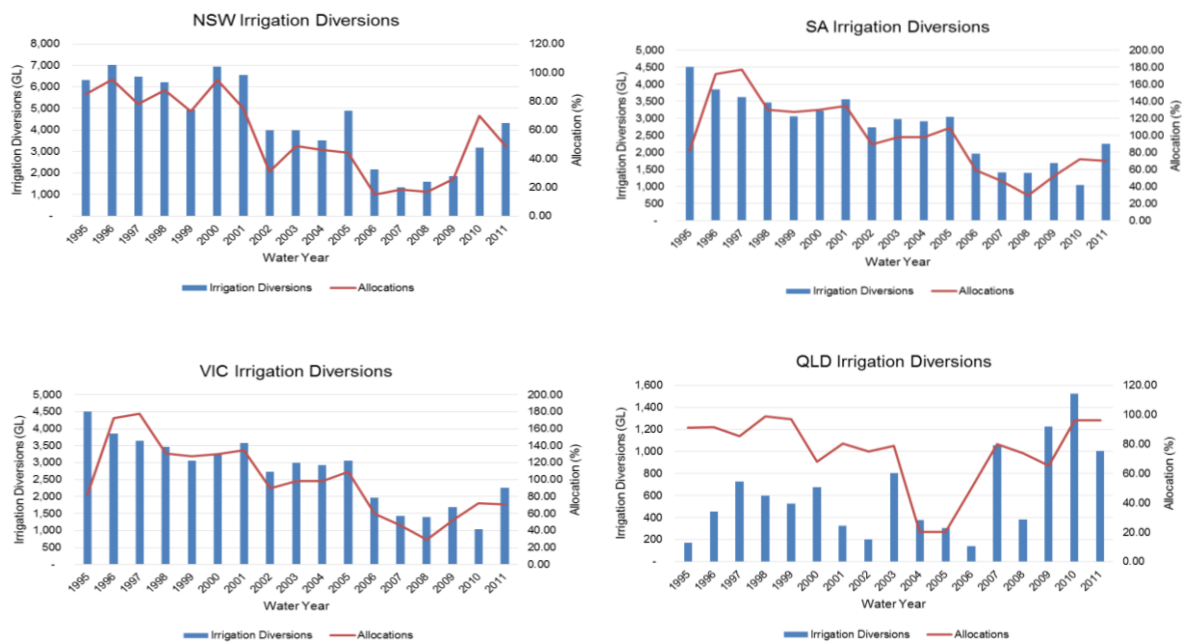
Source: Own Figures. Data provided by MDBA.

Drought also caused considerable widespread decline in the flow-dependent ecological condition of the Basin. For example, in 2001-2002, the Murray Mouth neared closure following 630 consecutive days of no flows through the barrages from the river Murray, due to drought conditions (Phillips & Muller, 2006). As a result, the Murray River did not functionally run to the sea for this period and resulted in a loss of hydrological connectivity. The reduction in hydrological connectivity between the estuary and the ocean resulted in the salinization of the Coorong Lagoon and salinities in the areas of the Coorong Lagoon (which is typically estuarine in nature), reaching marine and hypersaline levels in 2002-03 (Geddes, 2003). This resulted in considerable decay in water quality and a loss of ecosystem services in the order of $810 million within the Lower Murray region (Banerjee et al., 2013). Chapter 4 provides greater detail and analysis of drought management in the Coorong, Lower Lakes and Murray Mouth (CLLMM).

The drought also caused considerable ecological damage in upstream regions of the sMDB. For example, in 2003 it was estimated that up to 90% of River Red Gums in the MDB were water stressed due to drought conditions, which reduced the volumes available for the environment (MDBC, 2003). The Hatha Lakes demonstrated the largest extent of River Red Gum stress and dieback, where it was estimated that only 5% of the trees were maintained in a healthy condition during the drought period (Cunningham et al., 2010).

In 2010-11, the sMDB received a rapid annual increase in water availability, creating flooding events in the Murray, Murrumbidgee, Goulburn, Ovens, Campaspe and Loddon rivers, signalling the end to the extended drought period in the MDB (MDBA, 2011). This also resulted in increases in active water

storage, which recovered past the long-term average for the first time since 2001 and improved water security in the sMDB (MDBA, 2011).

2.6.3 Climate Change in the MDB

The global costs and benefits of realised climate change impacts are largely uncertain and highly heterogeneous between regions (IPCC, 2007). In arid and semi-arid regions, the negative consequences of climate change impacts are likely to outweigh potential benefits, by changing the spatial and temporal distribution of water supply and demand (IPCC, 2007; Elbakidze and Cobourn, 2013).

In the sMDB climate change is likely to have negative impacts on the quality and quantity of water available (Quiggin et al., 2010; Chiew et al., 2011). For a median climate change scenario, overall surface water availability is expected to decrease by a median amount of 11% by 2030 (CSIRO, 2008a), accompanied by increased salinity loads (Beare and Heaney, 2002), water quality deterioration (e.g. algae blooms) (MDBA, 2016c) and increased variability of supply (Connor et al., 2012). The impacts of climate change within the MDB are likely to be spatially diverse (Beare and Heaney, 2002) with the highest average reductions in surface water availability expected to occur in the sMDB (CSIRO, 2008a). Reduced water availability has particular ramifications for irrigated agriculture, which is located predominately in the sMDB. Agricultural profits are expected to decrease only a small amount under a median climate scenario, although can be expected to decrease substantially if an extreme dry climate future is realised (Jiang and Grafton, 2012).

Overall, droughts are also projected to become more frequent and severe in the MDB (Chiew et al., 2011) and, as such, the reduction of water availability and impacts of the Millennium Drought can provide an example of future climate conditions faced in south-eastern Australia (SEACI, 2010; Grafton et al., 2014). Table 2.3 demonstrates the overall reduction in rainfall and runoff based on current, recent and projected climate change scenarios.

Table 2.3 Rainfall and Runoff Volumes for Historic, Recent and Future Climate Scenarios

Climate Scenario	Rainfall (mm)	Rainfall (%) change from historic)	Runoff (mm)	Runoff (% change from historic)
Historic (1895-2006)	457	0	27.3	0
Recent (1997-recent)	440	-4	21.7	-21
Future - median (2030)	444	-3	24.7	-9
Future - dry extreme (2030)	396	-13	18.3	-33
Future - wet extreme (2030)	495	8	31.7	16

Source: CSIRO (2008), p. 21.

Reductions in water availability will naturally affect the volumetric supplies for irrigation and environmental use at a spatial and temporal scale, depending on the reliability of the water entitlement

(Crase & Gawne, 2011). While town supplies (typically high security) are not likely to be reduced due to provisions; the supply of critical human needs, general and low security water entitlements will be largely affected (CSIRO, 2008a).

Although the *Basin Plan* (see Section 2.7.4) offered an opportunity to mainstream climate change risk-analysis into national decision-making, this was not fully realised (Kirby et al., 2014a; Pittock et al., 2015). Climate change management is instead intended to be managed by state water management plans (MDBA, 2012d). Crucially, there is a lack of clarity regarding how future reduced water availability will be shared between consumptive water and the environment under the current *Basin Plan* arrangements (Pittock et al., 2015). Failure to incorporate current climate change knowledge in the *Basin Plan* arrangements presents a major risk to the benefits expected to be gained by reallocating water to the environment, as part of the overall *Basin Plan* (Kirby et al., 2014a; Pittock et al., 2015). Based on the continuation of current water sharing arrangements, the impacts of climate change will be most prominent at the terminus of the MDB in the lower Murray region and the Lower Lakes (CSIRO, 2008a), which are internationally important wetland assets. While little can be done to prevent the recurrence of drought, realistic evidence-based policy can influence the consequences of drought (Booker, 1995). The risks associated with the current policy upon a return to drought conditions and a more variable future climate in the MDB are yet to be fully addressed (Pittock et al., 2015).

2.6.4   Land Use in the MDB

The MDB area supports a wide range of land uses, the largest of which by area is grazing (ABARES, 2014). Land use is shown graphically in Figure 2.10. Figure 2.10 demonstrates the concentration of irrigated agriculture in the sMDB, owing to a higher level of water resource development in this region. As can be seen, conservation and natural resource areas are also largely situated in the sMDB. Note that the area of land-use is not indicative of the volume of water consumed, as irrigated agriculture and urban areas (including the water supply industry) are the major water users aside from the environment in the MDB, though combined they occupy only 4% of the total MDB land mass (ABARES, 2014).

Figure 2.10 Land Use in the MDB
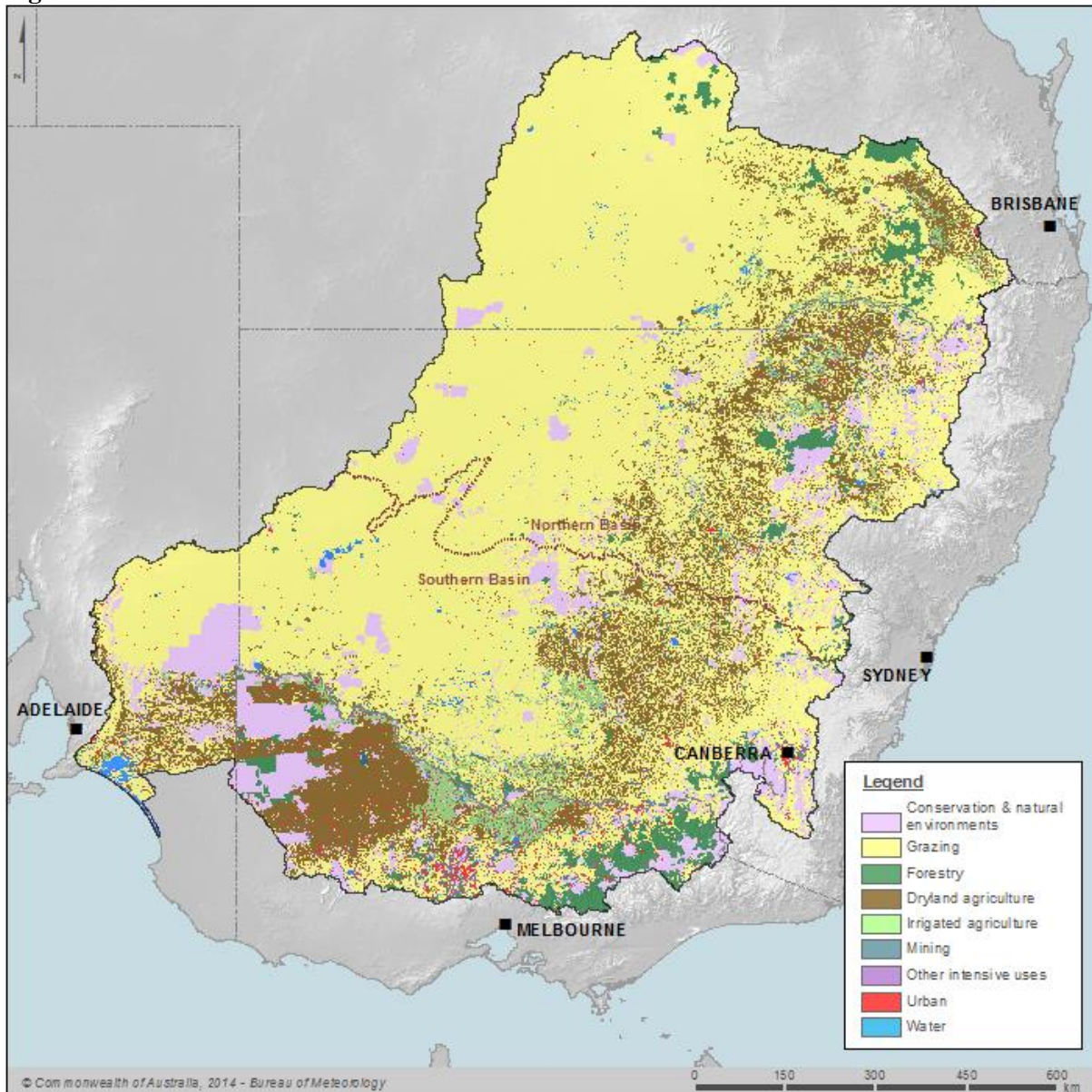
Agriculture

Agriculture is an important aspect of the Basin's economy, identity and social fabric. Approximately 38% of all Australian farmers reside within the MDB (ABS, 2008a) and the basin contains around 70% of all irrigated farmland in Australia (MDBA, 2015). Maximizing irrigation efficiency is a key theme in current MDB policy, and therefore irrigated agriculture water-use is discussed here.

The total water abstraction for irrigated agriculture in the MDB was 7,720GL in 2005-06, which accounted for 66% of total irrigation water abstractions across all of Australia. The distribution of irrigation water usage is shown in Table 2.4.

Table 2.4 MDB Water Use, Irrigated Water Use, and Irrigated Agriculture (2015-2016)

State	Total Water Use ('000ML)	Water Used for Irrigation ('000ML)	Number of Agricultural Businesses ('000)	Number of Irrigating Agricultural Businesses ('000)
New South Wales	2 805.3	2 610.9	26.1	5.3
Victoria	2 095.0	1 946.1	20.8	6.0
Queensland	2 646.1	2 433.5	18.2	5.4
South Australia	858.8	777.8	9.5	3.1
ACT	0.3	0.1	0.0	0,0
MDB Region	*5 209.9*	*4 938.4*	*35.3*	*9.2*

Source: ABS (2017). Available at:
http://www.abs.gov.au/ausstats/abs@.nsf/PrimaryMainFeatures/4618.0

Urban

The MDB is a populated region and approximately 2 million people reside within the Basin, with the majority within NSW and Victoria (ABS, 2008b). The MDB also supplies Adelaide, located outside of the MDB, which can source 85% of its potable water from the River Murray during years of low rainfall (DWLBC, 2009). The largest urban centre in the MDB is Canberra, which uses, treats and returns water to the Murrumbidgee River for further use downstream (BOM, 2014c).

Environment

Listed conservation and natural resource areas (excluding conservation on private land) extend over 10% of the total MDB area (ABARES, 2014). The MDB is host to a number of nationally and internationally important conservation areas, including one world heritage site (Willandra Lakes) and 16 wetlands listed on the Ramsar Convention on Wetlands of International Significance (MDBA, 2010a). Further discussion of the CLLMM, a Ramsar listed wetland at the terminus of the MDB is presented in Chapter 4. A significant portion of natural conservation areas in the MDB are dependent on adequate surface water flows. For example, approximately 30,000 individual wetlands cover an area of 25,000km^2 and a further 60,000km^2 of MDB area is covered by floodplains (BOM, 2014c), for which flooding is a primary determinant of ecological health (Junk et al., 1989).

2.6.5 Environmental Health in the MDB

During the development of water resources in the MDB, water entitlements were allocated freely to consumptive users with little or no recognition of environmental water requirements (Quiggin, 2001). An over-allocation of water rights coupled with rapid water resource development began the process of ecological degradation in the Basin (Kingsford, 2000). In 2004-2007, the first Sustainable Rivers Audit (SRA) was undertaken to assess the ecological health of the MDB and conducted again in 2012. The SRA is a comprehensive evaluation of the condition of fish, birds, vegetation and

macroinvertebrates, physical form and hydrology of the Basin, and ranks each catchment on a scale of 'good condition' to 'extremely poor condition' (Davies et al., 2010).

Overall, it was found overall that MDB assets, including bird and fish species, vegetation and wetland areas, are in moderate to poor conditions (Davies et al., 2010; MDBA, 2012e). Paroo catchment, which is the least regulated with the lowest diversions in the MDB, was found to be healthiest catchment in the Basin in both SRA iterations (Davies et al., 2010; MDBA, 2012e). Table 2.5 reports the findings from the 2004-2007 and 2012 SRA results and demonstrates the overall trend in ecosystem health within each catchment. As evident in Table 2.5, in the period between the two studies there has been a continuation of mostly 'poor' to 'very poor' conditions in 12 of the river valleys, and a declining trend in six valleys predominately from 'moderate' to 'poor' conditions. Four regions (Castlereagh, Kiewa, Mitta Mitta and the Murrumbidgee) have improved conditions from 2007, and have moved from 'very poor' to 'poor' overall. The overall declining trend in ecological health within the MDB indicates that although policy to improve ecological condition has been implemented, there remains a considerable degree of environmental damage that has yet to be halted and reversed in the Basin. Greater background and discussion of flow dependent environmental assets and environmental water management is presented in Chapters 3, 4 and 6.

Table 2.5 Summary of Environmental Trends in the MDB

River Valley	2004-2007 SRA Health Rating	2012 SRA Health Rating	Trend
Paroo	Good	Good	↔
Border Rivers	Moderate	Poor	↓
Condamine	Moderate	Poor	↓
Namoi	Moderate	Poor	↓
Ovens	Moderate	Poor	↓
Warrego	Moderate	Moderate	↔
Gwdir	Poor	Poor	↔
Darling	Poor	Poor	↔
Lower Murray	Poor	Poor	↔
Central Murray	Poor	Poor	↔
Upper Murray	Poor	Poor	↔
Wimmera	Poor	Poor	↔
Avoca	Poor	Poor	↔
Broken	Poor	Very Poor	↓
Macquarie	Poor	Very Poor	↓
Campaspe	Very Poor	Very Poor	↔
Castlereagh	Very Poor	Poor	↑
Kiewa	Very Poor	Poor	↑
Lachlan	Very Poor	Very Poor	↔
Loddon	Very Poor	Very Poor	↔
Mitta Mitta	Very Poor	Poor	↑
Murrumbidgee	Very Poor	Poor	↑
Goulburn	Very Poor	Very Poor	↔

Source: Adapted from Davies et al. (2010), p. 766; MDBA (2012), p.25.

2.7 Phases of Water Management and MDB Water Policy Reform

Natural resource management in the twenty first century is a highly complex problem of sustainability (Foerster, 2011). As the complexity of issues has increased, so too has the complexity of the institutional arrangements governing water resources. Throughout time, this has been mirrored in MDB water management through the transition of four stages of water policy phases, as shown in Figure 2.11 including; the exploration phase, the development phase, the scarcity or maturity phase, and the current phase of transitioning to sustainability. The transition to sustainability can also be thought of as the contraction stage of water policy, characterised by a reduction of irrigation in the Basin and a more balanced and efficient allocation of resources (Adamson, 2015). A central characteristic of the transition to sustainability has been the use of water markets to balance extractive and environmental water demands (Grafton and Horne, 2014). Table 2.6 shows an overview of characteristics of each stage of water in the MDB.

Figure 2.11 Phases of Water Policy in the MDB
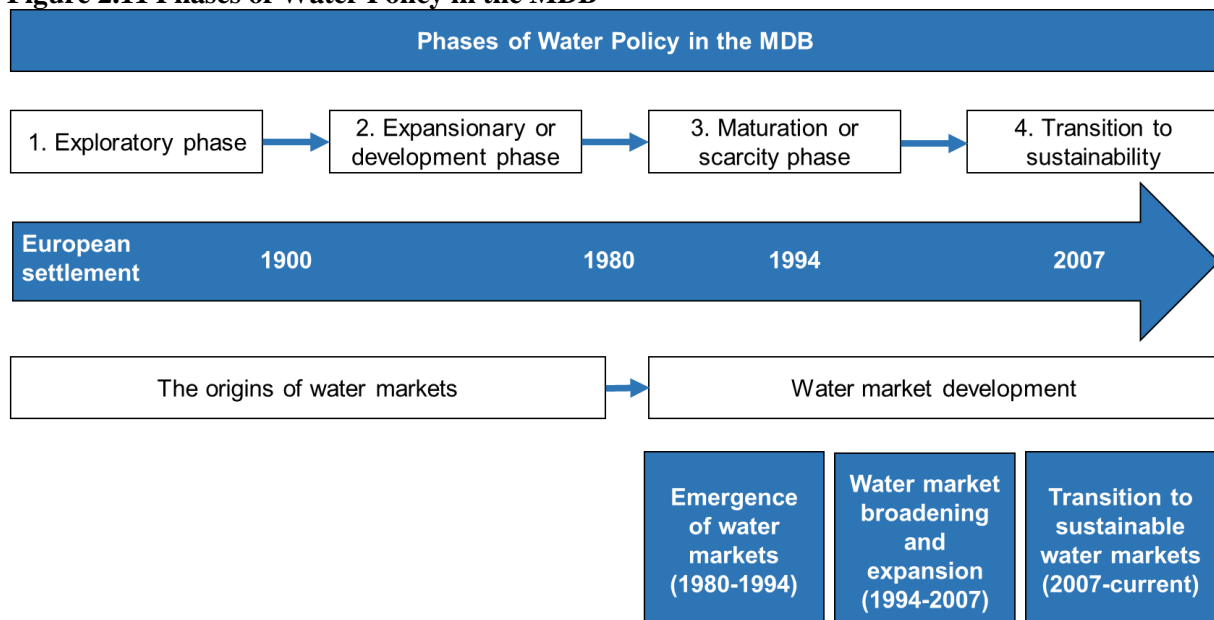
Source: Recreated from NWC (2011), p. 4.

At each phase of management, policy and institutional reform were required to amend existing laws to better overcome resource challenges (e.g. scarcity), society's changing preferences (e.g. the environmental movement), external driving factors (e.g. macro-economic influences) and new information (e.g. climate change). A summary of the legislative reforms which have occurred over the past century are shown and briefly described in Table 2.7. The overall reform process has led MDB water policy from a largely supply-driven approach, to the consideration of water rights for irrigators and the environment as a tradable commodity, along with a market-based governance of water. The following section provides an overview of the phases of water policy and the reform process, focussing on the transition to sustainability and legislative reform from 1994 onwards, which most directly shapes current environmental water marketing and future directions in this policy space.

Table 2.6 Characteristics of Phases of Water Policy in the MDB

Water Policy Phase	Exploration	Development	Scarcity	Transition to sustainability
Consumptive water demand	No or very low water demand due to low population and water-use, elastic at low prices and inelastic at high prices.	Growing demand for irrigation and urban sectors, prices are elastic at low prices and inelastic at high prices.	High level of demand leading to over-allocation of consumptive rights, prices are elastic at low prices and inelastic at high prices.	High level of demand growing in urban sectors and stable (some decrease) in agricultural sectors, prices elastic at low prices and inelastic at high price.
Competition between water users	No or minimal competition between users, supply exceeds consumptive demand.	Competition increases during droughts early in the 20th century as demands continue to increase.	High competition between all water users, formal use of markets incorporated as a scarcity management tool.	Demand remains high and competition is managed through formal reallocation programs and scarcity management tools. Competition during extreme drought is evident.
Level of water resource development	Minimal, European settlers beginning agricultural development.	Extensive construction of large-scale infrastructure projects, transforming the sMDB from a free flowing river to a sequence of water storages.	Water resources fully developed, very few new developments (exception is Snowy hydro), early infrastructure constructions beginning to deteriorate.	Water resources fully developed, infrastructure managed and maintained by states, some large scale infrastructure in need of remodelling and repair (e.g. barrages at end-of-system).
Cost of publically subsidising water supplies	Very low or zero.	Cost rising as inexpensive places for infrastructure progressively eliminated.	High costs of subsidised supply, government rolls back some agricultural subsidies.	Public funding of infrastructure occurring during drought periods, overall reduction of publically subsidised supplies.
Ecological condition of the basin	Healthy ecosystems, without development environmental conditions.	Environment deteriorates due to water resource development and growing levels of abstraction.	Environmental conditions very poor due to over-allocation and realised impacts of long-term river regulation.	Recovering as water is reallocated to the environment to meet ecological watering requirements.
Long-term sustainability	No	No	No	Yes (possible)

Source: Adapted from Adamson (2015).

Table 2.7 Overview of Significant Water Reforms in the MDB

Year	Initiative	Description	Policy Phase
1886	*The Irrigation Act*	Provides for state control of water. Water management carried out by private irrigation trusts.	Exploration and expansion
1901	Federation of the Australian states	Section 100 of the Australian Constitution allows states reasonable use of water resources. States exhibit command-and-control management of water.	Development and supply
1915	River Murray Waters Agreement	First transboundary water sharing agreement between Victoria, New South Wales and South Australia.	
1985	Murray-Darling Ministerial Council established	Comprises of ministers from each Basin state responsible for decision-making regarding water sharing, funding and integrated policy.	Maturation and scarcity
1987	*The Murray-Darling Basin Agreement*	Supersedes River Murray Water Agreement and creates Murray-Darling Basin Commission (MDBC); addresses limited environmental issues, mostly salinity.	
1989	*Salinity and Drainage Strategy*	Sets out specific salinity reduction targets and management strategies to reduce future soil and water salinization.	
1994	Council of Australian Governments (COAG) Water Reform Framework	Instructs the development of necessary institutional arrangements to facilitate water trade and the recognition of the environment as a legitimate water user.	
1997	The cap arrangements put into place	Introduced to limit all future extractions to 1993-94 levels in response to predictions of unsustainable levels of future use.	
2004	*National Water Initiative* (NWI)	Continues and reinforces 1994 COAG reforms and prescribes state-level objectives for water sharing, water trading and water pricing.	
2007	*The National Plan for Water Security (NPWS)*	$10 billion is committed to return water to the environment through market mechanisms and infrastructure upgrades. Instigates legislation pertaining to the referral of state powers to the Commonwealth.	Transition to sustainability
2007	*The Water Act*	Establishes the Murray-Darling Basin Authority (MDBA) which replaces the MDBC. MDBA charged with the development of the *Basin Plan*.	
2010	*Water for the Future*	A restructure of NPWS, committing $3.1 billion to purchasing water entitlements from willing sellers through the water market, to be used for environmental waiting. Referred relevant state powers to the Commonwealth.	
2012	*The Basin Plan*	The plan developed by the MDBA is passed through federal parliament into law and aims to return extraction in the Basin to sustainable levels.	

Source: Own Table.

2.7.1 The Exploration Phase

After European settlement, there followed a period of exploring and establishing water resources in the MDB. The ruling of Australia as *terra nullius* (land belonging to no one) meant that Aboriginal water rights were not recognised, and water management proceeded following the European traditions and objectives. At this time, riparian rights inherited from British common law formed the institutional basis for water management (Ward, 2009). The riparian doctrine permits landholders with land adjacent to a water course the right to make reasonable use of the water, provided it does not unreasonably interfere with other riparian rights holders. However, it became clear that the riparian doctrine inherited from British Common Law did not suit the scarce and highly variable nature of Australia's water resources.

In 1886, the *Irrigation Act* was enacted and replaced the inherited riparian doctrine with state control of water, which was managed through state irrigation trusts (Crase et al., 2015). Soon after, the Australian states were federated and the Constitution of Australia (1901) formally vested the right to water in the individual states. Each state exhibited a command-and-control style governance over water resources, the legacy of which still exerts influence on water management today. Foreshadowing future conflicts, water governance was a point of debate even in the early stages of development of the Australian Constitution (Clark, 2002). Controversy about upstream water extraction (Sim and Muller, 2004), and concerns about water and soil quality degradation due to overstocking and frequent drought, challenged traditional European farming ideals that were inherent in early water institutions in Australia.

2.7.2 The Development and Supply Phase

Throughout the twentieth century, water provision was a strategic resource in the industrialization and agricultural intensification across many rapidly expanding economies, including Australia. Following the federation of the Australian states, water management in the MDB entered the development phase characterised by state-led control and optimistic expansion of water supply. During this time policy and development were characterised by planning for growth, command-and-control regulation, state ownership of resources, and universal provision of water. The expansion of irrigated agriculture, enabled by supply provision, was closely entwined with national development objectives and underpinned much of the economic growth of the MDB (NWC, 2011). The close link to national policy and the state's right to water meant that state governments took a lead role in water resource management and top-down governance prevailed throughout this phase.

Within this period, water was considered unequivocally as a public good (Ward, 2009). This thinking led to the generous outlay of water entitlements to promote the expansion of irrigated agriculture and establishment of towns and cities. In order to ensure secure supplies for the growing urban and agricultural development, this period saw the construction of large water infrastructure projects across

the MDB, such as Hume Dam (1934); the Goolwa Barrages (1940); Dartmouth Dam (1979); Snowy Mountain Scheme (1974) as well as a number of weirs and locks along the river (1922-1939). The development of water resources has since transformed the MDB into a series of storages, now capable of storing 22,214 GL of water at full capacity across the MDB, though all storages are seldom full (MDBA, 2014b). Water infrastructure development received a wide political and financial support due to their necessity for population and industrial growth in Australia, and projects were often undertaken with little cost-benefit analysis or political opposition (Ward, 2009). Large scale infrastructure developments of this time have been criticised for going ahead impact assessments, resulting from the owing to the lack of legislative obligation to consider external consequences of development (Paterson, 1987). The impact of large water resource infrastructure is evident in effect on the natural flow regime, which naturally consists high floods and variability (Overton et al., 2009). Figure 2.12 compares the annual end-of-system flow over the Barrages at the Coorong, Lower Lakes and Murray Mouth (CLLMM) region of the MDB for 2009 (current) development conditions and modelled without development conditions.

Figure 2.12 Annual flow in the Murrumbidgee River at Maude Weir for Current Development Conditions (2009) and Without Development Conditions
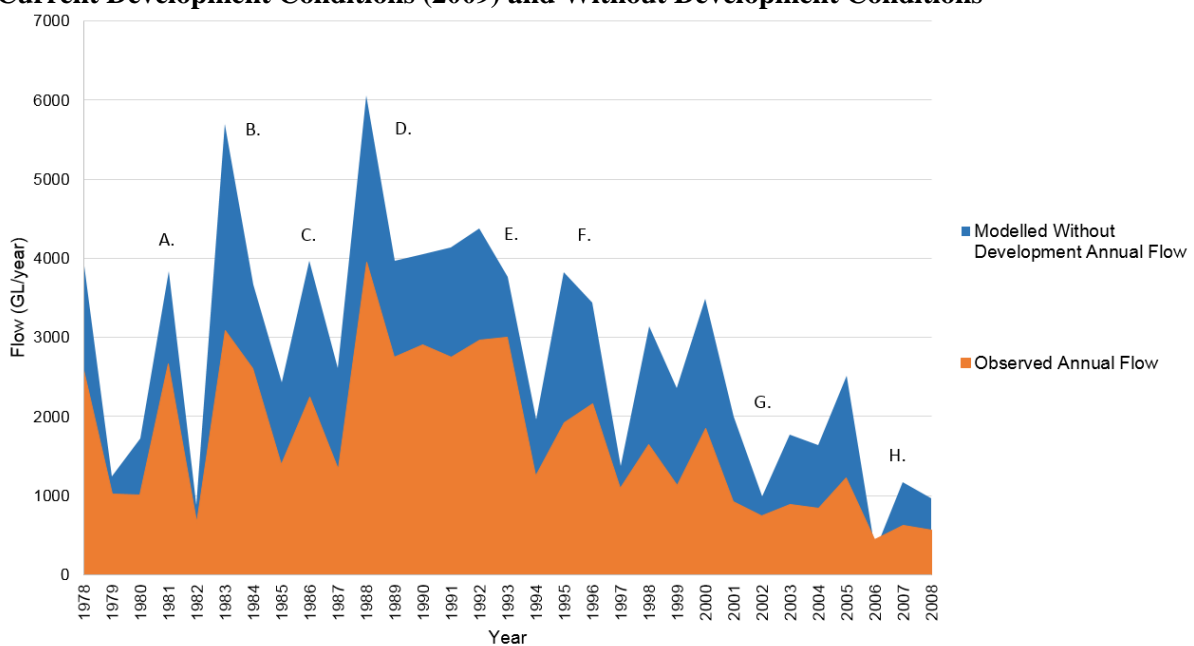
A. and C.   Reducation in moderate inundation events.
B. and D.   Large reduction in large overbank flooding events.
E. and F.   Changes in food peak duration.
G. and H.   Overall reduction of flow during naturally low flow periods.

Source: Own Figure. Flow data supplied by CSIRO.

2.7.3   The Scarcity Phase and The Introduction of Economic-based Instruments

Beginning in the 1960s and gathering strength in the 1980s, a range of factors combined to provide an impetus for policy change and the formal introduction of water markets into the Australian water policy landscape. Up until this point, market mechanisms had been largely excluded from the water sector

worldwide due to the fear of large externalities, such as negative impacts on return flow and downstream water users. Failure to manage such externalities and address additional market hurdles, such as high-transaction costs and information asymmetry, can lead to market inefficiency (Chong and Sunding, 2006). The introduction of market mechanisms was a clear sign of a change in policy objectives, signalling a transition from national development objectives toward aggregate economic efficiency objective (NWC, 2011). Selected factors contributing to the introduction of markets are discussed below.

2.7.3.1 The Rising Cost of Supply Management

A key factor in the transition to demand-based water management policies was the lack of willingness of governments to fund large-scale infrastructure projects. This was partially a result of the sharply rising cost of supply (Randall, 1981) because many of the more viable options for increasing supply had been exhausted during the development phase (NWC, 2011). Continued government involvement in supply provision was being questioned, as was the merits of the developments themselves (Davidson, 1969). The growing emphasis on demand-based policies, such as water pricing and markets, signified that the Australian water sector was becoming a mature water economy. Characteristic of this, the cost of developing new supply infrastructure was becoming inelastic (Randall, 1981). As Australia's water economy continued to mature, the entrance of economists and environmentalists into the water realm (traditionally dominated by engineers) led to the expansion of new water management ideas in the water sector (Musgrave, 2008; Ward, 2009) and the development of more integrated networks. Catalysed by government fiscal pressures (NWC, 2011) and the increasing role of economics in water management decisions, cost-benefit analyses were also increasingly undertaken on large-scale infrastructure projects and irrigation schemes for the first time.

2.7.3.2 Public Environmental Concern

In addition to waning political will and the cost of generating new supplies, concerns about the environmental impact of large-scale water infrastructure were beginning to be established in public discourse (Musgrave, 2008). The issue was compounded by the recognition of over-allocation of water entitlements in some states of the MDB. The ecological damage and economic limits caused by the undervaluing and over-allocating water in the Basin began to be questioned in the late 1960s (e.g. Davidson, 1969). In the years following, it became clear that the institutions formed throughout the establishment and development of Australian water resources (both the constructed formal institutions and the inherited social perceptions) had left a legacy of high extractive demands far above the levels of ecological sustainability. In response, South Australia placed a moratorium on the issue of new water licenses in 1969; New South Wales placed a full embargo in 1981; and Victoria restricted unregulated stream use in 1968 (Wheeler et al., 2014a). Water scarcity conditions were exasperated in 1982-1983 by severed rainfall deficiencies in Victoria and New South Wales, known as the South East Australia drought.

In 1987, the original *River Murray Waters Agreement* (1915) was superseded by the *Murray-Darling Basin Agreement*, which established the Murray-Darling Basin Commission (MDBC). Though the MDBC had limited environmental responsibilities, the resulting Salinity and Drainage Strategy produced by the MDBC in 1989 provided one of the earliest formal environmental management schemes in the Basin and signalled changing policy direction and growing environmental concern. The 1987 agreement was in turn superseded by the *MDB Agreement* (1992), which displayed a pronounced shift toward the idea of balancing environmental water requirements with consumptive use (Connell and Grafton, 2008). In these early stages of environmental water management, changes in government and policy were largely spurred by exogenous variables, such as drought and salinity concerns.

More widely, environmental concerns about the rate of development were internationally institutionalised with the release of the Bruntland Report (1987), which provided a compelling definition of sustainability. In addition the 1992 Earth Summit in Rio de Janeiro brought water scarcity and ecosystem vulnerability to the forefront of international concern leading to the declaration of the *Dublin Principles*. The four principles form the basis of integrated water resource management (IWRM), which strives to manage social and economic development, without compromising the sustainability of natural ecosystems (GWP, 2012). IWRM principles are prevalent in many progressive water management strategies and many reforms in the MDB provide examples of IWRM.

2.7.3.3 Macroeconomic Factors

The shifting focus from supply augmentation to demand-based policies in the MDB occurred among a background of wide ranging political liberalization and changing trade agreements. Up until the late twentieth century, Australia had been largely sheltered from international agricultural prices due to high import tariffs and embargos on the importation of agricultural products (Musgrave, 2008). However, a general trend of reduced government involvement in agriculture and trade in the 1980s and 1990s meant that government became less likely to invest in agricultural supply provision. The gradual curtail of government protection over Australian agriculture cumulated in international agreement in 1994, following the Uruguay Round of Negotiations initiating reform in agricultural trade. The reform fully exposed Australia to international commodity prices at the farm level, resulting in the need for additional flexibility in water management to navigate global supply and demand patterns.

The trend toward market instruments over command-and-control management was institutionalised in Australian law in 1994 through the Council of Australian Government (COAG) reforms, which introduced Australia's competition policy. Despite some criticism, water was included in Australia's competition policy and declared a tradable and marketable good. As was the case in Australia, this time period signified a transition from "state to market" in most OECD countries (Haggard and Webb, 1994) resulting from the waning institutional power of the state and erosion of authority from traditional political sources (Roger and Hall, 2003).

2.7.4 Water Policy Reform Leading to the Transition to Sustainability

The MDB is currently in a transition to sustainability and employs a more pluralistic governance approach which features water markets as a central management tool (Wheeler, 2014). This stage of policy is the result of decades of reform. From an environmental water perspective, a key outcome of the reform process is that the environment has become a statutorily recognised water user in its own right and has been awarded the same degree of security as water entitlements held for consumptive use (COAG, 2004). The following sections detail the reform process which has led to the eventual introduction of markets for environmental water, beginning with a brief overview.

2.7.4.1 Overview of Water Market Development

A key aspect of the transition to sustainability is the multiple values of water recognised in policy designed to achieve efficient water resource allocation (Grafton and Horne, 2014; Grafton and Wheeler, 2015). This idea of the various and competing economic values of water is cemented in international institutions by the *Dublin Principles* (1992) which states that: w*ater has an economic value in all its competing uses and should be recognised as an economic good (Guiding Dublin Principle #3)* (GWP, 1996, p.1)*. The economic concept of water highlighted in the *Dublin Principles* provides the basis by which to evaluate and inform water-use trade-offs. However, the economic conception of water is sometimes a point of political contention due to confusion between the commodification, privatization and marketization of water (Bakker, 2005), as well as the symbolic nature of water as an essential life-giving human right (Hanemann, 2006).

Water markets are considered by many economists to play a vital role in the efficient allocation of water resources in the MDB (Wheeler, 2014). Unofficial water trade in Australia first stemmed from water scarcity during drought. There are a number of reports of water "swapping" that occurred during the 1940s drought. South Australia stopped issuing new water licenses in the late 1960s, followed by other states, and these restrictions did lead to informal MDB markets for temporary water in the 1960s and 1970s (Connell, 2007; NWC, 2011). The first official markets for temporary water were officially trialled in the early 1980s, and trade between private diverters and district irrigators was allowed in 1995.

The broadening and formalization of markets in the MDB is a result of considerable policy reform. Specifically, towards the end of the 20th century a 'cap' on water extractions at the 1994 level formally put an upper-limit on the volume of extractive use, effectively laying the groundwork for a cap and trade framework, which paved the way for formal water marketing (Grafton and Horne, 2014). Progress in the reform process through the *National Water Initiative* (2004), which instigated the un-bundling of water rights from land rights, was a definitive step in fully allowing water to be traded separately from land between water users, in a formal water market setting (Bjornlund, 2006). Progress of expanding and maturing the MDB water market continued with the passage of the 2007 *Water Act,*

which had important implications for water property rights and trade, namely; defining and securing water property rights; reforming markets to improve effectiveness; and removing barriers to trade water out of irrigation districts and between regions.

At present, the MDB hosts one of the most mature water markets in the world with respect to economic efficiency, environmental sustainability and robust institutions. Functional water markets in the MDB now means water allocation is no longer solely prescribed by administrative top-down regulations, but distributed in a way that can flexibly respond to price signals, changing supply and demand profiles, and irrigator preferences. Building on the more pluralistic nature of market-based governance fostered by reform in the MDB, water markets have also been used by the federal government acquiring water for the environment (Grafton and Horne, 2014). The use of market mechanisms in environmental management is a landmark decision in Australia, as environmental management has been historically regulatory in nature (Weal, 1992). Further details and discussion is presented in Chapter 3. The following section details the reform process which has led to the formal introduction of water markets in the MDB.

2.7.4.2 COAG Reforms and the 'Cap'

In 1994, the Council of Australian Governments (COAG) instigated what would become an ongoing water reform process lasting over two decades. The water reform framework established a system of tradable property rights for water to facilitate flexible reallocation and the recognition of the environment as a legitimate water user (COAG, 1994).

As part of the COAG reforms and the ongoing concern over the environmental implications of over-allocation, a water audit of the MDB was undertaken. The audit found that water extractions had significantly increased since 1988 and were likely to continue to increase unsustainably if extraction was not curbed. It asserted that modifications to existing water arrangements must be amended to prevent major adverse impacts on the health of the MDB (MDBMC, 1995). In response to the water audit, a temporary cap was introduced to limit all future extractions to levels of 1993-1994 levels and was made permanent in 1997 (Connell, 2007).

The cap on extraction, as well as further restrictions such as limits on out-of-basin trade, played a role in activating underused (dozer) or unused (sleeper) licenses. This had the effect of temporarily increasing water-use in the Basin, as holders of sleeper and dozer licenses entered the market to meet demands of water users impacted by the cap. Further, as the aggregate availability of water was constrained by the cap, water users began to explore additional ways of obtaining water, including increased groundwater use and farm dams to capture unregulated flows. This had the ultimate effect of reducing return flow to the river and further contributing to river water availability concerns. Far from establishing a self-regulating cap-and-trade market with the introduction of the cap, state involvement

became important for the role of regulating private extractions from streams, as well as metering and charging the use of groundwater and farm dams (Bell and Quiggin, 2006).

Despite the negative changes to return flows and the temporary increase in water-use, the cap represented a strong commitment by the federal government to address water scarcity in the Basin and a milestone in cooperation between the Basin states (Wheeler, 2014). In a review of the cap operation it was found that while the cap did not necessarily provide for sustainable use of Basin resources, it was an essential first step without which degradation of Basin resources would have been significantly worse (MDBC, 2000).

2.7.4.3 The National Water Initiative

The *National Water Initiative* (*NWI*) is a transboundary agreement signed by New South Wales, Victoria, South Australia, and the Australian Federal Government. In 2004, the *NWI* was signed and continued the process of water reform instigated by the COAG reforms in 1994. The *NWI* replaced the fragmented state-based control of water resources with a state-federal management system supported by joint agreements, federal objectives, and individual state commitments enshrined in legislation (Foerster, 2011). By signing the agreement, governments made a commitment to address eight interrelated aspects of water management to achieve a more integrated approach to the planning, management, and trading of water in Australia (NWC, 2004). Among other key objectives, the *NWI* sought to address over-allocation of water entitlements, improve river condition and reform the structure of water rights, as discussed below.

2.7.4.4 Unbundling of Water Rights

A key objective of the *NWI* was the removal of barriers to trade through the restructure of water rights, such as changing the definition of a right from a volumetric entitlement to a share of a common pool resource (NWC, 2004). In addition, the *NWI* prescribed the removal barriers to trade such as the bundling of land and water rights. In doing so, the *NWI* removed the legal ties between water and the land, to allow each to be bought or sold separately (Crase et al., 2015). Further and more nuanced unbundling of water rights were instigated and the following four aspects of a water right were constructed: a) a water access entitlement; b) a water allocation account; c) a delivery share ensuring the irrigators have a right to get the water delivered to their property; and d) a water-use entitlement, giving irrigators the right to use the water. The history of water licenses, commercial farming, and established user-owned irrigation systems facilitated unbundled water trading with relative ease. Figure 4.3 provides an overview of the unbundling process.

The unbundling of land and water was a politically charged issue, due to fears that the separation of land and water would result in the privatization of water rights and the emergence of water barons. There were also concerns about the negative impact of water markets on the environment resulting from changes to the natural timing and location of river flows (Wheeler et al., 2014b). Though concerns

continue to be voiced, evidence suggests that the impact of water trading is small compared to the impacts of drought and river regulation, and that key ecological assets in the southern MDB have not been significantly impacted by water trading (NWC, 2012). In addition, tradable water rights provided policy makers with an additional policy tool to reallocate water from consumptive use to the environment, through the purchase of water entitlements from willing sellers.

Figure 2.13 Unbundling of Water Rights
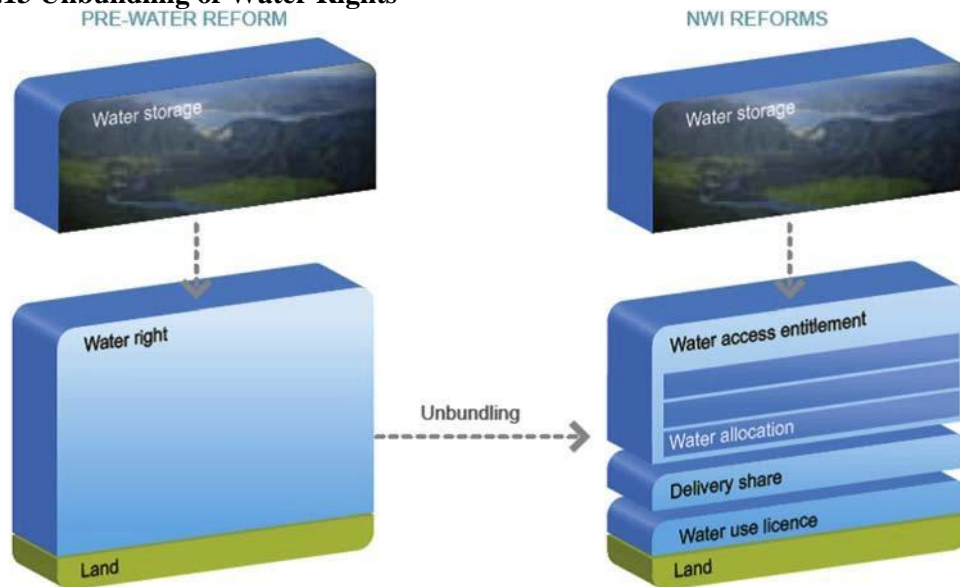
Source: **NWC (2011), p. 88.**

2.7.4.5 The Living Murray Program

The *NWI* required state commitment to address environmental concern arising from scarcity and over-allocation. In particular, governments committed to preparing water plans with provision for the environment and in dealing with over-allocated water systems (NWC, 2004). However, commitment to environmental provision forms only a single component of a comprehensive water reform agenda, concerned with a variety of objectives including urban water management, expansion of trade, and improved storage and delivery, to name but a few. In the vastness of the reform agenda and its many objectives, concerns about the sidelining of environmental goals were validly raised (Connell, 2007).

The *NWI* planned to return consumptive water to the environment through dual mechanisms. Namely, an increase in water-use efficiency and the use of the Australian entitlement water market to buy water entitlements from willing sellers. *The Living Murray (TLM)*, initiated by the Murray-Darling Basin Ministerial Council (MDBMC) and beginning in 2003, was developed in response to evidence of the continued ecological decline of the Basin environment and was a key water recovery policy mechanism included in the *NWI*. The program identified and targeted six environmental icon sites across the Basin for environmental works and measures, as well as the delivery of environmental water. *TLM* First Step involved a commitment of 500GL to be delivered to the six icon sites by 2009 with a budget allocation

of $500 million. *TLM* and the *NWI* were initially presented as a combined package, given they were agreed upon at the same COAG meeting (2004). However, lack of coordination and tension occurred between the policies, particularly regarding the volume of water required to provide adequate environmental flows (Connell and Grafton, 2011). Though the *NWI* was effective in setting out high-level national objectives such as the watering of environmental sites, large amounts of supporting detail, and enforcement were left to the states. The degree of commitment and pace of implementation differed between states, resulting from varying state interests and benefit of reforms. Achieving a common approach on highly politicised issues of the reform agenda, such as the expansion of water markets and purchase of environmental water, were particularly difficult to coordinate (Foerster, 2011). Hesitation among some states prevailed, as it proved a difficult political decision for one state to unilaterally decrease water consumption if efforts were not matched by the other Basin states. As the reform continued, it became evident that individual states were unable to manage the ongoing and pressing ecological issues in the Basin and that coordinated state and federal action was required, leading to the development of the *Water Act* (2007).

2.7.4.6 *The Water Act and Institutional Change*

In 2007, the federal *Water Act* was passed into law and sought to achieve large legislative, regulatory and stakeholder reform through coordinated federal-state action. The passage of the Act came at a turbulent year in Australian water resource management due to height of the continued drought (discussed in Section 2.6.2). Folke et al. (2005) suggest that ecological crisis or abrupt change, such as drought, can provide an opportunity to catalyse the transition to new forms of governance. In the case of governance of MDB water resources, the Millennium Drought provided a catalyst for political will to embrace alternative policy tools, such as market mechanisms to recover water for the environment, delivering additional strategic management tools to irrigators (Wheeler, 2014).

The governance arrangements as prescribed by the *Water Act* are still essentially a top-down approach. Where previously the right to water was vested in the states, relevant constitutional powers were agreed to be referred to federal agencies to enable it to manage the MDB in the national interest (McCormick, 2007; Foerster, 2011). For example, the *Water Act* legislation established the independent Murray-Darling Basin Authority (MDBA) and endowed it with functions and powers needed to ensure that the Basin's resources are managed in a sustainable and integrated way (*Water Act,* 2007). For the first time in Australian constitutional history, the Basin was managed and overseen by a federal semi-independent authority rather than the individual states. The MDBA, which replaced the MDBC in 2008, was charged with the responsibility of developing a plan (the *Basin Plan*) to secure the future sustainability of water resources in the MDB. For the following three years, the MDBA worked on the *Basin Plan*, amid much political change and scientific debate.

In addition, the *Water Act* strategically established a number of new or expanded roles for federal government agencies to undertake and inform the reform process: the Commonwealth Environmental Water Office (CEWO) responsible for acquiring environmental water to be owned by the Commonwealth; the Commonwealth Environmental Water Holder (CEWH) mandated with the task of managing environmental water holdings for positive ecological outcomes; the ACCC to monitor water charges within the basin; the Bureau of Meteorology to develop water accounts (BOM, 2011); the National Water Commission (now dissolved) to monitor and report on the reform process; and the CSIRO to determine the level of sustainable yields from the Basin (CSIRO, 2008a). The strategic creation of these roles increased the speed of water reform, environmental water recovery and better defined objectives. However, some argue that the addition of a central federal governing body added an extra layer of institutional complexity to the governance landscape (Foerster, 2011), as well as a greater distance between the development of high-level objectives and sub-catchment scale water planning.

2.7.4.7 *Water for the Future Initiative*

The *Water for the Future (WFF)* initiative (2010), originally called the *National Plan for Water Security (NPWS)* (2007), represents a milestone in Australian water market-based governance. The *NPWS* was released in 2007 following five years of ongoing drought and committed $10 billion over 10 years to increase production with less water and to improve environmental outcomes.

Following a change of federal government in 2007, the *NPWS* was rebadged as a new intergovernmental agreement on water management; the *WFF* initiative (2010). The *WFF* initiative was a 10-year program involving a commitment of $12.9 billion from state and federal governments. The initiative built upon many of the guiding principles of the *NPWS* but differed from *NPWS* in the explicit reference to climate change impacts and adaptation strategies. Following the template set out by previous reform objectives, the *WFF* uses dual mechanisms of market instruments and infrastructure upgrades to achieve environmental watering targets and large subsidies were offered to irrigators to encourage investments to modernise irrigation. For example, $5.8 billion was provided to rural water-use and infrastructure projects to achieve increased on-farm water use efficiency (DEWHA, 2010). An additional $3.1 billion was committed to purchasing water entitlements from willing sellers through the water market, to be used in environmental watering for the period 2007-2008 to 2016-2017.

The market-based aspect of the program committed to returning over-allocated resources to the environment was labelled *Restoring the Balance* in the MDB *(RtB)*. The program aims to buy water entitlements from willing sellers, which, since 2008, have been mostly selected using a series of competitive tenders (Wheeler and Cheesman, 2013). The allocation water assigned to the water entitlement is used to improve the health of the Basin through watering of key environmental assets

such as nationally significant wetlands, floodplains, and rivers. The *RtB* program is also the key market mechanism used to return water to the environment under the *Basin Plan* (see the following section).

Despite the relative success of environmental buybacks, critiques of the program assert that it exploited financial difficulties faced by irrigators after years of drought, and therefore exploits desperate sellers rather than engaging willing sellers. The counter evidence to this is that desperate sellers were able to sell their water at a high price during a time they were struggling on their farms and many sold water other than for debt reasons. Wheeler and Cheeseman (2013) found that the majority of irrigators who sold water entitlements to the government did so to reduce debt or increase farm income, as well as strategic planning such as farm restructure, farm exit, or disposal of surplus water. Hence, the sale of water either facilitated some to leave the industry, others to reduce debt or to become more viable in general to facilitate farm longevity. Further details on the challenges and social implications of water entitlement recovery are discussed in Section 3.10.

2.7.4.8 The Murray-Darling Basin Plan

The *MDB Plan* developed by the MDBA as mandated by the *Water Act* (2007), is a high-level document specifying coordinated water management actions by Basin states and territory[5] and the Australian federal government. In November 2012, the *Basin Plan* was passed into law despite several attempts to disallow it (Loch et al., 2014b). The *Basin Plan* is now the overarching framework which guides the reallocation from consumptive water to the environment. The timeline of the *Basin Plan* development and implementation is shown Figure 2.14.

At its core, the aim of the *Basin Plan* is *"to promote the use and management of the Basin water resources in a way that optimises economic, social and environmental outcomes"* (*Water Act* 2007, Part 2, Subdivision B, 20(2)). A central step to reach this goal was the specification of Basin-wide, long-term, average sustainable diversion limits (SDLs), which replaced the original cap on water extractions, and are an upper limit on the volume of surface and groundwater which can be taken from the Basin water resources for consumptive use. Debate over the SDLs and the volume required to achieve sustainability was ongoing during the *Basin Plan* development. The Wentworth Group of Concerned Scientists suggested a volume of 4,400GL/year (WGCS et al., 2010). Additional evidence from the MDBA suggests that benefits of reallocating volumes below 4,615GL would be highly uncertain (Grafton, 2011). Despite the (ongoing) debate, it was eventually determined that 3,200GL needed to be reallocated from consumptive water users to the environment, comprised of an initial 2,750GL target and an additional 450GL adjustment mechanism to be achieved through supply and infrastructure initiatives (DoAWR, 2017).

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[5] Queensland, New South Wales, Victoria, South Australia, Australian Capital Territory

Figure 2.14 Timeline for the Development and Implementation of the *Murray-Darling Basin Plan*
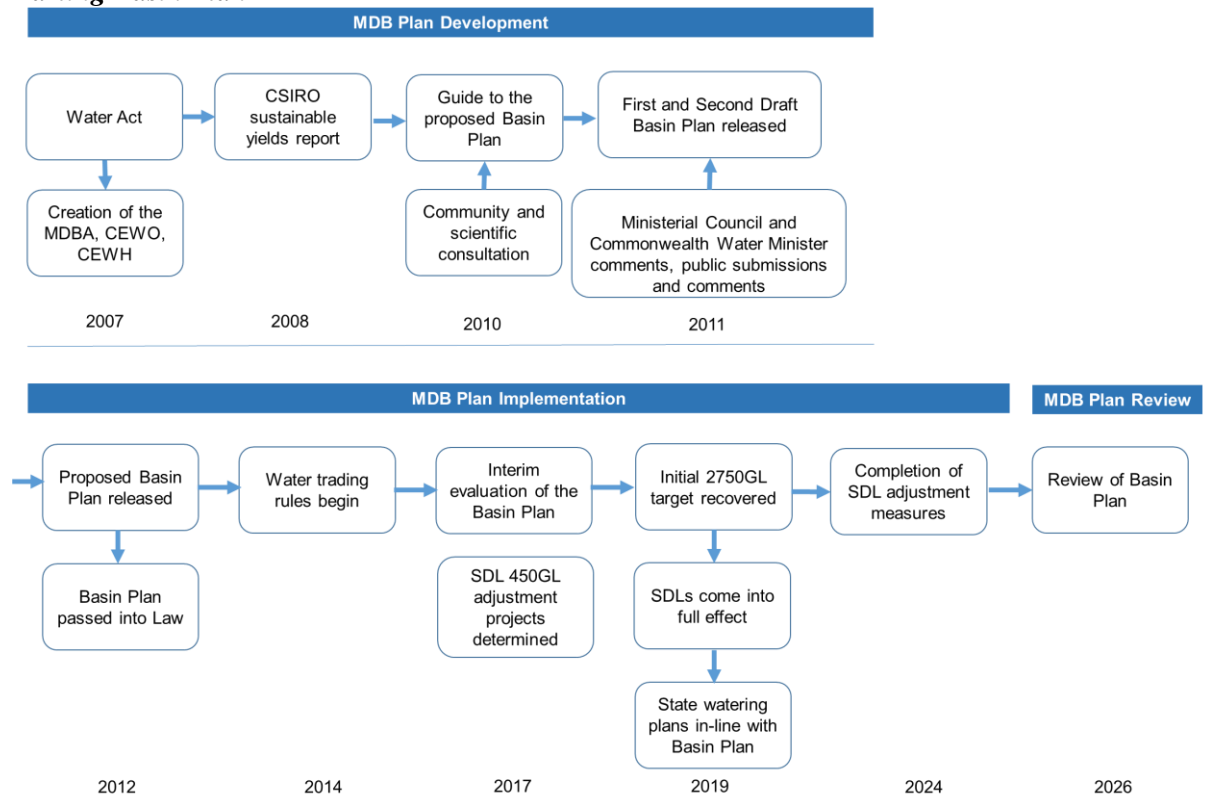
At present, the consumptive use in the Basin is above the SDLs set and water must be reallocated to 'bridge the gap' between current consumptive use and the SDL targets. To facilitate this, the *Basin Plan* utilises a common property approach to rectifying over-allocation of natural resources (Adamson, 2015). The common property approach seeks to balance the ownership of water rights in the Basin by moving the 3,200GL of water from private individuals (e.g. irrigators) to the federal CEWO. The newly created common property water rights are held in the public trust by the CEWO and managed by the CEWH, which has a mandate to manage and deliver the federally owned water to achieve maximum environmental outcomes. Figure 2.15 shows the steady acquisition of Commonwealth environmental water holdings and their long-term average annual yield. As is evident, water holdings have increased steadily since 2008 following the creation of the CEWO in 2007. As of April 2017, progress towards the federal recovery targets was estimated to be approximately 2,500GL of the initial 2,750GL target. This has been achieved with an expenditure of $5.3 billion, which is around half of the $8.9 billion committed by the federal government (Grafton, 2016).

Figure 2.15 Commonwealth Environmental Water Holdings and their Long-term Annual Average Yield as of February 2017
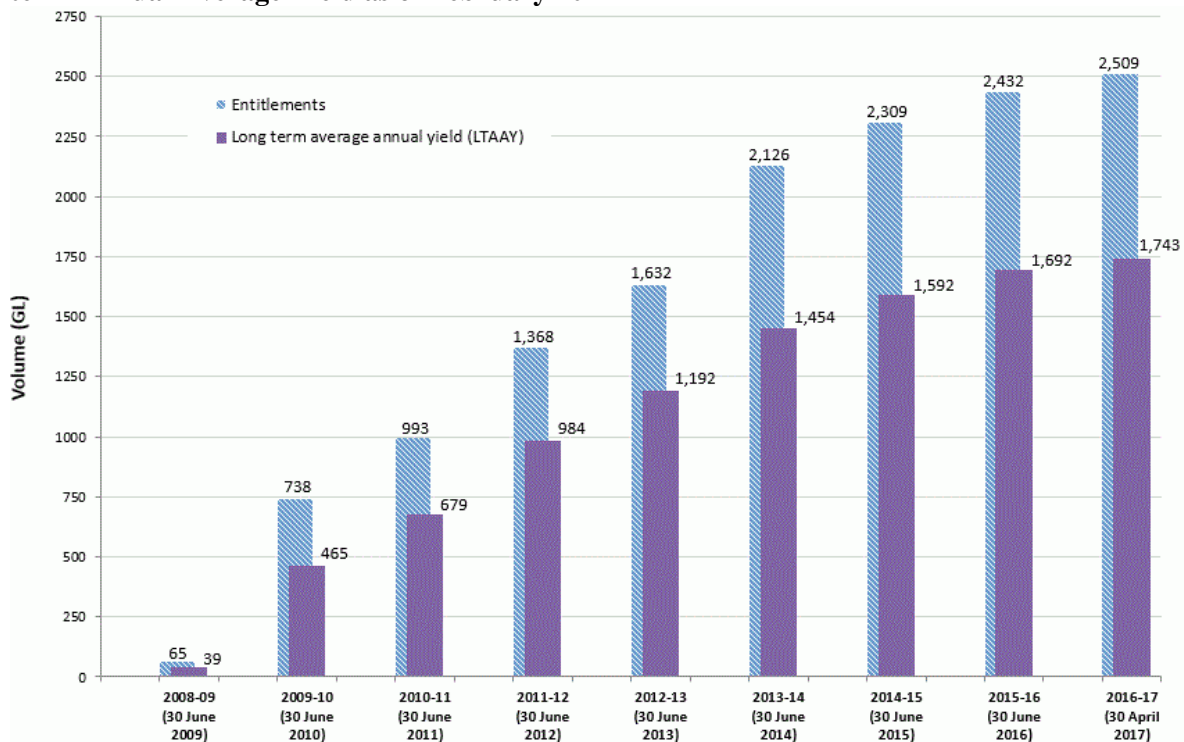
Source: DoEE (2017). Available at: http://www.environment.gov.au/water/cewo/about-commonwealth-environmental-water [Accessed June 10th, 2017].

To achieve the initial 2,750GL water recovery target, two primary mechanisms have been implemented as well as an SLD adjustment mechanism allowing an additional 450GL to be obtained through efficiency or supply measures. These two primary mechanisms involve a market-based approached called *Restoring the Balance (RtB)* and an infrastructure based approach called the *Sustainable Rural Water Use Infrastructure Program (SRWUIP).* To date, the majority of water towards the recovery target has been acquired through entitlement purchases as shown in Figure 2.16.

Figure 2.16 Federal Progress towards the 2,750GL Water Recovery Target
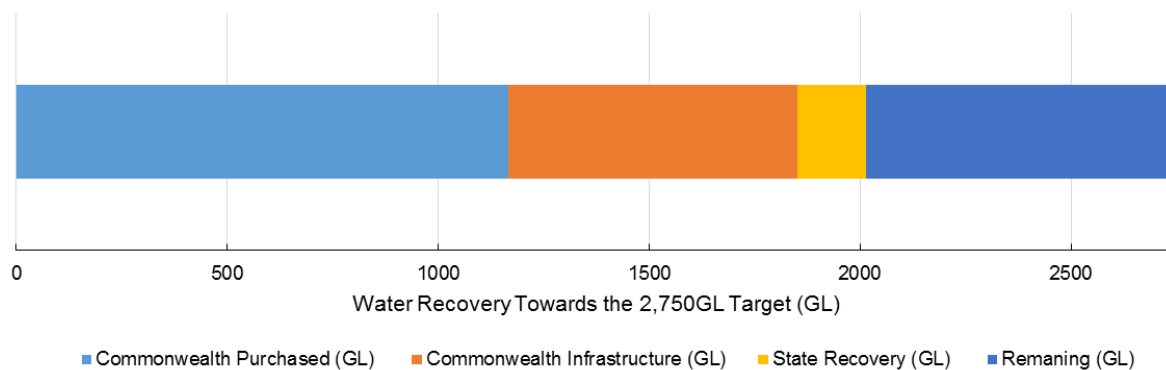
Source: Own Figure. Data sourced from MDBA (2016a).

Although entitlement purchases are the favoured option by economists due to their efficiency in recovering water (Grafton, 2010), policy emphasis has shifted between these two approaches in response to community and political preferences for infrastructure-based reallocation (Loch et al., 2014b). Since 2011-12, water recovery expenditure has focused predominately on infrastructure efficiency grants (MDBA, 2016b) and budget expenditure has been allocated at a ratio of $0.7 on the *RtB* for every $1 on the *SRWUIP*.[6] The trend in policy emphasis and expenditure between these two approaches is shown in Figure 2.17.

Figure 2.17 Australian Federal Government Expenditure on Water Recovery Projects
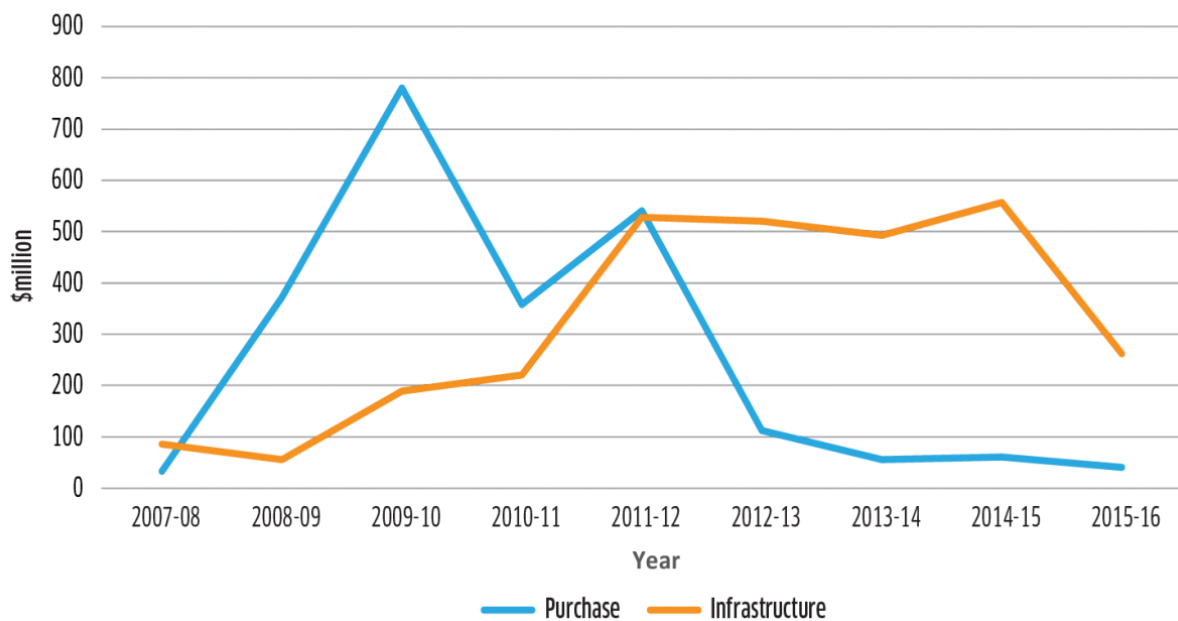
Source: **MDBA (2016b), p. 28.**

The following sections provide an overview of the *RtB,* the *SRWUIP* programs, and the 450GL sustainable adjustment mechanism. A discussion of the advantages of each approach and the overall rationale for market-based reallocation are presented in Section 3.5.

Market-based Reallocation: RtB

The *RtB* operates with a budget of $3.1 billion over ten years to acquire water entitlements for the environment from voluntary sellers, who are compensated at market price. In this instance, there is a direct (1:1 transfer) of water from private ownership (e.g. irrigator) to the common property manager (CEWO). The *RtB* program relies heavily on the purchase of water entitlements rather than other water products (Wheeler et al., 2013a) and the transfer of rights between irrigators and the CEWO is the core of Australia's environmental water market. However, in 2015 a bill was introduced into parliament, known as the *Water Amendments Bill,* to legislate an upper limit on the volume of water able to be

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[6] Based on an expenditure of $2.2 billion on water entitlement purchases and $3.1 billion on infrastructure (Hart, 2015).

acquired through the market to 1,500GL (Hunt et al., 2015). The Bill was introduced by the newly elected coalition government as a means to proclaimed means to improve supply certainty and water security to for regional communities in the MDB (Hunt et al., 2015). The Bill is reflective of the political preference for the use of infrastructure-based water recovery in the Basin, rather than the continued use and expansion of market based policies. As a result of the Bill, all further water recovery from that point was acquired through infrastructure-based reallocation programs or supply measures.

The introduction of the 1,500GL cap could be seen as a contradictory to the general transition towards the use of demand-based instruments and market mechanisms. Emphasis on demand-based and supply-based water policy occurs at the political level and is therefore a function of political philosophy, the preferences of the governing political party's constituents, and lobbying by interest groups. Water policy in this context is therefore inherently transitory with changes in government and policy. The ongoing transition in political emphasis, and hence budget expenditure, is shown in Figure 2.7 which demonstrates the fluctuations in budget expenditure on market- and infrastructure-based reallocation over time.

Infrastructure Reallocation: SRWUIP

The *SRWUIP* program is designed to increase water-use efficiency through an investment of $5.8 billion in farming technology and infrastructure upgrades (Crase and O'Keefe, 2009). Subsidies to improve on-farm irrigation efficiency have been implemented in the Basin since 2009, most recently with an announcement in October 2014 of a $350 million package to fund new irrigation efficiency projects in NSW (DoE, 2014). The *SRWUIP* also provides funding for the Environmental Works and Measures (EWM) Feasibility Program, to fund the installation of additional low-impact infrastructure[7], improving the efficiency of environmental watering (DoAWR, 2016).

Sustainable Adjustment Mechanisms: 450GL up-water

The second volume is 450GL of additional water as part of the sustainable diversion limit adjustment mechanism provided under the *Water Act* (2007). The additional water is to be reallocated beyond the 2,750GL to be recovered under the *Basin Plan* (giving a total of 3,200GL). The 450GL of water is to be recovered through a suite of supply, efficiency and constraint measures agreed upon by the MDBMC in 2016, which is to be funded by a federal commitment of $1.7 billion over 10 years from 2014-15 (DoAWR, 2017). The completion timeframe for the full range of recovery volumes totalling 3,200GL is now extended from 2019 to 2024 (Adamson and Loch, 2014), as also shown in Figure 2.14.

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[7] For example, low-flow bypasses, flow regulators, weirs and channels to facilitate environmental watering.

2.8 Conclusion

This chapter has provided a comprehensive overview of MDB water resources, physical characteristics, water policy reform, and situated the challenges faced in the MDB within the wider international context. Importantly, this chapter has traced the process of reform which has led to the introduction and use of water markets as a central water management mechanism. It was shown that Australia has undergone significant reform, from command-and-control regulation of resources to a market-based water governance regime. This has been accompanied by a shift in policy direction, departing from a focus on national development objectives, to focusing on aggregate economic efficiency. Significant attention has also been directed at addressing over-allocation in the Basin through a variety of infrastructure and market-led solutions, notably the federal government acquisition of water entitlements through the *RtB* program. The ongoing reform process signals a transition from top-down government to market-based governance arrangements jointly reliant on state, market, and civil participation to achieve positive environmental outcomes, as well as more flexible arrangements to governance of environmental water. Building on the background presented in this chapter, Chapter 3 provides a detailed examination of water market-based environmental water recovery strategies.

CHAPTER 3: Rebalancing the System: Acquiring Water for the Environment through the Market

This chapter presents an expanded version of a book chapter published in *Water for the environment: from policy and science to implementation and management* (2017). The original chapter was written as an introduction to environmental water markets in the MDB and the western US, specifically covering topics of water market fundamentals, water products for the environment, environmental water holders and common challenges faced in market-based reallocation. Additional background material (e.g. MDB physical description) from the original version of this book chapter has also been removed to prevent repetition.

The expanded version presented here begins with a more detailed discussion of water markets in the MDB, current market status and scope, and farmer adoption of water markets. Having established the basics of MDB water markets, the chapter explores the use of the market as a method of reallocating water to the environment through entitlement purchases. To situate the MDB experience within an international context, two international case studies of environmental water markets (the western US and northern Mexico) are also discussed. Drawing on these case studies and the detailed discussion of the environmental water market in the MDB a number of challenges of entitlement acquisitions are identified.

In summary, this chapter sets the scene by establishing the current scope of the environmental water market in the MDB and identifying gaps in literature and policy, which are expanded on further in the review and analysis chapters of the thesis. Importantly, this chapter demonstrates that although considerable institutional progress has been made to acquire water for the environment in the MDB, a number of challenges remain, thus requiring ongoing water resource and policy research.

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Contribution to the Paper	Paper research, analysis, writing, tables and figures, conclusions, editing.		
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By signing the Statement of Authorship, each author certifies that:

- the candidate's stated contribution to the publication is accurate (as detailed above);
- permission is granted for the candidate in include the publication in the thesis; and
- the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Sarah Wheeler		
Contribution to the Paper	Content guidance, discussion and editing.		
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3.1 Introduction

The World Economic Forum listed global water shortages among the top three global risks by impact in 2015 and 2016, whereas ten years ago this risk did not even appear on the list (WEF, 2015; WEF 2016). For drier climate projections in semi-arid regions such as the MDB, environmental water volumes and water dependent ecosystems will bear a considerable amount of this risk (Kirby et al., 2014a; Adamson, 2015; Pittock et al., 2015). This is especially the case in fully appropriated or over-allocated river basins, where there is little water left to maintain riverine ecosystems (e.g. base-flow environmental water) or where environmental property rights are weak.

Reducing consumptive use and reallocating water for environmental use has gained increasing international consensus as an important strategy for restoring and protecting flow dependent ecosystems (Benayas et al., 2009; Bullock et al., 2011; Lane-Miller et al., 2013). However, what is less agreed upon is how water should be protected or (re)allocated for the environment. Broadly speaking there are two approaches in allocating water to the environment (Horne et al., 2017): a) restricting water abstractions by non-environmental users; and b) creating property rights for the environment. Table 3.1 summarises key characteristics of these approaches.

Table 3.1 Legal Instruments to Acquire Water for the Environment

Legal instrument	Description	MDB Example
Regulatory Approaches		
Cap on consumptive use	Limit on the total volume of water abstractions and/or the issue of new licences above the cap.	The cap on extractions at 1994 levels and the *Basin Plan* SDLs.
Water licence conditions	Conditions on water-use licences, including limits on the volume and timing of abstraction.	Use-specific water access entitlements, such as irrigation, domestic or environmental uses.
Storage conditions	Conditions on storage release patterns, such as volume, timing, frequency and temperature characteristics.	Weir pool manipulation in the Lower Lakes (lakes cycling).
Property Rights Approach		
Environmental base-flow	Protection of an environmental water right or reserve prior to extraction of consumptive water rights.	Planned environmental water.
Environmental water rights	Water rights owned and managed by an environmental water holder.	Held environmental water.

Source: Partly adapted from Horne et al. *(2017).*

3.2 Water Markets and Trade in the MDB

As documented in Chapter 2, water reform process embedded water markets and trading into the MDB water policy landscape, to address policy inefficiencies in the Basin (Bjornlund, 2006). The following section provides a brief overview of market fundamentals, the current state and scope of water markets in the MDB, as well as market adoption and impacts to date.

3.2.1 Water Market Fundamentals

A water market is the voluntary interaction of willing buyers and sellers, with the purpose of exchanging a property right for water that is legally permitted to be traded. Water markets are an institutional response to water scarcity (Bjornlund, 2003), and where water is plentiful there will be less pressing needs for a market (a notable exception are markets for water quality, see Doyle et al., 2014). The exchanges undertaken in a water market are called water trades, transfers or transactions and can be temporary (lease) or permanent (sale) in nature. The water right, or aspect thereof, that is sold on the market is called a water product[8] and is characterised by the type of right (e.g. groundwater, surface water) and duration the product is exchanged for (e.g. permanent (entitlements), annual (allocations), split-season, options contact). Surface water allocation trade are the most common water market products, due to relative ease of measurement and mobility of the resource, although groundwater trade is also possible within hydrologically connected groundwater jurisdictions.

Under ideal conditions, water markets provide an efficient, compensated and voluntary mechanism to flexibly allocate water between water uses (Howe et al., 1986). Water markets increase the productive value of scarce water supplies by facilitating the flexible allocation of water from low to high value use through trade, and signal opportunity costs to market participants (Howe et al., 1896; Rosegrant and Binswanger, 1994; Chong and Sunding, 2006). Water markets also provide a means to reduce the economic damages from drought by providing an additional tool to cope with scarce supplies and storage constraints (Booker et al., 2005) and as a climate change adaptation strategy (Loch et al., 2013). Howe (2000) identifies key characteristics of markets that are advantageous when applied to water, namely: a) flexibility to achieve efficient use of water; b) security of property rights (depending on terms of the right); c) presenting water users with opportunity costs; and d) ability to minimise transaction costs (if litigation is avoided). Overall, the efficiency and fairness of water markets depends on the geographical, hydrological and institutional underpinnings (Howe, 2000).

Water markets have evolved in a few regions around the world since the 1970s (Chong and Sunding, 2006) in varying degrees of formality (Bjornlund, 2004). In an informal market the right to use an agreed upon volume of water for a short time, such as a year or season, is the water product traded (Bjornlund, 2004). Informal water markets typically occur within irrigation districts between irrigators and proceed with relatively little administrative input. Formal water markets are less common and there are only few examples of mature formal water markets around the world. Formal water markets allow for the legal transfer of the water right on a perpetual basis (as opposed to the right to use for a limited

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[8] Note that because this thesis draws on case studies from Australia and the wester US, the generic term 'water product' coined by Wheeler et al. (2013) is used to describe the type of water right, or aspect thereof, traded in the water market. This approach is used in the general discussions to avoid the issues which arise from using highly case-study specific terminology. Table 3.2 in the following section describes the specific nomenclature of water rights across the states in the MDB, and Chapter 8 provides a discussion of some of the types of transactions that occur in the western US.

period of time), are significantly more institutionally complex, and require a higher degree of administrative input (Bjornlund, 2004). In the MDB, water marketing has progressed to a point of maturity and formality, such that water can be bought and sold through water brokers or online trading platforms.

Water markets can operate at a range of institutional scales involving a variety of willing buyers and sellers. Because agriculture typically uses the majority of water (up to and over 70% in many countries), irrigators tend to be the main sellers of water rights (Hanak and Stryjewski, 2012; Wheeler et al., 2014b). Water sales can occur within an irrigation district from one irrigator to another, or can occur from agriculture to urban, agriculture to industry, or agriculture to the environment, and back again. For example, in the western US, the primary source of water in the market is irrigators, who sell predominately to other irrigators, cities and the environment (Hanak and Stryjewski, 2012). When the environment is recognised as a legitimate and beneficial water user (Garrick et al., 2011), water markets can be used as a platform to acquire water for the environment through the voluntary and compensated reallocation of consumptive water rights, provided a number of institutional criteria are met (Garick et al., 2009). This is discussed further in Section 3.3.

To date, water markets of varying degrees of formality have been observed in India, Pakistan, Mexico, Chile, Spain, Australia, China, South Africa and the United States (Rosegrant and Binswanger 1994; Klozen, 1998; Easter et al., 1999; Bauer, 2004; Nieuwoudt and Armitage, 2004; Grafton et al., 2011a; Moore, 2014; Wheeler et al., 2014b; Palomo-Hierro et al., 2016). The characteristics of water markets are therefore as diverse as the regions in which they operate and are a function of the local market-enabling institutions (Garrick et al., 2011). Although laws permitting the formal transfer of water rights exist around the world, progress and adoption of water markets remains highly contentious and underwhelming. For example, legislation passed in 1999 (Law 45/1999) incorporated formal water markets into Spanish regulatory framework, but high transaction costs, owing to administrative, legal and cultural difficulties, has led to a very narrow water market (Palomo-Hierro et al., 2015). While formal and mature water markets are globally limited, Australia provides an example where water markets have become an integral part of the water management toolkit (Grafton et al., 2016). Figure 3.1 outlines all regions in Australia where water allocation trading currently occurs. As can be seen, trade activity is predominately located in the south-eastern region of Australia within and around the MDB. The following section discusses water market framework, adoption and impacts in the MDB.

Figure 3.1 Australian Catchment Regions with Water Allocation Market Activity in 2015-2016
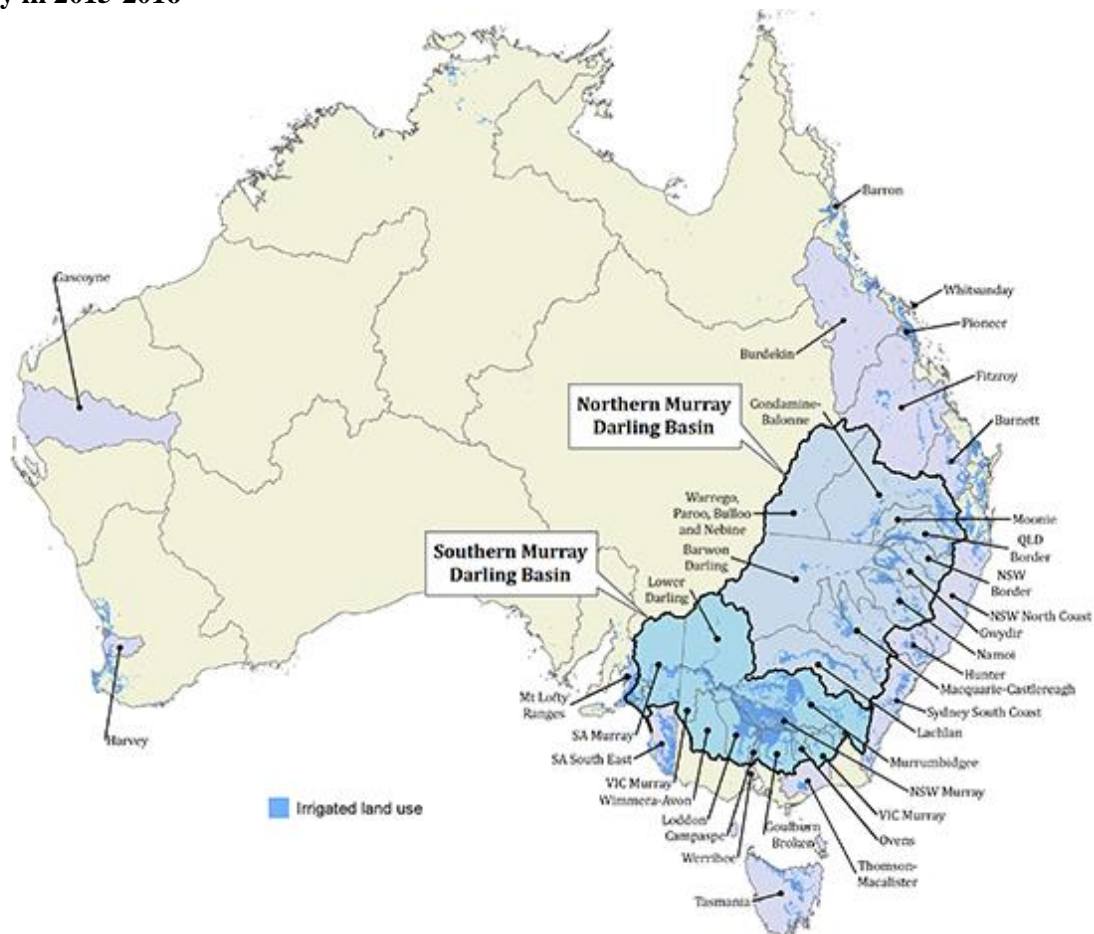
Source: ABARES (2017). Available at: http://www.agriculture.gov.au/abares/research-topics/water/aust-water-markets-reports/southern-murray-darling-basin#seasonal-conditions [Accessed 20th June, 2017]

3.2.2   Overview of Water Markets in the MDB

There are two parallel water markets in the MDB: a) the seasonal market for water allocations; and b) the market for permanent water (access) entitlements. Water entitlements and allocations are defined by the COAG (2004, p. 30):

> Water (access) entitlement: *a perpetual or ongoing right to exclusive access to a share of water from a specified consumptive pool as defined in the relevant water plan.*

> Water allocation: *the specific volume of water allocated to the water entitlement in a given season, defined according to rules established in the relevant water plan.*

Although water reforms and inter-jurisdictional legislation (e.g. the *Water Act*, the National Water Registry) intend to standardise the naming of types of water rights, a number of differences remain. Table 3.2 compares terms between states.

Table 3.2 Water Entitlement and Allocation Naming Conventions across MDB States

Water Product	NSW	Victoria	SA	QLD	ACT
Water entitlement	Water access licence	Water share	Water access entitlement	Water allocation	Water access entitlement
Reliability	High and general	High and low	n/a	High priority and medium priority	n/a
Water allocation	Available water determination	Seasonal determination	Water allocation	Announced entitlement	n/a

To avoid ambiguity, this thesis adopts the naming convention "water entitlement" to describe the right to access a share or entitlement of water from a consumptive pool which yields a seasonal volumetric allocation, and "water allocation" to describe the amount that can be legally extracted by the entitlement holder within a season.

3.2.3   Geographic and Institutional Scope of the Water Market

Water-use and water market activity are highest in the sMDB (Grafton and Horne, 2014), which hosts the most sophisticated water market in Australia. Water allocation trading is the most common form of trading. Entitlement trade is less prominent and entitlement sale is often a strategic decision such as farm exit or restructures (Wheeler et al., 2012). However, entitlement trading has grown considerably since 2007-08, following the Millennium Drought and entry of the CEWH into the market as part of the *RtB* program.

Within the sMDB there are a number of hydrologically connected water systems that transcend state boundaries across which water trade can occur, subject to trading rules and restrictions. In 2015-2016, 89% of surface water allocation trading and 42% of entitlement trade volume occurred in the sMDB (ABARES, 2017). Table 3.3 provides further detail on sMDB trade volumes and numbers. The northern and southern regions of the MDB are not hydrologically linked and therefore trade between them is not possible. Groundwater trade is possible within hydrologically linked aquifers, but the volumes of groundwater trade are very small compared to surface water trading (as seen in Table 3.3). As demonstrated in Figure 3.2, trade in the sMDB has also increased significantly over time.

Table 3.3 Water Trading Activity in the sMDB in 2015-2016

	Allocation Trade	Entitlement Trade
Surface water trade volume (GL)	4,978.5 [89% of national trade]	565 [42% of national trade]
Number of surface water trades	23,087 [87% of national trade]	4,947 [69% of national trade]
Groundwater trade volume (GL)	123.4 [53% of national trade]	84.4 [25% of national trade]
Number of groundwater trades	405 [38% of national trade]	347 [15% of national trade]

Source: ABARES (2017).

Figure 3.2 Volume of Water Allocation and Entitlement Trade in the sMDB from 1983-2013
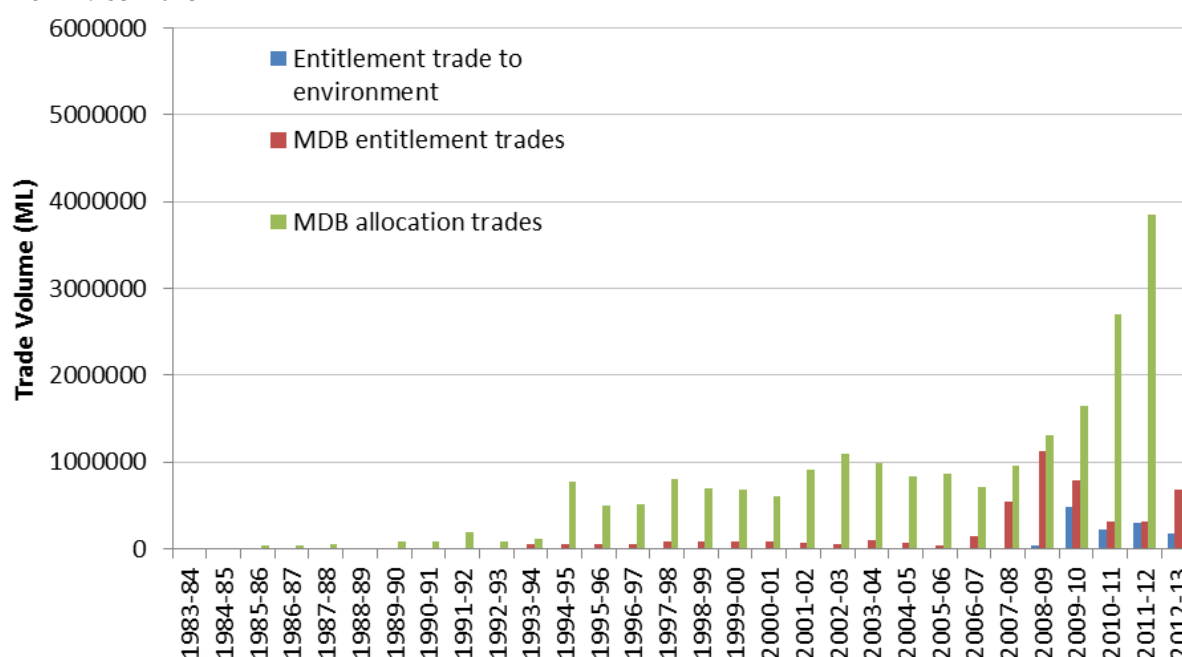
Source: Settre and Wheeler (2017), pp. 402.

Between 1998-99 and 2010-11 water has been largely bought by horticultural irrigators in SA and dairy farmers in Victoria, while sellers of water have predominately been annual cropping irrigators in the NSW Murrumbidgee and the Lower Darling (Wheeler et al., 2014b). Since 2007 significant volumes of water were transferred inter-state to South Australia, where water was bought to maintain supply for permanent high value crops such as vineyards (BOM, 2011; Wheeler et al., 2014b). Figure 3.3 demonstrates the direction of net trades in the most recent water years: 2014-2015 and 2015-2016.

Figure 3.3 Net Trade Volumes in the sMDB in 2014-2015 and 2015-2016, Excluding Environmental Trading
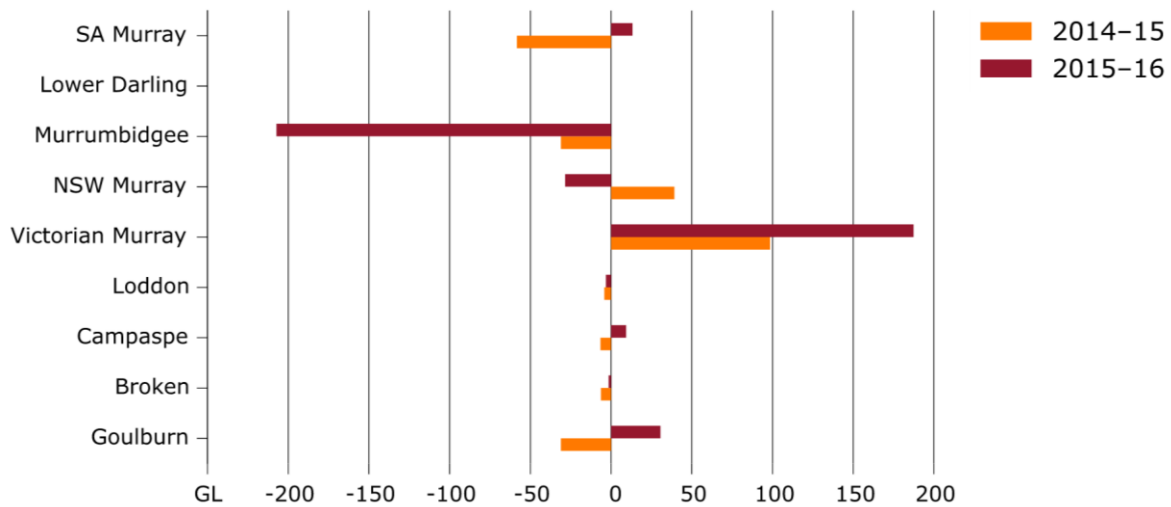
Although water market transactions are possible within the entire sMDB, trading across state boundaries and irrigation districts is complicated by each state's differing legislative histories, naming conventions of water rights, reliability definitions, and rules for tradable water products (as evident in Table 3.2). States have also implemented a range of artificial trade barriers which influence the trade of water entitlements. Some of these barriers are intentionally created protection mechanisms (e.g. minimising permanent trade out of a region) and others are hydrologic in nature. Primary restrictions on trade include (BOM, 2011): a) physical constraints; b) environmental constraints; c) hydrological connections and water supply considerations; and d) low hydrological connectivity. In particular, there have been a number of artificial barriers to trade in the MDB, such as a 4% annual limit on total volume of water entitlements that may be traded out of an irrigation district in Victoria, and a 10% limit on the volume of water entitlement allowed to be held by non-water users (lifted in 2009) (NWC, 2010). A similar 4% annual limit was introduced in NSW, as was a limit placed on some irrigation districts in SA (e.g. Central Irrigation Trust), although this limit was not reached (NWC, 2010). Artificial barriers to trade also affect buy-back for the environment in NSW, where an embargo on water trade to the environment existed from May-December 2009 (NWC, 2010). The example of the NSW embargo on environmental water sales demonstrates that the legacy of the state's constitutional right to water continues in the narrative of environmental water management, as states are naturally cautious about the sale of water to the Commonwealth (Crase et al., 2015). However, as part of the *NWI* there have been efforts put in place to progressively remove the institutional barriers to trade, to facilitate the broadening and maturing of the water market in the sMDB (COAG, 2004).

3.2.4 Irrigator Adoption of Water Markets in the MDB

Water markets have been used informally for over half a century in the MDB, specifically since the Second World War Drought (Wheeler, 2014). In the past few decades, formal water trade has become a vital farm management tool for irrigators in the MDB. For example, water allocation trades are used by irrigators to mitigate their supply risk by purchasing water when it is most needed (Zuo et al., 2014). It is estimated that by the end of 2010, the majority of irrigators in the Basin have engaged in water trade in one way or another, specifically: 86% of irrigators in NSW; 77% in Victoria; and 63% in South Australia (Wheeler et al., 2014b). Adoption of water market by irrigators is not uniform across the Basin, as is evident in Figure 3.4 where it can be seen that NSW allocation trade was adopted earlier and at higher volumes than allocation and entitlement trading in other Basin states (Wheeler et al., 2014b). Wheeler et al. (2009) show that water trading adoption conforms to the expected trend in adopting innovations, with a period of incremental increase followed by a rapid period of growth and an eventual plateau (S-curve). This is also evident in Figure 3.4.

Figure 3.4 Adoption of Water Trading by MDB Irrigators
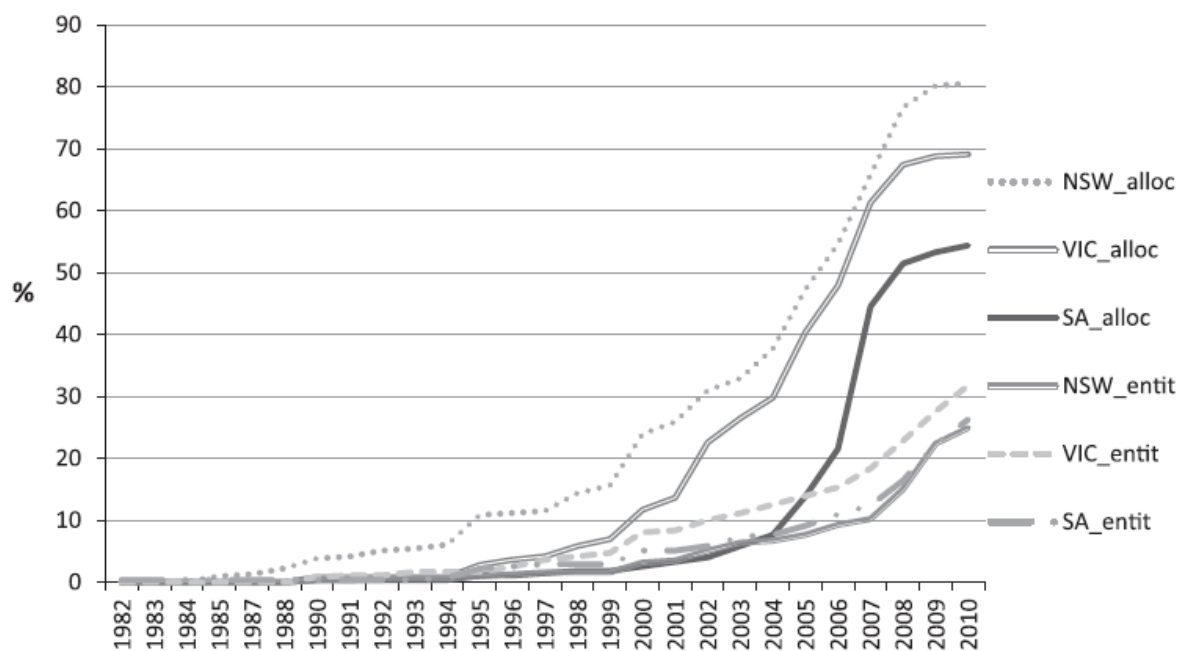
Source: Wheeler et al. (2014), p. 34.

3.2.5 Trends in Australian Agriculture

Water markets in the MDB and the reallocation of water from consumptive water use to the environment has induced a number of structural changes in the Australian agriculture water use. This section provides a brief exploration of how the agricultural and irrigation sectors have changed over the past decade and discusses some of the trends in relation to MDB water policy. However, because there is no counterfactual to test (e.g. without the existence of water markets), it is not possible to test for correlation and causation between water policy and changes in the agricultural sector cannot be

inferred from the data provided below. In particular, it is impossible to disentangle the effect of drought and the effect of water policy on irrigated agriculture. Rather, this section aims to provide a small overview of agriculture in Australia in relation to changing water policy.

Figure 3.5 shows trends in the number of agricultural businesses, irrigation businesses and agricultural water use in Australia. As can be seen there is a significant decrease in irrigation water use from 2006 to 2009, followed by a sharp increase at the break of the Millennium Drought in the southern MDB. There is a slight downward trend in the number of agricultural businesses irrigating from 2006 to 2010, as is highlighted in the wider literature (Haensch, 2017), however this trend does not seem to be evident in the total number of agricultural businesses overall in Australia. This is likely due to the transition of irrigated agriculture (such as dairy farming) to rain fed agriculture (such as meat cattle).

Figure 3.5 Trends in Agricultural Businesses in Australia
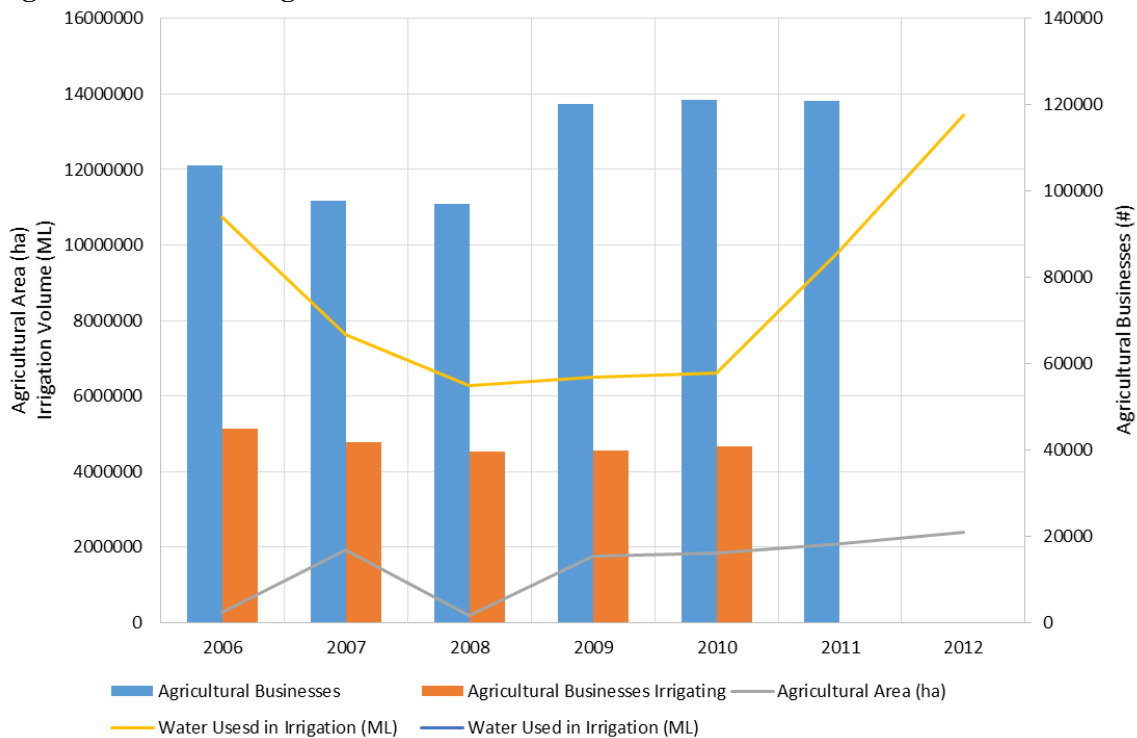
Source: Own figure. Data sourced from DoA (2013), ABS (2011), ABS (2013).

The changes in composition of agriculture in Australia, such as the example of irrigated dairy to dry land meat cattle, is evident in Table 3.4. As can be seen, there is significant changes year to year for annual irrigated crops such as cotton which reaches a minimum number of businesses in 2007-2008 at the height of the Millennium Drought and the passage of the 2007 *Water Act*. However, there is little difference observable overall in perennial crops such as grapes.

Table 3.4 Trends in the Number and Composition of Agricultural Businesses in Australia

Commodity	Number of Agricultural Enterprises in Australia					
	2006	**2007**	**2008**	**2008**	**2010**	**2011**
Grapes	4,650	4,628	4,880	5,536	5,230	4,966
Fruit growing	6,054	6,022	5,089	6,363	6,225	5,952
Vegetable	4,325	4,429	3,670	5,818	4,653	4,481
Grains	102,772	12,735	12,199	13,256	12,555	13,698
Sheep farming	9,329	8,117	8,235	10,705	11,236	11,944
Beef farming	29,411	26,678	25,613	40,864	39,328	38,752
Dairy cattle	8,737	8,373	7,645	8,594	7,573	8,202
Poultry, pig and deer farming	2,116	1,815	1,649	1,807	1,846	1,623
Sugar cane	3,899	3,662	3,703	3,634	3,487	3,476
Cotton	477	293	276	361	853	736

Source: Department of Agriculture (2013), p. 48.

3.3 Environmental Water in the MDB

Environmental water management in the MDB consists of the co-ordinated management of planned (i.e. rules-based or regulated) and held (i.e. adaptive or acquired) environmental water (*Water Act, 2007*). Planned and held environmental water are defined under the *Water Act* (2007), the *Basin Plan Guide* (2010) and the *Basin Plan* (2012), and are as follows:

Planned environmental water: water for the environment prescribed under the rules of a water sharing plan, the *Basin Plan* or state legislation. In regulated rivers, water sharing plans specify the timing, frequency and volume of water to be released or delivered to specific environmental assets. In non-regulated rivers, planned environmental water can be achieved through water take restrictions such as cease-to-pump rules, to ensure a minimum environmental flow. Importantly (and in contrast to held environmental water which may be sold under certain conditions), planned environmental water cannot be used for any other purpose than environmental watering.

Held environmental water: water that is available under a water access right, water delivery right or irrigation right that is held (owned) by an environmental water holder (e.g. CEWH) and managed for the purpose of achieving positive environmental outcomes. Held environmental water is subject to the same property right regime, and allocation and storage rules as consumptive water rights. In the MDB, the vast majority of held environmental water is owned by the federal government and managed by the CEWH for positive public benefit and environmental outcomes.

This thesis is concerned with the acquisition and management of held environmental water in the MDB. For simplicity, held environmental water will refer to the water that is purchased by the CEWO, either through direct water rights purchases (the *RtB)* or water acquired from irrigators through investments in irrigation infrastructure efficiency (the *SRWUIP*). Water that has been recovered under these

programs is classified as 'reallocated' and is used interchangeably with 'held' water to refer to water rights owned by an environmental water holder and used for environmental outcomes. The following section details the approaches available to policy makers to acquire held water for the environment, followed by a rationale for a market-based reallocation approach.

3.4 Methods of Reallocating Water to the Environment

When determining how to reallocate water to higher value and environmentally sustainable uses, policy makers were faced with two fundamentally different approaches, as introduced in Table 3.1. This choice included a) the administrative curtailment of water rights across the board; or b) the introduction of unbundled, tradable property rights for water, water pricing and markets, which can be used to trade water towards higher value uses (Bjornlund, 1998). A third option, conserving water through efficiency upgrades, is also a valid means of reallocating water to the environment, although at present there is a lack of clarity regarding how conserved water is reallocated to the environment (Adamson et al., 2017) and this approach has been heavily criticised, as discussed in Section 3.5. Administrative curtailment, market-based reallocation and conservation through water-use efficiency are discussed below.

Administrative curtailment: a regulatory action that involves restricting abstractions by non-environmental users. Administrative curtailment typically involves setting a cap on extractions, limiting the allocations of new rights and reducing the volume of water taken out of the consumptive pool by extractive water users. Establishing a cap on abstractions is seen by many as a critical step in achieving sustainable water management (Richter, 2014). Administrative curtailment can be compensated or occur with no compensation for the water-use forgone (PC, 2010). The degree of administrative curtailment may be applied based on security (or priority) of consumptive rights (Lund et al., 2014). In the case of the MDB, administrative curtailment of rights in regulated systems was found to be an inefficient option, as it has no mechanisms to acquire water from lowest value uses (PC, 2010).

Conserving water through water-use efficiency: an approach intended to conserve water by increasing the efficiency of water delivery (off-farm) and water-use infrastructure (on-farm) to reduce the amount of water escaping from consumptive use through reduced runoff, seepage, or (Grafton and Williams, 2017). Water use efficiency improvements can also lead to river water quality benefits by reducing the volume of saline or otherwise contaminated return flows to the river (Heaney and Beare, 2003). Efficiency upgrades are funded (or subsidised) by the government and as such the saved water becomes common property to be managed for environmental benefit (i.e. the government indirectly purchases the saved water by subsidising the efficiency upgrade). Examples of efficiency upgrades include installing concrete lining within an unlined irrigation ditch to reduce losses through seepage.

Market-based reallocation: an approach involving the purchase of consumptive water rights (e.g. water entitlements) by an environmental water holder(s), which are then managed for positive social and ecological outcomes. Water-users are compensated for forgoing the water right at the market or

negotiated price. The water right acquisition may be short-term or permanent in nature, as is discussed in Section 3.11. Market-based reallocation occurs on a voluntary basis and water is acquired from willing sellers, and can be used strategically to acquire water from low-value uses to minimise negative economic impacts on aggregate farming profits (Quiggin et al., 2012). When water rights are fully appropriated or where a cap on extractions is established, reallocating water through voluntary and compensated market-style transactions is often more politically palatable than administratively curtailing consumptive water rights.

The choice of recovery mechanism between these options depends on who is responsible for the costs of reallocation. If the government is to be responsible for the cost, voluntary compensated reallocation mechanisms of rights is the appropriate option because as the purchase cost of the water entitlement purchases will be borne by the government (PC, 2010). Conversely, if private irrigators are responsible for the cost, then administrative curtailment with no compensation may be the appropriate choice because the cost of forgoing water will be borne by the irrigators (PC, 2010). Each method of environmental allocations influences the level of environmental protections, the risk assumed by the environment, and the capacity for adaptive management (Horne et al., 2017). Regulatory and property right allocation mechanisms can be, and often are, used in combination with one another. Evidently, water policy in the MDB exhibits characteristics of all these approaches.

3.5 The Rational for Market-Based Water Reallocation

The efficiency and effectiveness of both market-based and infrastructure-based methods of water reallocation in the MDB have been questioned (Crase and O'Keefe, 2009; Loch et al., 2014a). Overall, market-based entitlement buy-back is widely considered to be the most efficient method of reallocating water to the environment (Grafton et al., 2010). Conversely, after accounting for third party impacts of infrastructure upgrades on downstream users and the possibility of a re-bound effect, net gains from efficiency improvements may be marginal or detrimental to the provision of ecosystem services other than food production (Gordon et al., 2009). The following section discusses these issues and presents the rationale for a market-based approach to environmental water acquisition.

3.5.1 Water-use Efficiency

The concept of efficiency has a number of variants (e.g. allocative efficiency, technical efficiency, basin efficiency) and definitions can vary between academic disciplines (Qureshi et al., 2011). For the purpose of this thesis, water-use efficiency is taken to mean irrigation water-use efficiency. Agricultural production research has long placed a strong focus on output-orientated efficiency based on the adage "more crop per drop" (Giordano et al., 2017). It is unsurprising therefore that an efficiency approach has been adopted to manage reductions in consumptive water use in favour of higher environmental water availabilities.

Reallocating water to the environment through increases in irrigation efficiency is based on the premise of technical efficacy gains (Adamson and Loch, 2014). Technical efficiency refers to the ability to minimise input use (e.g. water) within the production of a set of given outputs (e.g. crops), or conversely to maximise outputs for a given set of inputs (Kumbhakar and Lovell, 2000). Thus, by increasing technological efficiency, it is assumed possible to save water that was previously lost to non-beneficial water uses (e.g. evaporation) and allocate the saved water to the environment (Grafton and Williams, 2017). In the MDB, this approach is used as a key rationale for the *SRWUIP*, through which water savings are expected to occur via improvements in irrigation efficiency. The 'saved' water in turn becomes available for environmental use (Cummins and Watson, 2012). At face value, improved technical efficiency appears to be a viable means to save water and reallocate it to the environment, through a reduction in water losses (Qureshi et al., 2011). However, there are a number of caveats and limitations to this approach which refute this conclusion.

Namely, investment in technical efficiency has been shown to: reduce agricultural return flows, aquifer recharge and runoff; prompt over-investment and under-utilization of irrigation infrastructure; create technological lock-in and reduce irrigation capacity for adaptation; reduce supply availability, reliability and quality for downstream and environmental water users and; reduce flow dependent ecosystem services (Ward and Pulido-Velazquez, 2008; PC, 2010; Qureshi et al., 2011; Adamson and Loch, 2014; Loch and Adamson, 2015; Grafton, 2016; Grafton and Williams, 2017; Adamson et al., 2017). The effects listed above combine to create the "hydrological paradox" (Grafton and Williams, 2017, p.3) which states that increases in water-use efficiency at the farm level can cause overall reduced availability at the basin scale. In addition, there is also general lack of clarity regarding how water saved through irrigation efficiency upgrades will be quantified and protected for environmental use (Adamson et al., 2017).

There is considerable evidence to suggest that increases in water use efficiency can have the adverse impact of increasing water resource depletion (Adamson and Loch, 2014; Loch and Adamson, 2015). This is explained by the Jevons' Paradox[9] (Jevons 1865), also referred to as the 'rebound' or 'backfire' effect (Alcott, 2005; Loch and Adamson, 2015), which describes higher consumption in response to efficiency improvements, instead of the expected contraction of resource use (Alcott, 2005). In the case of the MDB, improvements in water-use efficiency can prompt irrigators to over-capitalise in on-farm capital investment or an expansion of cropping area (Adamson and Loch, 2014), especially if they are 'allowed' to keep the conserved water (PC, 2006). This has been estimated to cause higher land and

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[9] In "The Coal Question" (1865), William Jevons suggested that technological efficiency gains in coal production will increase, rather than decrease, the consumption of coal and other industrial resources (e.g. iron). This challenged widely held (and still implemented) policies that efficiency gains would reduce resource consumption by saving losses (Alcott, 2005). In the energy and water sectors, the Jevons Paradox has profound implications for current methods of adapting to climate change and implies higher costs of adaption and undermines sustainability methods which focus on energy reduction through use efficiency improvements (Sorrell, 2009).

water consumption, particularly in the nMDB (Loch and Adamson, 2015), which can expose irrigators who over-invest in farm capital or irrigated areas to more risk during drought when supply decreases and production is unable to be maintained (Adamson and Loch, 2014).

3.5.2 Cost-effectiveness

An additional issue associated with recovering water through efficiency upgrades is the high cost of this approach. Water acquired by the federal government and used for public environmental benefits is paid for with public funding. Expenditure on water recovery must therefore be efficient and defensible to ensure the Australian community receive the highest net return on money spent in the Basin.

Initially it was conceived that additional water should be sourced from infrastructure improvements only when the cost of doing so is consistent with the market price of water saved (Quiggin et al., 2011). However, using empirical data, the Productivity Commission estimated that acquiring water through infrastructure may be four times more expensive than acquiring water through entitlement purchases, resulting in a premium of $7,500/ML of water acquired through infrastructure improvements (PC, 2010). For example, in the Sunraysia Modernization Project and Northern Victoria Irrigation Renewal Project, expenditure of $14,714/ML (Grafton, 2016) and $10,000/ML (PC, 2010), respectively, has been spent to acquire water through infrastructure. These figures do not compare favourably to expenditure of $2,382/ML for high reliability water entitlements acquired in the Goulburn through the RtB tender process (PC, 2010) or the approximate long-term average price of $2,000/ML able to be acquired through the market for entitlements. In addition, Loch et al. (2014) provide a more detailed examination of mean expenditure per mega litre of water acquired for the periods 2004-2009 and 2009-2012. The results are shown in Table 3.5. As can be seen, Loch et al. (2014) found that water recovered from efficiency-based methods is 2.0 to 3.3 times more expensive per mega litre than that acquired through market-based recovery.

Table 3.5 Water Recovery Expenditure as Mean $/ML

Period	Mean $/ML	
	Market-based recovery	**Efficiency-based recovery**
Weighted mean 2004-2012	$1,527	$5,109
Weighted mean 2009-2012	$1,599	$3,302

Source: Adapted from Loch et al. (2014), p. 401.

In essence, Loch et al. (2014) demonstrate that efficiency-based recovery is less efficient at turning money into water than a market-based approach. As a result of the high purchase price of water recovered through efficiency upgrades, it remains questionable as to whether the current budget allocated to water acquisition will be able to acquire the remaining volumes through infrastructure investment. Conversely, for the current budget of $3.1 billion and a range of climate scenarios there is

sufficient money and water available to meet the SDL goals through entitlement purchases[10] (Adamson, 2015). If public money is to be used for the public benefit, then expenditure should be allocated to the projects that create the highest marginal recovery of water. Therefore, there is a strong argument to be made for the acquisition of environmental water through market-based, rather than efficiency-based, reallocation programs.

3.6 Adaptive Market-Based Reallocation as a Driver of Ecological Health

Environmental water protection, in cases where it exists, has typically been managed though the provision of an ecological reserve or a base-flow for the environment (Poff et al., 1997). The base-flow approach stipulates a minimum volume of water to remain in the river at all times throughout the year, but does not necessarily give consideration to the management of flows above this volume. This approach was, and still remains, a result of a vastly prevailing mentality that overtopping a river bank to fill wetlands or seep into groundwater through floodplains is a waste of the resource (Kingsford, 2000; Lane-Miller et al., 2013). In the MDB, this is evident in the conception of floods as 'surplus flow', natural river catchments as 'drainage' and groundwater seepage as 'losses' (Kingsford, 2000). Planned or rules-based environmental water (as described in Section 3.3) in the MDB is an example of the minimum flow requirement approach to environmental water protection.

As ecological science has progressed it has become evident that the base-flow approach is not sufficient to maintain the ecological integrity and resilience of flow dependent ecosystems. Rather, it is now known that all aspects of the flow regime (magnitude, frequency, duration, timing, rate of change) contribute to ecological health and resilience. Specifically, overbank flooding (also known as a flood pulse) of sufficient magnitude and variability is the primary determinant of the ecological health in hydrologically connected river-floodplain ecosystems (Junk et al., 1989; Poff et al., 1997). Figure 3.6 demonstrates the relationship between timing, volume, area inundated and ecological effects of overbank flooding.

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[10] For normal state of nature this would be achieved through a combination of high security (184GL), general security (1,876GL) and supplementary water (3,017GL) (Adamson, 2015).

Figure 3.6 Overbank Flood Magnitude, Frequency and Impacts
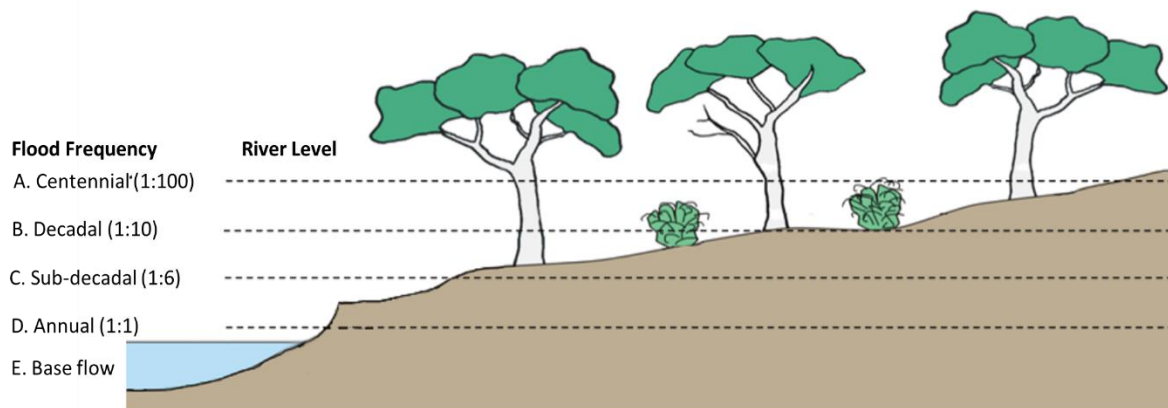
A. Centennial floods cover whole floodplain, can uproot trees and deposit them in channel, creating fish habitat
B. Decadal floods inundate mature floodplain forest and maintains late successional species
C. Periodic flooding inundates low-lying floodplains inundated and carry nutrients back to the channel, riparian vegetation maintained for bird habitat
D. Annual flooding transports fine sediments to the channel and creates fish spawning habitat in adjacent wetlands
E. Base flow is insufficient to maintain riparian vegetation or inundate floodplain

Source: Adapted after Poff et al. (1997) and CSIRO (2012).

As shown in Figure 3.6 flood volumes at various time scales are needed to inundate floodplain areas. One way to achieve these flows is to use an adaptive market-based reallocation approach, whereby water can be annually reallocated to the environment when it is needed most (Kirby et al., 2006; Connor et al., 2013; Wheeler et al., 2013a). For example, Connor et al. (2013) demonstrate that an annually trading EWH can reduce ecological damage to floodplain wetlands by increasing the frequency of moderate flooding events and during drought conditions where ecosystems would benefit most from flooding.

To demonstrate the potential application and benefit of an annually adaptive market-based approach, Figure 3.7 shows a 30-year hydrograph at Maude weir in the Murrumbidgee catchment, along with a hypothesis of how the hydrograph may be altered for improved ecological outcomes through trade. As can be seen, annual trading could be employed to: a) extend the duration of moderate flow events; b) extend the volume of naturally occurring high flow events; c) mitigate the effects of isolated low flow years; d) generate profits by selling allocations after flood events when water is less needed by the environment; and e) generate profits in moderate flow years to finance purchases in future low flow scenarios. These concepts are treated empirically and thoroughly explored in Chapter 7. In addition to annual environmental water trading opportunities, there are a number of intra-annual market-based mechanisms, such as split-season leasing, which can be employed on a seasonal basis to meet dual irrigation and environmental outcomes. Alternative water products and their merits are described in full in Chapter 8. Overall, the purpose of this discussion is to highlight the importance and benefit of an annually adaptive approach which can be achieved through the use of market-based reallocation tools.

Figure 3.7 Hydrograph of Flow at Maude Weir with and Without Environmental Water Trade
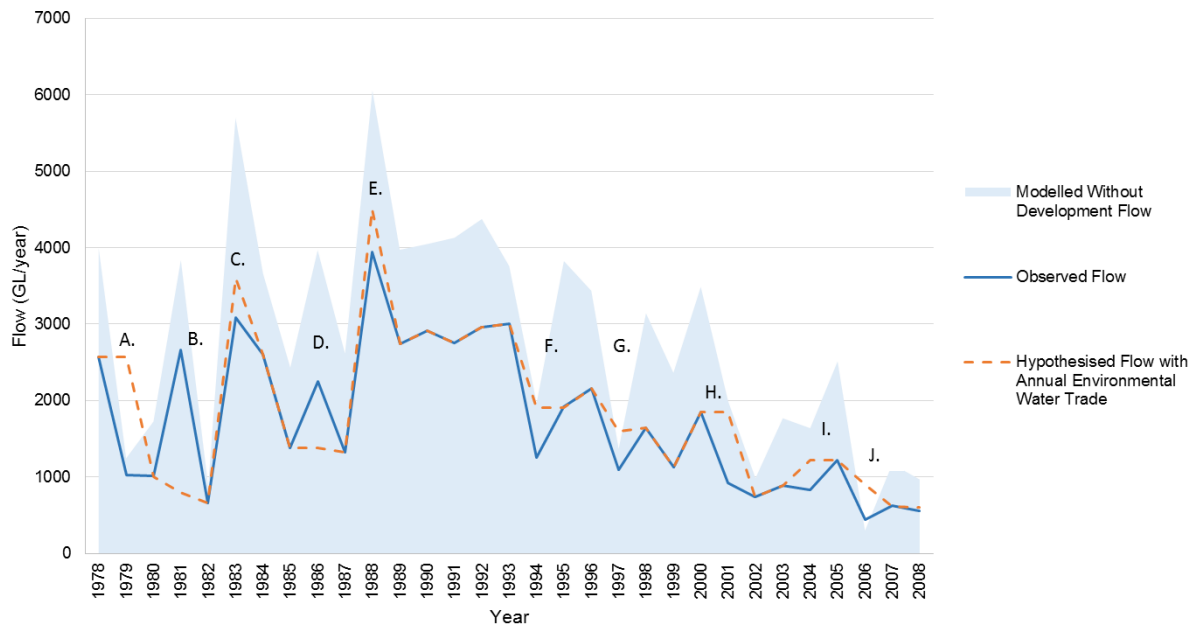
A. and H. Buy allocations after moderate flooding to extend flood duration.
B. and D. Sell allocations when less needed by the environment following previous flood events.
C. and E. Buy allocations to extend peaks of naturally occuring high floods ("piggy-back flows").
F. and G. Buy allocations to mitigate the effects of isolated low flow years.
I. and J. Use profits from previous allocation sales to finance purchases in low flow years.

<u>**Source:**</u> **Own Figure. Observed and modelled flow data provided by CSIRO.**

3.7 Conditions for the Environment Entering the Water Market

In the previous section, it was argued that direct market-based acquisitions are the most efficient and cost-effective means of acquiring water for the environment. However, for water rights (either entitlements or allocations) to be acquired through tenders or markets, a number of conditions must first be met. The conditions include:

a) Establishment of rights to and limits on freshwater extraction;
b) Recognition of the environment as a legitimate water user; and
c) Authority to transfer existing water rights to an environmental purpose (Garrick et al., 2009):.

In addition to these conditions, acquiring water rights for the environment requires basic market principles to be satisfied, such as well-defined, tradable and legally defendable property rights. Poorly defined property rights is a key limitation to instream flow acquisitions (Scarborough, 2012). Further discussion on natural resource property rights can be found in the seminal work by Schlager and Ostrom (1992). Bjornlund and McKay (2002) also highlight the importance of:

a) Effective channels to communicate market information to water-users;
b) Removing spatial and intuitional obstacles to trade;

c) Quantification of total resource use, including environmental water demands and unused[11] water rights;

d) An understanding of the legacy of past and current policies, religious values and cultural norms;

e) Balancing private market forces and government regulation; and

f) Formal specification and registration of all water rights and trade policies

Among the most fundamental conditions listed above is the recognition of the environment as a water user it its own right. Globally, the recognition of the environment as a legitimate water user remains on a relatively small scale. However, growing environmental awareness and a raft of changes to water laws signal a slow transition to legitimising the environment as a beneficial user. Examples of legislation progressively recognizing the environment as a legitimate water user are: the introduction of an ecological reserve flow in South Africa's 1998 water reform (Nieuwoudt and Armitage, 2004); the 2005 reforms to the Chilean Water Code allowing the state to reject new water right appropriations on the basis of ecological protection (Bitrán et al., 2011); the *'New Start for Fresh Water'* reforms in New Zealand (Ministry for the Environment, 2009); the establishment of the CEWO and CEWH in Australia; the provision of a pulse flow for ecological benefit in the Mexican reaches of the Colorado Basin (King et al., 2014); and the state-based and public recognition of beneficial instream use in the western US (Loehman and Charney, 2011).

A second fundamental condition for water to be bought and sold for the environment is the existence of an environmental water holder (EWH) with capacity and authority to act on behalf of the environment. In existing environmental water markets (discussed further in Section 3.9) EWHs have evolved at variety of institutional levels including the public, private and non-government sectors. A discussion on the types of EWHs is provided in Section 3.8.

Once the above conditions have been met water rights may be acquired for the environment under certain conditions. To assess the efficiency, effectiveness and equity of environmental water requirements, Wheeler et al. (2013) provide the framework presented in Table 3.9. Table 3.9 examines the management of environmental water from an irrigators' (because they are the largest sellers of water) and EWHs' perspective. For the purpose of this thesis, criteria applying to water acquirers (column two) and management implications for EWHs are most relevant for the subsequent analysis.

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[11] In the MDB, un-used water rights are called "sleeper" licences and under-used rights are called "dozers".

Table 3.6 Environmental Water Acquisition Criteria

Indicators	Water Acquirers	Water Sellers	Management
Efficiency	Least-cost path to buy environmental water (price)	Most beneficial path to increased welfare	**Irrigator:** sell when monetary benefits outweigh production profits **EWH:** sell water when monetary benefits outweigh environmental demand
Effectiveness	Secure adequate volume (quantity) Willingness to participate in the market Transaction costs of acquisition	Retain right to water for future use if desired Willingness to participate in the market Transaction costs of sale	**Irrigator:** sell water in accordance with farm plan **EWH:** optimise portfolio of rights and delivery of water **Irrigator, EWH:** Ability to operate in the market Transaction costs of management
Equity	Minimise third-party impacts Avoid stranded assets Facilitate restructure	Minimise third party impacts	**Irrigator:** no harm principle **EWH:** Minimise market distortion in both acquisition and management phases Contribute to irrigator infrastructure when shared for environmental deliveries

3.8 Environmental Water Holders (EWHs)

An environmental water holder (EWH) is an organization with a mandate to acquire and manage water rights for positive environmental outcomes. The existence, legal jurisdiction and effectiveness of an EWH are subject to market enabling conditions and institutions at federal, state and local levels. EWHs can be public (e.g. an arm of the government), private (e.g. conservation investment funds) or non-profit (e.g. environmental non-governmental organizations), or combinations of these. There are capacity trade-offs between public, private and non-profit EWHs depending on the wider institutional context influencing water management, as discussed in the below section and also in Chapter 8.

Federally-led water purchases have the advantage of funding availability and supporting institutions to promote the economically efficient acquisition of water. For example, in Australia, federal leadership in water reallocation has resulted in considerable budget for purchasing water, around $3.1 billion, a sum which would likely be unmatched by state-led programs, private enterprise or NGOs. An additional advantage of federally led reallocation is that because water is purchased with public money (through tax revenue, for example), the government is beholden to the public to use it in a way which maximises social and environmental benefit for public good. Ideally, this results in the dual benefit of transparent government expenditure and efficient allocation of funds to reallocate water to the environment.

There are also advantages derived from NGO involvement in environmental water markets, such as flexibility to develop relationships with irrigators and build community trust. In situations where there is reluctance to sell water to the state or federal governments (see Chapter 8), there is an opportunity for private enterprise to enter the market and acquire water in place of the government. Flexibility in acquisition strategies and ability to work out case-by-case, as consistent with subsidiary theory regarding water management (Garrick et al., 2011), are some benefits of this approach. Further, in cases where political will to reallocate water to the environment is lacking or environmental objectives are not recognised, NGOs can potentially play a unique role by filling this gap. The role of private individuals and NGOs participating in the market is governed by local institutions and laws. In the US, for example, state-based legislation varies considerably regarding the ability for private entities acquiring rights for the environment. In the state of Colorado, a private entity is allowed to acquire water for environmental water rights but the acquired right must be donated to the governing state body (Scarborough, 2012). Comparatively, in California water rights may be purchased and managed for the environment by a private individual (Mooney et al., 2003; Scarborough, 2012). In Australia, the private purchase of water rights for the environment occurs on a relatively small scale. However, where it does exist there remains a number of unresolved questions regarding NGO EWHs, specifically the question of financial sustainability given this is ongoing and vital to the success of market-based programs. A further discussion, literature review and novel findings about the role of NGO EWHs and their use of the water market are presented in Chapter 8.

3.8.1 Environmental Water Holders in the MDB

Within the MDB, EWHs operate at the level of federal, state, private, NGO and Aboriginal Nations. Table 3.7 summarises the types of EWHs currently active in the MDB.

Table 3.7 Examples of Environmental Water Holders in the MDB

Institutional level	Example	Program (commencement date)
Federal	Commonwealth Environmental Water Holder	*RtB Program* (2007-08)
	The Living Murray	*The Living Murray* Program (2003)
State	NSW Office of Environment and Heritage	*River Bank* (2005)
	Victorian Environmental Water Holder	-
Private Sector	Kilter Rural	MDB Balanced Water Fund (2015)
Non-profit Sector	Nature Foundation SA	*Water for Nature* (2008)
Aboriginal land and water rights	Ngarrindjeri Regional Authority (NRA)	CEWH and NRA Watering Agreement (2015)

Source: Own Table.

The CEWH is the largest owner of environmental water in the MDB and can trade water allocations for the environment (CEWO 2014; 2016, see Section 3.11.3). However, as shown in Figure 3.8 for the 2015-2016 water year, the CEWH contributes a relatively small portion of net interregional trade, indicating that CEWH environmental transfers occur primarily within regions (ABARES 2017).

Figure 3.8 Net Environmental Allocation Trades in the sMDB 2015-2016
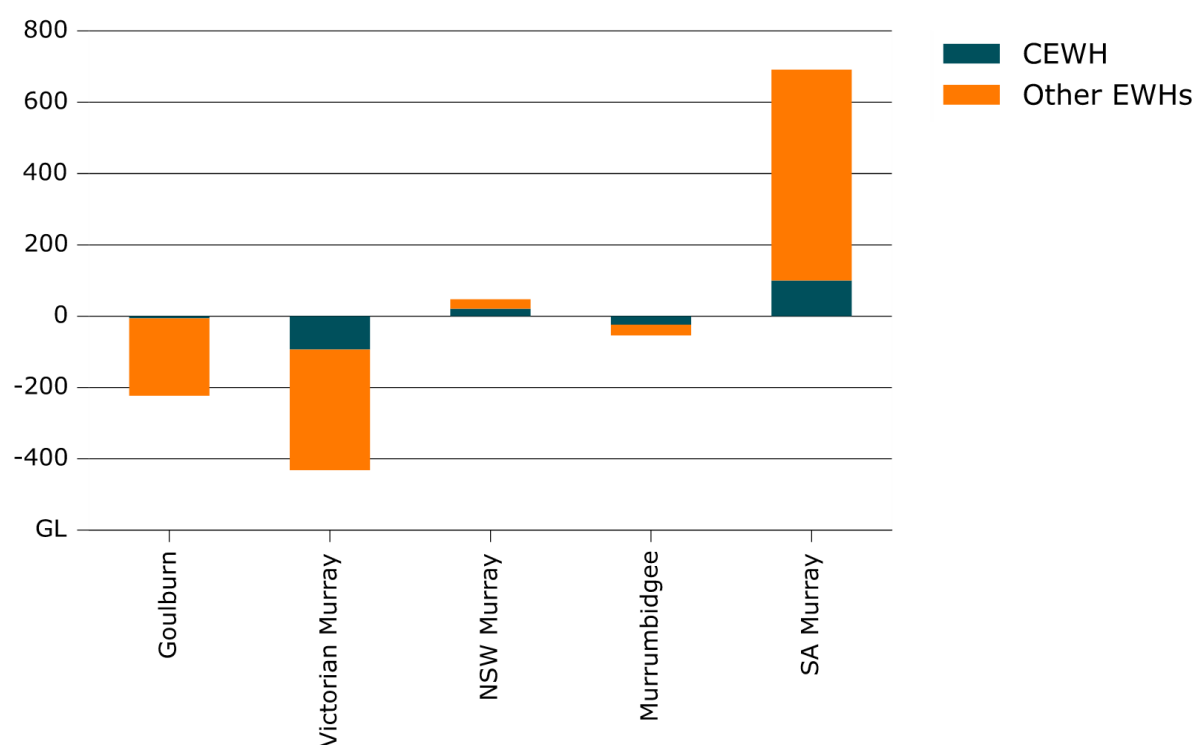
At a state level the NSW Office of Environment and Heritage and the Victorian Water Holder operate as EWHs in partnership with the CEWH. Both state-based EWHs have acquired a moderate level of water rights (NSW OEH, 2015; VEWH, 2016). At present there is no state-based EWH operating in SA due to a number of challenges regarding water storage, some of which are detailed further in Chapter 4.

Non-profit non-government organizations, often referred to as 'water trusts' (King, 2004) also operate on the fringe of the environmental water market in Australia. For example, The *Nature Foundation SA* manages a modest holding of approximately 35ML of water, which it delivers in partnership with the CEWH to environmental assets in the Coorong, Lower Lakes and Murray Mouth region. Although NGO engagement in the Australian water market has been modest to date, there is evidence to suggest that there may be scope and benefit of expanding this aspect of the market, as discussed in detail in Chapter 8.

Recent events in October 2016 suggest that private EWHs in the form of environmental impact investors have recently become engaged in the MDB water market. Private EWHs are distinguished from non-profit EWHs as private firms seeking to generate a profit for their investors. An example of this kind of private EWH is the Murray-Darling Basin Balanced Water Fund (MDB BWF) administered by Kilter Rural Investments and directed by The Nature Conservancy (TNC). The MDB BWF raised $22 million in equity and $5 million in debt to be used for the purchase of water entitlements in their

first capital drive to purchase water entitlements (Kilter Rural, 2016a). Each year 20% of the allocations from the MDB BWF portfolio are to be dedicated to environmental flows predominately on private lands, while the remaining 80% is to be sold on the market to generate revenue for investors (Richter, 2015 pers. comm.). Water acquired under the MDB BWF does not contribute towards the water recovery to meet *Basin Plan* objectives, although possible collaboration between the CEWH and managers of the MDB BWF has been flagged (Kilter Rural, 2016b).

Aboriginal water ownership and the role of Aboriginal groups as EWHs have been heavily debated in Australia. Water management in the MDB to date has fallen short of Aboriginal aspirations (Tan and Jackson, 2013). However progress is evident in the development of an agreement between the CEWH and the Ngarrindjeri Regional Authority (NRA), which formally provides for the transfer, delivery and monitoring of Commonwealth environmental water in the Lower Murray by the NRA (CEWH and NRA, 2015). The agreement also enables NRA participation in SA State water planning. Although Australia has seen a shift in the past decade in giving increased precedence to environmental flows, the next paradigm shift may be to recognise, in law and in practice, the importance of Aboriginal participation in water allocation decisions to maintain the social, cultural and ecological health needs.

3.9 Examples of Existing Environmental Water Markets

Some existing water markets have expanded as vehicles to provide the transfer of water from consumptive uses to the environment (Debaere et al., 2014). Key examples found outside of Australia occur in the western United States and northern Mexico. Before providing a detailed examination of the *RtB* acquisition in the MDB, it is beneficial to explore a few international cases to provide an international context. The following section provides a brief case study of the Columbia River Basin Water Transaction Program and the Colorado River Delta Restoration Program.

3.9.1   The Western United States

The mostly arid wester US water economy is characterised by scarce and unevenly distributed water supplies, rapidly growing populations and increasing urban demands for water (Booker and Young, 1994). In this mature water economy the capital cost of new supply infrastructure is prohibitive and alternative approaches such as agricultural water rights, water pricing and water markets provides an alternative management approach. Where present, water markets are predominately limited to the semi-arid and arid western[12] states where scarcity pressure and changing demand profiles necessitates the movement of water between agriculture, cities, industry and the environment. Water markets function within the wider context of western US water law, which rest on the pillars of prior appropriation and beneficial use. Prior appropriation assigns water to users based on the seniority of their right, which is determined by the date the water was appropriated (Tarlock, 2001). This guarantees that senior

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[12] Western states of the US include: Washington, Oregon, California, Montana, Utah, New Mexico, Wyoming, Arizona, Idaho, Nevada, Colorado.

beneficial users are last to lose supply during periods of low flow, so long as they maintain a beneficial use of the water without prolonged interruptions (Garrick et al., 2011). Traditionally, beneficial use has referred to water used for productive means such as diversions for agriculture, mining, industry, domestic and urban use (Clayton, 2009).

The emergence of the environment as a legitimate water user and the subsequent environmental water transactions[13] (EWTs) in the Western US have been the result of ongoing legislative reform and an increasing recognition of the importance of water for the environment. The environment has been incorporated into the water rights system as a legitimate water user, initially though avenues of common law such as the *Public Trust Doctrine* (Garrick et al., 2009). Based on the *Public Trust Doctrine*, private water rights holders do not have the right to infringe on the quality of public resources held in trust by the state (Mooney et al., 2003). Administrative rulings with their roots in the *Public Trust Doctrine,* have been used to appropriate water for the environment as a beneficial use. Federal level legislation such as the *Endangered Species Act* (1973), which prohibits federal agencies from carrying out actions that threaten the populations of endangered species and places endangered species the highest priority in water use (Ward and Booker, 2003) (and to a lesser extent the *Clean Water Act* (1972)) have also driven progress in administrative and market-based reallocation to ensure minimum levels of environmental flows for endangered aquatic species are met.

State-based legislation is also paramount for enabling the protection and transfer of water for instream use. The US states of the Columbia River Basin were among the first to take the legal leap in establishing environmental flow provisions (Neuman et al., 2006). For example, laws allowing administrative designations of minimum or instream flows were passed in Washington (*Minimum Water Flows and Levels Act*, 1967) and Idaho (*Minimum Stream Flow Act*, 1978) followed by *The Instream Water Rights Act* in Oregon (1987). The degree to which market-based transactions can be used as an environmental management tool is subject to the maturity of the supporting water market institutions and state or regional instream flow legislations (Garrick et al., 2011). As a result, institutional development of water markets in the US has progressed unevenly owing to differing state-based rules, local institutions and limitations on water trading, meaning that water management can remain governed by many different formal and informal institutions which interact across the landscape.

In regions of the US where institutional arrangements facilitate water trade and the environment is recognised as a legitimate and beneficial water user, EWTs can occur. An environmental water transaction (or instream flow transaction) is the voluntary reallocation of water from consumptive use to the environment. EWTs occur within a market framework where water rights holders forgoing water-use are compensated at a market or negotiated rate, and they have emerged as a small but non-trivial

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[13] Note that in this context, environmental water trades, transfers and transactions are used interchangeably.

segment of the water market. From 1990 to 2003, EWTs accounted for 17% of volume leased (4,785GL) and 16% of volume sold (234GL) in the western states of the US (Brown, 2006). Commercial water market literature suggests that EWTs account for up to 40% of total volume of trade in the western US (WestWater Research Inc., 2014), although this result is difficult to validate. Environmental water transfers or products are typically traded at lower prices per acre foot compared to agricultural water trades, owing to the short-term nature of the transfers (Dohety and Smith, 2012) and the derivation of compensation price through negotiation rather than mature markets. Agricultural water rights holders are the primary sellers of water to the environment, and state and federal agencies are the largest buyers by volume (Hanak and Stryjewski, 2012). EWTs can differ by volume, duration, timing and individual contractual agreements. Types of water transfers carried out for the environment are explored in detail in Chapter 8.

3.9.1.1 The Columbia River Basin Water Transactions Program

A key example where state- and federally-based laws have enabled market-based water recovery to protect *Endangered Species Act (1973)* listed salmon species is the Columbia River Basin. The Columbia River Basin is located in the Pacific Northwest covering areas in Washington, Oregon, Idaho, Nevada, Wyoming, Utah and Montana in the US and British Columbia in Canada, covering an area of 258,000 square miles (66,821,693ha) (Leopold, 1994). The headwaters of the Columbia River are located in British Columbia, and the river flows southwest into the US. The Columbia River is heavily dammed for hydropower and flood control purposes. Dams are administered by Bonneville Power Administration (BPA), a government department that markets wholesale electrical power generated by federal hydroelectric projects. Figure 3.9 shows the Columbia River Basin and demonstrates the location of federal and non-federal dam projects. Extensive literature regarding the impact of extractions and damming on native salmonoid species has been developed, and the findings are thoroughly detailed by the Committee on Water Resource Management, Instream Flows and Salmon Survival, in the Columbia River Basin et al. (2004).

Figure 3.9 The Columbia River Basin

The Federal Columbia River Power System Biological Opinion (US-FWS, 2000) mandates that the operation of the Federal Columbia River Power System does not jeopardise the continued existence of salmonoid listed under the US *Endangered Species Act* (1973). One means by which the BPA funds fish and wildlife restoration activities in the Columbia Basin is via the funding of the *Columbia River Basin Water Transactions Program* (CBWTP). The CBWTP specifies a set of Qualified Local Entities (QLE) who are able to buy water on behalf of the environment. As part of the CBWTP, extensive permanent acquisitions, irrigation efficiency projects, temporary water transactions (annual, short-term leases) and split-season leases are used to return water to the river in permanent and critical periods. Table 3.8 summarises volumes traded to the environment in the most active sub-basins of the Columbia River Basin between 2003 and 2010, totalling 8,521GL at a cost of $20,374,545 (noting that the total volume may not represent permanent reallocations to the environment).

Table 3.8 Water Recovered, Recovery Budget, and Expenditure for 10 Active Water Trading Sub-basins in the Columbia River Basin from 2003-2010

Sub-Basin, State	Average Annual Water Recovered (GL/year) 2003-2010	Total Water Recovered (GL/year) 2003-2010	Total Program Budget 2003-2010 (USD 2007)
Bitterroot, Montana	189.32	1,515.42	$975,553
Blackfoot, Montana	87.51	701.01	$942,076
Deschutes, Oregon	294.69	2,359.31	$7,121,465
Grande Ronde, Oregon	22.33	179.49	$829,389
John Day, Oregon	50.90	408.99	$1,337,186
Salmon, Idaho	151.81	1,215.37	$1,066,464
Umatilla, Oregon	21.43	1,72.35	$820,454
Upper Columbia, Washington	57.15	459.00	$1,844,845
Walla Walla, Washington	69.65	557.23	$2,012,633
Yakima, Washington	1,18.77	952.83	$3,424,480
TOTAL	1,063.56	8,521.01	$20,374,545

Source: adapted from Garrick (2015), pp. 143-145.

The QLEs in the CBWTP are a mixture of state, river basin conservancies and NGOs, and provide a successful example of nested governance arrangements in a market-based water management context. Similar to Australia, state and federal agencies remain the primary agencies acquiring instream water rights in the US, with federal funding the principal source for NGO water acquisitions (Scarborough, 2012). Overall, the majority of this water acquired for the environment is purchased by federal agencies, including the Bureau of Reclamation and the US Fish and Wildlife Service.

In addition to the CBWTP, environmental water transfers occur in many other instances across the western US at a regional and patchwork level to varying degrees of success. Environmental water acquisitions, expenditure, volume, frequency of trade and type of right vary significantly across states and between basins (Scarborough and Lund, 2007). Notably in recent years there has been notable growth in philanthropic and fundraising organizations fuelling environmental water market activity in the USA and Mexican reaches of the Colorado Basin, as discussed in the following section.

3.9.1.2 The Colorado River Delta Restoration Program [14]

The Colorado River Basin partly covers seven US states and two states of Mexico. The headwaters of the Colorado are in the Rocky Mountains, Colorado, US, and the watershed discharges in the Colorado River Delta in Baja California, Mexico. Within the US, the Colorado River is separated into the Upper and Lower Basins as shown in Figure 3.10. For the purpose of this brief case study, attention is focused to the flows across the border from the US into Mexico and the restoration of the Colorado River Delta

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[14] The target restoration site (the Delta) and the water market used to acquire water for reallocation is located in Mexico. However, considerable bi-lateral agreements (e.g. Minute 319) and American expertise has been dedicated to this project and is therefore explored within the context of the western US water market case study for the purpose of this thesis.

through bi-lateral agreements and acquisitions of water for instream base flows. Colorado River water is shared between Mexico and the US according to the 1944 *Treaty for the Utilization of the Colorado and Tijuana Rivers and of the Rio Grande* (the 1944 Treaty), which was developed to resolve boundary and water allocation issues exasperated by increasing number of water users on either side of the border (Umoff, 2008). A thorough analysis of the treaty is provided by Umoff (2008).

Agriculture in the Colorado Basin is almost completely dependent on irrigation (as oppose to rain-fed farming) and the Colorado River is the main source of water for irrigation, as well as for consumptive supply in Las Vegas (Booker, 1995). As a result, with the exception of unusually wet years, very little water has reached the Colorado River Delta since 1960 as up to 90% of water is diverted from major upstream dams in Arizona, Nevada and California (USA), and the remaining 10% diverted in the Mexicali Valley (Mexico) before reaching the Delta (Flessa, 2001). The environmental implications of high water usage on both sides of the border are exasperated because the *1944 Treaty* does not provide adequate protection or allocation of ecological watering needs (Umoff, 2008). In part to address ecological degradation in the Colorado River (among a number of other critical issues, see: King et al. (2014)) the Mexican reaches of the Colorado were established as the site of an internationally unique binational agreement, known as *Minute 319. Minute 319* is a binational partnership incorporating US and Mexican federal governments, binational NGOs and US water providers, which aims to manage the river as a shared resource by sharing shortage and surplus (King et al., 2014).

A key provision of *Minute 319* is the delivery of a 'pulse' and 'base' flow to the Colorado River Delta. The pulse flow is a once off flow of 105,392 acre-feet (126GL), jointly provided by the US and Mexican governments, including the intentionally created Mexican allocation (ICMA) generated by conservation activities in the Mexicali Valley, released from Lake Mead. The pulse flow was released in March 2014 and scheduled to mimic natural floods at the time of native cottonwood seed release. The pulse flow entered the Gulf of California in May 2014 (Flessa, 2014). Secondly (and most importantly for this thesis), a base-flow of 52,696 acre-feet (64GL) purchased through the market by a coalition of NGOs has been acquired and delivered to priority restoration areas along the Mexican reaches of the river. The NGO coalition was named the Colorado River Delta Water Trust (The Trust) and became the first water trust actively acquiring and leasing water for the environment in Mexico.

To acquire the water for the base-flow, the Trust makes use of the active water market in the Mexicali Valley, through which it buys water to be delivered to the environment. Funding for the purchase of water rights to provide the base-flow has been largely philanthropic in nature, with limited government funding committed. A notable example of philanthropic capital drives is the *'Raise the River'* campaign, implemented to raise funds for instream water purchases and re-vegetation in the Mexican reaches of the Colorado River (Raise the River, 2016). Progress towards achieving the target volume has been facilitated by the Trust both purchasing and leasing water rights. In the remaining years of the *Minute*

319 pilot period (ending 2017), the Trust will continue to purchase and lease water from irrigators in the Mexicali Valley (Zomora, 2015, pers. comm.).

Figure 3.10 The Colorado River Basin

Source: Colorado River District (2016), available: http://www.coloradoriverdistrict.org/map-gallery/, [Accessed 27th June, 2017]

3.9.2   South-eastern Australia

The following section looks in detail at the market-based *RtB* acquisition program implemented to acquire water rights for the environment in the MDB. MDB physical and policy background, including the reform process which has led to the use of market-based methods as a means to acquire water, is presented in Chapter 2. In the following section only surface water acquisitions are considered.

3.9.2.1 Restoring the Balance Program

The *RtB* is the federal government's primary market-based method for acquiring water for the environment. Initially, the market-based reallocation was provided a fixed budget of $3.1 billion over 10 years (Qureshi et al., 2010), however budget allocation and expenditure has been altered since 2007-

2008 (MDBA, 2016b). Water entitlements were[15] purchased with this funding through a regional tendering system to acquire water entitlements directly from irrigators (Wheeler et al., 2012). Rounds of tenders are announced to the public providing details on target catchments, the budget for water entitlement acquisitions rules for buying and selling, and the final submission date. Irrigators interested in selling water entitlements submitted offers to sell, which were then evaluated by the government against a set of financial and environmental criteria. The *RtB* program exclusively acquired water entitlements to provide permanent reallocation to the environment (as opposed to, for example, allocation or options, which provides annual or conditional reallocation). Figure 3.11 shows *RtB* tenders names, starting years, location and budget committed.

The *RtB* water tenders began in 2007-2008, before the *Basin Plan* was finalised in 2012, on the basis that acquiring water for the environment would be a no regrets action regardless of outcomes. As more information became available through the *Draft Basin Plan* (2010) and *Basin Plan* (2012), the tender process changed over time to reflect environmental watering targets. In the period between *RtB* inception in 2007-08 and the passage of the *Basin Plan* in 2012, the government received 7,591 applications to sell water and accepted 4,189, the majority of which were sourced from the southern Basin (Wheeler and Cheesman, 2013). As is evident in Figure 3.11 the frequency and budget of the tenders peaked in 2010-2011, by which time the federal government had used approximately $1.25 billion to purchase water entitlements under the *RtB* (Wheeler et al., 2012). AusTender data lists six addition tenders from 2013-2015, indicating a slow in the rate of acquisition through the tender process.

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[15] Note that although the *RtB* program is technically still active, recent funding has been directed to infrastructure-based reallocation programs following a bill introduced to parliament to limit the volume of water acquired through entitlements to 1,500GL. See Chapter 2, Section 2.7.4 for further details. As a result, this section is written in past tense.

Figure 3.11 *Restoring the Balance* **Tenders**
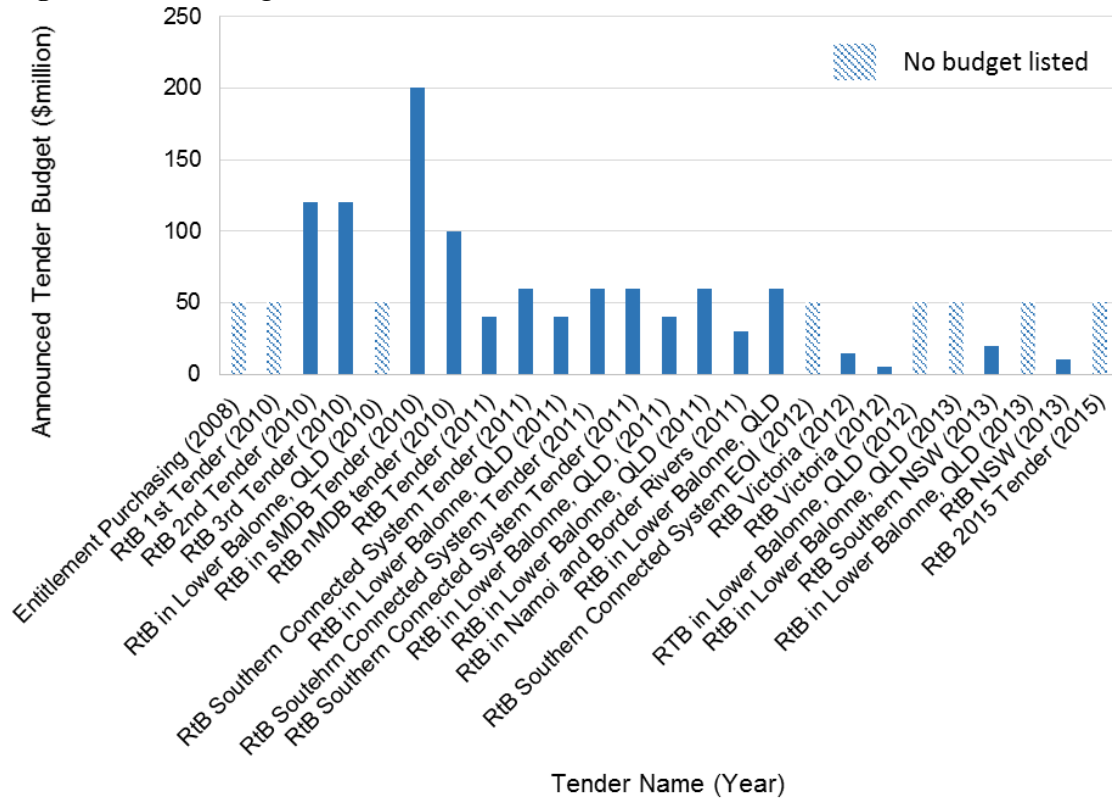
As part of the *RtB* tender process, water was acquired by the government at around market price, although Wheeler and Cheesman (2013) found that participants who sold water to the *RtB* did so believing they would receive a higher price by selling to the government over other irrigators, or because there were no buyers for their water within their region. Because irrigators own the largest volume of water in Australia, they are the primary sellers of water to the environment (Wheeler et al., 2012).

Wheeler et al. (2012) find that there are two primary motivations of water sales from an irrigator's perspective: a) water sales as a last resort to personal and financial circumstances such as debt, divorce, and death; and b) water sales as a strategic response to water prices, farm investment and water surpluses. Some irrigators also participate in the *RtB* program as a chance to restructure the farm business or to exit farming into retirement (Wheeler and Cheeseman, 2013). Wheeler et al. (2012) show that the human and social capital (e.g. personal values and environmental attitudes) and financial characteristics (e.g. farm debt) play a large role in determining irrigator willingness to consider selling water for the environment. Contributing factors towards willingness to sell include being more commercially orientated, more environmentally orientated, being male, having worked in farming for fewer years, and having low water allocations on average (Wheeler et al., 2012). Conversely, holding more traditional farming values and handing down the farm to the next generation negatively affected irrigator willingness to sell water (Wheeler et al., 2012). Figure 3.12 shows the main reasons for

irrigators selling or offering to sell water between 2008 and 2011. However, these findings were determined using data from 2008-2009 and 2010-2011 when the method for water recovery was entirely entitlement purchases. As the water market becomes more diversified (e.g. trade in environmental allocations), there may be an increase in irrigators strategically engaging in the water market as an additional farm and environmental management tool. This is discussed at length in Chapter 8.

Figure 3.12 Main Reasons for Irrigators' Sale or Offer to Sell Water
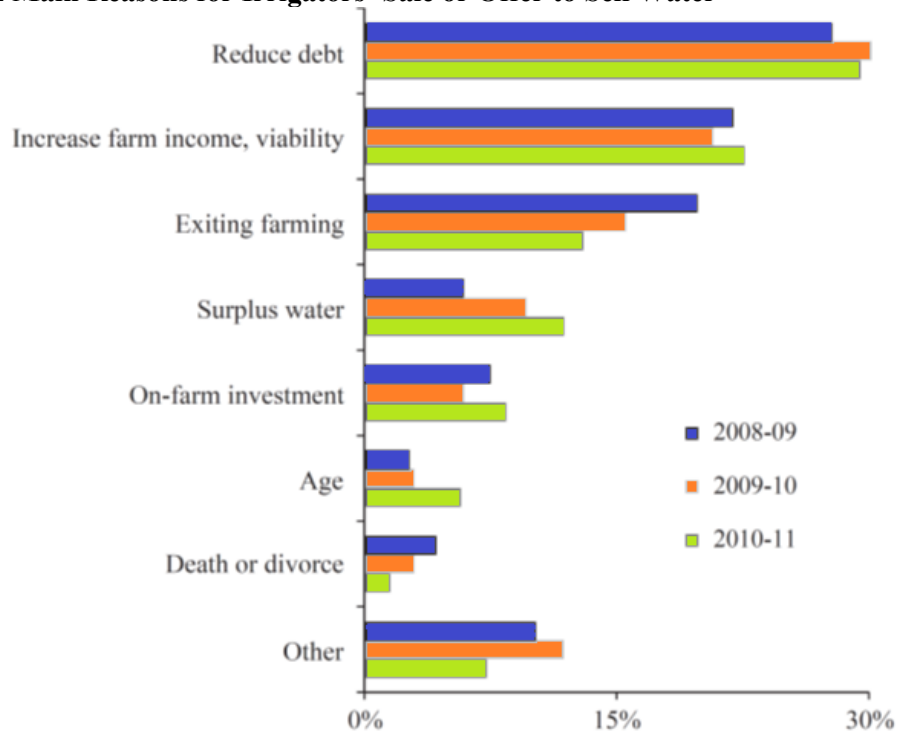
Source: Wheeler and Cheeseman (2013), p. 345.

3.10   Challenges to Market-Based Entitlement Recovery

As described above, the *RtB* is a market-based approach relying primarily on entitlement purchases. Although this program has been internationally recognised as a significant step towards water resource sustainability (e.g. Garrick et al., 2011), there remain a number of challenges to this approach, which have not yet been fully rectified. These challenges are summarised in Table 3.9 and detailed further below. The remaining issues and hurdles to water recovery demonstrate a continued need for research and policy analysis to inform the judicious expansion and diversification of environmental water markets in the MDB.

Table 3.9 Summary of Issues Identified with the Current Entitlement Recovery Strategy

Issue	Description
Negative community perception	Negative community and local media perception of entitlement recovery as a "claw back" of rights; perceived injustices in farming communities.
Irrigator engagement and preferences	Hesitation to sell water to the federal government; irrigator preferences for irrigation infrastructure expenditure and additional short-term water products over permanent entitlement acquisitions.
Third party impacts	Reduced community income due to possible reductions in farming jobs; stranded irrigation assets and reduced irrigation profitability for reaming irrigators.
Private transaction costs	High transaction costs levied by irrigation districts on the sale of entitlements compared to allocation sales.
Surface water-groundwater substitution	Irrigators increasing their groundwater use to supplement water sold to the government, possibly leading to quicker rates of groundwater depletion.
Entitlement supply profiles	Entitlement supply profiles designed for steady and reliable irrigation demand may not be suited to highly variable environmental watering demands.
Volume as success metric	Volume of water entitlements acquired is not a comprehensive indicator of the subsequent improvements in environmental condition.
Climate change risks	Reduction in volume of water availability due to climate change may erode the water reallocated to environment.

Source: Own Table.

3.10.1 Negative Community Perception

The success of water entitlement purchase programs requires stakeholder acceptance and participation (Lane-Miller et al., 2013), both from irrigators who own the rights and the EWHs who acquire and manage them. The entitlement purchases sparked considerable community anger and fear in rural MDB communities. In particular, the release of the *Draft Basin Plan* (2010) after years of prolonged drought resulted in the *Draft Basin Plan* being extremely poorly received by communities and rural media outlets (Gross, 2011). The negative community perception of the subsequent *Basin Plan* was heightened by fear that the water was to administratively reallocated rather than being purchased at market rate from willing sellers. Even when understanding that water was to be purchased at market rate, agricultural lobby groups and rural community representatives viewed the design and implementation of entitlement recovery as preying on *desperate sellers* rather than *willing sellers* (e.g. South Australian Murray Irrigators, 2015; Murrumbidgee Valley Food and Fibre Association, 2015). Overall, the *RtB* and *Basin Plan* were subsequently vilified in media outlets and peak irrigation bodies as the "claw back" [16] of water rather than a "buy-back". Wheeler and Cheesman (2013) found that

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[16] The conceptualization of the *Basin Plan* objectives as a "claw back" of water rights remains evident in current media even though the drought has broken. For example, in 2016 an article was published which refers to the *Basin Plan* as the "Federal Government's water clawback plan" (Kotsios, 2016, available: http://www.weeklytimesnow.com.au/news/water/murray-darling-basin-job-shock-as-water-buyback-hits-farms/news-story/1e879f02487eb52d2475e46e3aa2ce3f.) and in 2017 a similar rural paper detailed the "diabolical impacts of water clawback" (Jones, 2017, available:

reducing farm debt is a primary determinant of why farmers sell entitlements, indicating that there may be truth to these concerns.

Negative reactions from agricultural and rural communities can be seen through the lens of the perceived injustice of the reform process and a lack of trust between irrigators and the federal government. Gross (2011) argues that perceived injustice resulting from disrespectful treatment of rural community interests, inadequate inclusion of communities in decision-making, and the perception of an unjust outcome has impacted community acceptance and engagement with entitlement recovery. Likewise, Wheeler et al. (2017) shows that irrigators feel that they are already contributing their fair share of the burden of water reform, but do not trust the federal water agencies to manage water to solve community and environmental flow problems. As an increase in environmental flows and better environmental management is core aim of the *Basin Plan,* this sentiment revels a very fundamental trust issue in the reform process. This is shown in Figure 3.13.

Figure 3.13 MDB Irrigators' Best-Worst Scores in Response to Statements Regarding Water Reform
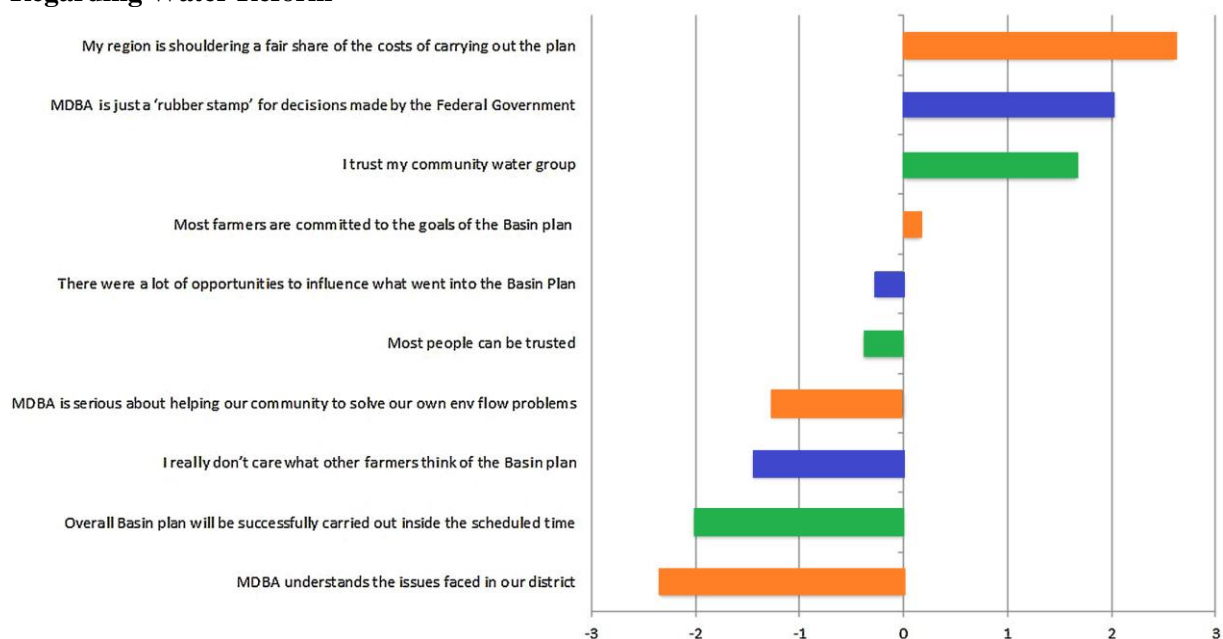
Source: **Wheeler et al. (2017), p. 251.**

In a similar vein, Evans and Pratchett (2013) argue that failure to account for irrigator preferences and local values has contributed to the slow[17] progress of water reform and a government failure "*to win the hearts and minds of the rural community*" (Evans and Pratchett, 2013, p. 541). Likewise, Tan (2012) argues that local values and irrigator preferences have not been adequately accounted for in the design of the *RtB* program, thereby compromising the amount of irrigator engagement in the program. The

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http://www.goondiwindiargus.com.au/story/4503389/the-diabolical-impact-of-water-clawback/). In both cases, the media articles refer to the irreversible and "devastating impact on the morale of those communities"

[17] The classification of the reform process as 'slow' is the opinion of Evans and Pratchett (2013). Other analysis has suggested that aspects of reform including the *RtB*, have occurred at faster than optimal pace (PC, 2010).

contentious nature of the *RtB* and divided community acceptance is highlighted by Wheeler and Cheesman (2013), who identify 50%, 50% and 22% of farmers in NSW, Victoria and SA, respectively, disagree with the overall objectives of the *RtB* program. More widely, a survey of regional wellbeing identifies that MDB residents think that the *Basin Plan* will have a negative or neutral impact on the economy and communities in the MDB, as shown in Figure 3.14. Interestingly, Figure 3.14 also demonstrates that only the minority of surveyed MDB residents expect that the *Basin Plan* will have a positive impact on the health of the MDB environment or their local area. This suggests overall that MDB irrigators do not expect the *Basin Plan* to achieve its most fundamental stated objective.

Figure 3.14 MDB Residents Perception of the Likely Effects of the *Basin Plan* on Economic Outcomes, Community Outcomes and Environmental Outcomes
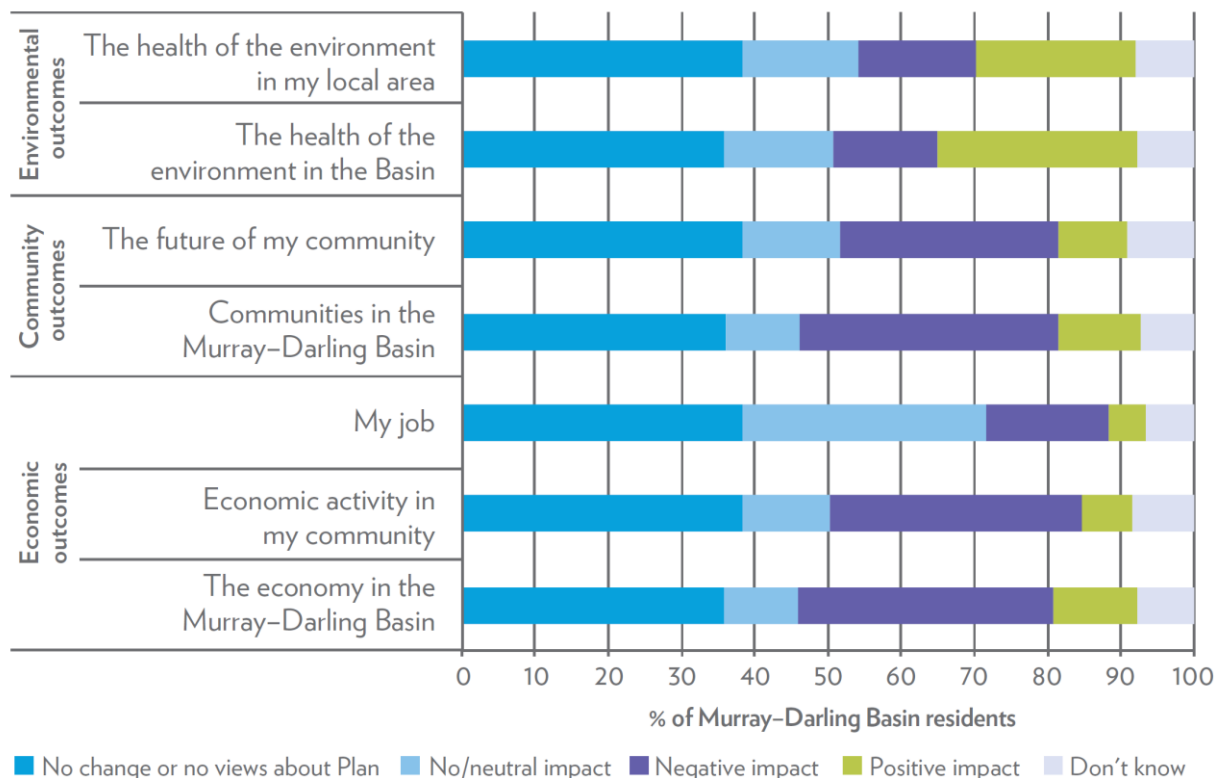
<u>**Source:**</u> **Schirmer and Berry (2014), p. 11.**

On the other hand, irrigator hostility to the water entitlement purchases has been identified as possible rent-seeking behaviour (Loch et al., 2014b), which can occur when large investments benefit a specific group of people (Tullock, 1989).

3.10.2 Irrigator Preferences for Alternative Recovery Mechanisms

Loch et al. (2014) show that irrigators in 2011-2012 in the sMDB have a moderate preference infrastructure-based reallocation programs (sum of 57%) over market-based reallocation (sum of 43%). This is shown in Table 3.10.

Table 3.10 Irrigator Preferences for Budget Expenditure on Water Recovery

Policy options	Water recovery mechanism	Mean percentage of irrigators agreeing (%)			
		NSW (n=176)	SA (n=205)	Vic (n=154)	Weighted sMDB average
Infrastructure based reallocation	Upgrading on-farm infrastructure	32	21	34	31
	Upgrading off-farm infrastructure	28	23	25	26
	Sum of infrastructure-based reallocation	**60**	**44**	**59**	**57**
Market-based reallocation	Water entitlement purchases	18	34	19	21
	Water allocations/entitlement leases/options contracts	12	6	11	10
	Exit packages and revegetation payments	6	11	7	7
	Standard exit packages	5	5	5	5
	Sum of market-based reallocation	**41**	**56**	**42**	**43**

Source: Adapted from Loch et al. (2014), p. 400.

In addition to a moderate preference for infrastructure-based reallocation, Wheeler et al. (2013a) show that just under one third of irrigators would consider selling water to the environment, if short-term (e.g. allocation sale or entitlement leasing) or contractual (e.g. options) water products were able to sold to the environment instead of just water entitlements. In particular, it was found that 21% of irrigators, who are not at all interested in selling entitlements to the environment, would consider selling allocations to the environment (Wheeler et al., 2013a). Entitlement leasing was marginally less preferred by irrigators in 2010-2011 than allocation sales, confirming a potential bias away from the transfer of entitlements to the environment (even on a short-term basis). Table 3.11 demonstrates irrigator willingness to participate in alternative *RtB* programs in the sMDB in 2010-2011. Both Wheeler et al. (2013a) and Loch et al. (2014) find that irrigator preferences for alterative water products and subsequent distribution of water recovery expenditure have not been represented in the current policy for acquiring water for the environment.[18]

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[18] Since these observations have been made there has been a change in policy emphasis towards water recovery through infrastructure upgrades and CEWH annual environmental water trading, which is discussed in Section 3.11.

Table 3.11 Irrigator Willingness to Participate in Alternative Water Sales to the Environment

sMDB state	Would participate in allocation trade	Would participate in entailment leasing	Would participate in options contracts
NSW	29%	26%	33%
VIC	27%	25%	25%
SA	41%	37%	41%

<u>**Source:**</u> **Wheeler et al. (2013), p. 433.**

3.10.3 Third Party Impacts and Possible Social Impacts of Water Trading

Negative community perception and reluctance to sell is closely tied with concerns of third party impacts of water entitlement sales. Some of these concerns, also valid for inter-regional entitlement trade, include: reduction in community income due to reduced farming demand for inputs and labour (Adamson, 2015); irrigators and families leaving the region after selling water (Lewis, 2008; Grand, 2017); and degradation of de-watered land which can result in invasion of pests and weed species to neighbouring properties (House of Representatives Standing Committee on Regional Australia, 2011). An additional prominent issue is the potential for stranded irrigation assets (the 'Swiss cheese' effect) in irrigation districts (Heaney et al., 2006; PC, 2010). Stranded assets have particularly been a concern in regions where irrigation fees are levied primarily on water entitlements rather than delivery entitlements (NWC, 2010). It is argued that stranded assets occur when irrigators sell their irrigation rights and leave the irrigation district, causing the fixed irrigation costs to be redistributed among the remaining irrigators (ACCC, 2009b). As more irrigators leave the district, the irrigation operator may re-evaluate the cost of water supply provision, resulting in higher fixed and variable irrigation fees for the remaining members (Heaney et al., 2006), thus creating a potential reduction in irrigation farming profitability (NWC, 2010).

In the past, concern over stranded assets has been an argument for limiting entitlement buy-backs or inter-regional trade (NWC, 2010). However, the unbundling of water rights (*NWI*, 2004) into delivery rights and water entitlements has in part mitigated the issue and has two important implications for managing stranded assets. First, it has allowed irrigators to sell water entitlements but retain delivery rights and purchase allocations on the water market to meet their irrigation needs. Second, it has allowed irrigation districts to levy ongoing irrigation fees against delivery rights (instead of water entitlements) in order to generate revenue from water users to maintain irrigation infrastructure (NWC, 2010). As a result, stranded assets are no longer a serious economic concern for entitlement recovery or inter-regional entitlement trade. However, the role of stranded assets in community concern is an important feature in understanding how irrigators engage with entitlement recovery for the environment.

In addition to the third party impacts on other irrigators due to water trading, there are a few papers that argue water trade can result in negative social ramifications (e.g. Kiem, 2013). Kiem (2013) argues that considering town, city and industrial supplies as 'high value use' and agriculture as 'low value use' has

the undesirable effect of enabling market forces to move water away from agricultural use. Further Kiem and Austin (2012) and Kiem (2013) argue that those with the lowest adaptive capacity within the agricultural community are likely to be most impacted by negative social impacts of water trade. However, contrary to Kiem (2013) there is a wide economic literature suggesting that the social impacts of trade are not necessarily negative in effect, including a number of National Water Commission Reports (e.g. NWC, 2010; NWC, 2012). Contrary to the arguments by Kiem (2013), the NWC (2012) report demonstrates that water trade increased the regional domestic product in the MDB throughout the drought years by the order of $1.5 billion dollars. This is achieved by providing farmers with an additional farm management tool and ensuring water could be transferred through the market from lower value annual crops such as rice, to higher value perennial crops which require a minimum value of water to be delivered, even in times of drought. This is particularly shown by Kirby et al. (2014) who provide evidence that although water availability decreased by up to 70% in the MDB during the Millennium Drought, the adjusted gross value of irrigated production declined by only 10% within this period owing to farmer adaptation, including the use of water markets (Kirby et al., 2014). Further quantitative evidence regarding the beneficial farm and basin impacts of water trade during drought are discussed in the literature review in Chapter 5. Another key paper providing evidence to disprove the arguments by Kiem (2013) is Grafton and Wheeler (2016). Grafton and Wheeler (2016) provide evidence that the majority of people trading water in the market are irrigators (not, for example, a mix of city buyers and agricultural sellers) and that there has been only very small volumes of water traded to mining, industry or electrical generation. This is evident in the fact that mining, manufacturing and electricity generation has not substantially changed its water use since 2008-09 (Grafton and Wheeler, 2016). On a farm scale, Wheeler and Cheeseman (2013) demonstrated that 80% of irrigators who sold all or part of their water entitlement did not change their farming set-up, crop-mix or the number of employees employed on the farm. In 94% of the cases where irrigators sold only part of their water right, the farmers retained the water delivery rights (Wheeler and Cheesman, 2013) indicating an intention to buy allocations/entitlements in the future to continue farming, or to ensure the farm retains its property value. Overall, based on a large scale survey of 520 water entitlement sellers in the MDB, Wheeler and Cheesman (2013) have found that, contrary to the arguments by Kiem (2013), irrigators and Basin communities are better off with water trading than without it.

3.10.4 Private Transaction Costs[19]

Transaction costs (TCs) in environmental water recovery refer to the resources that are required to define, establish, maintain and transfer property rights (McCann et al., 2005; Garrick et al., 2013). The TCs of trading complex resources are inherently difficult to define and manage, as they are affected by unique physical, technical, cultural and institutional factors inherent in any water resource system

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[19] Data and results from this section were obtained during research work undertaken for the preparation of: Loch, A., Wheeler, S. & Settre, C. (*forthcoming*) Private transaction costs of water trade in the Murray–Darling Basin.

(McCann et al., 2005; Garrick et al., 2013). The nature of TCs and the mechanisms put in place to minimise them will in part determine the long-run feasibility of environmental water markets (Garrick and Aylward, 2012). Private TCs of water trade include the resources required to: i) investing time in monitoring, identifying buyers/sellers and carrying out the trade; ii) negotiating terms and conditions of the trade; iii) monitoring and mitigating third party impacts; iv) conveyance fees to conduct the trade; v) possible dispute resolution; and vi) managing barriers to trade (Loch et al., *forthcoming*). When irrigators sell entitlements to the government through a tender process, such as in the *RtB,* a number of these transaction costs are removed, such as the time invested in finding a suitable buyer for the entitlement. However, there remain a number of private TCs incurred through fees levied by irrigation districts on entitlements being sold out of the district (Loch et al., *forthcoming)*.

One important aspect of private TCs are the termination fees incurred by irrigators selling entitlements. Termination fees are levied by irrigation districts to discourage irrigators selling water and leaving the district, and to protect the district from the third party impacts of stranded assets, as discussed above. Many irrigation districts levy compulsory termination fees at a multiple of the annual infrastructure access fee, typically covering the operator for 25-40 years of future "missed" infrastructure access fees (ACCC, 2009a). High termination fees increase the transaction cost of selling water to the government, resulting in barriers to permanent entitlement trades (Loch et al., *forthcoming*) and market distortion by reducing the volume of permanent water traded (ACCC, 2009a). In response to concerns that transaction costs levied by irrigation districts were uncompetitive, an investigation was undertaken by the Australian Competition and Consumer Commission and the *Water Charge (Termination Fees) Rules* were subsequently implemented in 2009, limiting termination fees to 10 times the total network access charge payable by the irrigator for the year in which termination occurs (ACCC, 2009b). Figure 3.15 shows the termination fees adjusted for CPI set by Goulburn-Murray Irrigation District (GMID) since 2008. Charges from GMID are used as they provide the longest time-series of data publically available. A significant drop in termination fees is evident after 2009 when the ACCC *Water Charge Rules* were put in place. The black line traces the annual termination fees that would be expected for a 3% increase in fees per year.

Figure 3.15 Indicative Termination Fees for Goulburn-Murray Irrigation District (CPI Adjusted)
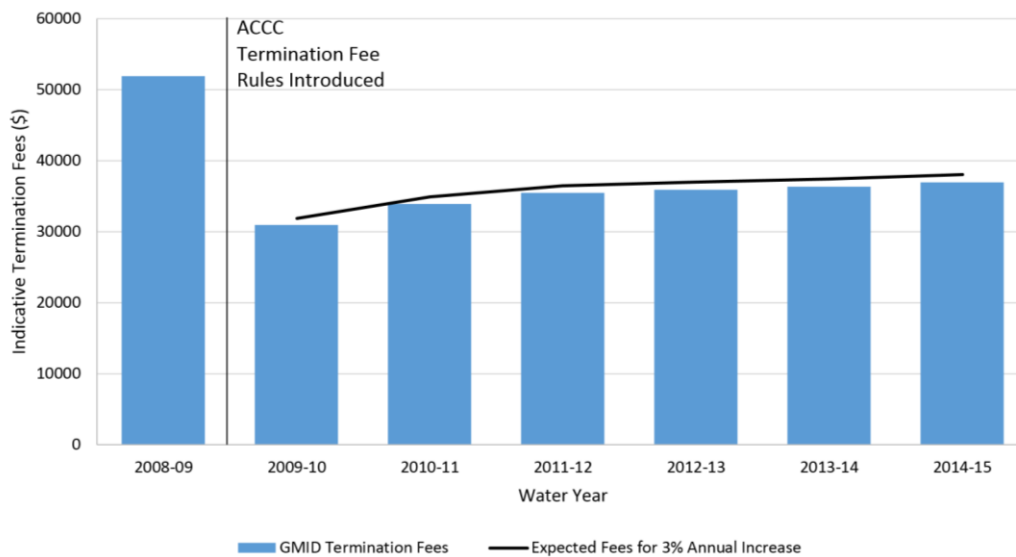
In addition to termination fees, Loch et al. (*forthcoming*) show that water trade TCs, levied by irrigation districts on water entitlement trade, are higher (and moderately increasing) than allocation trade TCs, which are lower (and decreasing). Key values illustrating this trend in water allocation and entitlement trade TCs are shown in Table 3.12 for Western Murray Irrigation (which provides the longest time-series to compare allocation and entitlement TCs). The higher TCs for entitlement trade, evident in Table 3.14 is consistent with international literature which shows that permanent sales for environmental purposes can attract prohibitive transaction fees relative to short-term arrangements, due to approval processes at the irrigation, county, and state levels (Garrick et al., 2011).Table 3.12 Irrigation District Allocation Transfer Costs between 2009/10 – 2016/17.

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[20] GMW reports indicative termination fees in their price schedule; actual fees chargeable to irrigators are calculated as a multiple of the fixed fees payable in the year of termination (GMW, 2009).

Table 3.13 Irrigation District Allocation Transfer Costs between 2009/10 – 2016/17

Irrigation Scheme	Allocation transfer costs	Unit	Mean	Std. Dev.	Percentage change in costs over time period
Western Murray Irrigation (2012/13-2016/17)	External Temporary Water Transfer fee	$/transfer	70.74	3.04	-7.56
	Internal Temporary Delivery Entitlement Transfer	$/transfer	25.25	2.16	-14.96
	Internal Temporary Water Transfer fee (Customer to Customer)	$/transfer	25.25	2.16	-14.96
	Internal Temporary Water Transfer fee (WMI to Customer)	$/transfer	56.76	1.34	-3.87
	Internal Internet Water Transfer fee	$/transfer	24.54	0.85	-3.36

Source: Western Murray Irrigation Pricing Schedules 2009-2017

Table 3.14 Irrigation District Entitlement Transfer Costs between 2009/10 – 2016/17

Irrigation Scheme	Entitlement transfer costs	Unit	Mean	Std. Dev.	Percentage change in costs over time period
Western Murray Irrigation (2012/13-2016/17)	External Water Entitlement Transfer (into WMI) fee	$/transfer	321.10	13.79	10.78
	Internal Water/Delivery Entitlement Transfer	$/transfer	262.86	54.13	65.84
	Share Movement fee (if there will be any movement of Shares with the transfer)	$/movement	145.37	4.34	2.79
	On-Farm Assessment for entitlement trade fee	$/assessment	83.99	68.59	-100.00
	Transformation admin. fee (water entitlements)	$/entitlement	321.10	13.79	10.78
	Change of Ownership fee (sale of water and land)	$/change	211.04	8.11	11.22

Source: Western Murray Irrigation Pricing Schedules 2009-2017.

The data shown in Table 3.12 and Table 3.14, demonstrates that permanent entitlement sales attract higher private transaction costs than short-term sale arrangements. This demonstrates that irrigators face a number of financial impediments to environmental water sales, if entitlement sales remain the only market-based water recovery mechanism, which play a part in reduced irrigator engagement in the buy-back program.

3.10.5 Groundwater-Surface Water Substitution

When entitlements are permanently sold there is evidence of substitution effects between surface water and viable groundwater resources (Wheeler and Cheeseman, 2013; Wheeler et al., 2016; Haenasch et al., 2016; Haensch, 2017). Haensch (2017) shows that this effect is particularly evident when the quality of surface-water is low, leading to surface water entitlement sales and increases in groundwater usage. The relationship between water entitlement sales and groundwater use is an important one, as it suggests

that permanent entitlement sales can have overall negative impacts on water availability and quality for irrigation, as well as the environment, due to increased groundwater pumping (Haensch et al., 2016).

Groundwater resources are also covered by the *Basin Plan* and groundwater SDLs will be implemented in 2019 with a cap on groundwater extractions at 3,334GL/year. Groundwater use should therefore not grow so substantially that it would exceed the groundwater SDLs. However, there are a number unmetered stock and domestic production groundwater pumps within the MDB (MDBA, 2009; CSIRO, 2011), suggesting that increasing groundwater use following the sale of surface water entitlements may increase without proper checks and balances to restrict use. Further, Adamson (2015) shows that the final version of the *Basin Plan* (2012) incorporates groundwater SDLs which have increased from the previously proposed limit (up to 929GL), therefore sending "mixed signals" (Adamson 2015) about the role of the *Basin Plan* in decreasing overall water use.

3.10.6 Entitlement Supply Profiles

When entitlements are bought for the environment, they retain the same characteristics of consumptive use entitlements, and are subject to the same allocation and supply rules. Water resource development in the MDB has progressed for the purpose of supplying reliable water supply for consumptive use (NWC, 2011). As such water entitlements typically have relatively constant inter-annual water allocation supply profiles. However, constant supply reliability is not a determining factor in environmental health; rather, MDB environmental assets require high flow variability with periods of flooding and drying (Overton et al., 2009), as is consistent with wider eco-hydrological behaviours (Junk et al., 1989; Poff et al., 1997). As a result, the supply profiles of consumptive water entitlements purchased for the environment typically do not match the varied demand profile of the environment (Loch et al., 2012; Connor et al., 2013).

In addition, environmental watering requirements themselves are temporally dynamic, such that there may be times in which there is no need for environmental watering in one year, and a higher need in another year (CEWO, 2016). Reallocating water to an environmental asset on a perpetual basis therefore means that environmental water may be surplus in certain years when environmental water demand is low. In these cases, the historic lack of mechanisms to return water back from the environment to irrigation on a temporary basis results in an inefficiency of water use. In addition, entitlements bought for the environment retain the same reliability characteristics as their consumptive use, such that allocation yields from environmental entitlements decrease during periods of drought. This poses a challenge to the CEWH in delivering inundation events during dry periods, and how this should be managed is uncertain (Loch et al., 2014b).

To address these issues, very recently the CEWH Environmental Water Trading Framework (EWTF) (CEWO 2014, revised 2016), which guides allocation purchases and sales for/from the environmental

portfolio, as discussed in Section 3.11.3). However, the CEWH's use of the water allocation market is relatively new and a number of policy questions remained unanswered.

The issues associated with the mismatch between entitlement supply profiles and environmental watering requirements indicate that more flexible arrangements need to be further considered, in order to maximise environmental outcome for the water acquired.

3.10.7 Volume as the Metric of Success

The long-term annual average yield of entitlements acquired has been utilised as the objective of success of the buy-back (Wheeler et al., 2012). This is conveyed in the language of the *Basin Plan* objectives and federal government documentation, which reports success as a percentage of the total 2,750GL acquired. A focus on volume as the sole metric of success implies a linearity between water recovered and environmental outcomes of entitlement recoveries. However, the eco-hydrological relationship between flow volume and ecological response is complex, and focusing on volume as a measure of success does not give adequate attention to the non-linearity between volume and environmental outcome (Crase et al., 2011). Consideration of volume as a sole metric of success will likely lead to an erroneous understanding of the environmental impacts of the buy-back.

Further, a focus only on volume of environmental water needs detracts from the importance of the temporal impact of environmental watering. For example, the health of environmental assets in the MDB is driven by both the volume *and* variability (function of timing) of flow (Overton et al., 2009). By measuring success of the buy-back of environmental water solely on volume, the temporal aspect, and the subsequent environmental benefit it has (or lacks) is overlooked. This issue is closely tied with the constant supply profiles of consumptive entitlements, as discussed above. Therefore, while the volume of entitlements recovered may give an indication of the *policy success* of the entitlement water recovery, it does not give an indication of the *environmental impact* of the water recovery.

3.10.8 Climate Change Risks

Climate change is a serious risk to the expected outcomes of the *Basin Plan* (Pittock et al., 2015). In particular, Kirby et al. (2014a) find that streamflow in the MDB is more sensitive to changes in water availability under the majority of climate scenarios than the impacts of reallocating 2,400-3,200GL to the environment. It is estimated that reallocation will result in improved environmental water availability only under no-climate change or wet climate change projections (Kirby et al., 2014a). Under other climate change scenarios the benefits of increased environmental water availability under the SDLs will be eroded throughout time by climate change impacts (Kirby et al., 2014a), and it is not clear how amendments will be made to current *Basin Plan* arrangements to balance reduced water availability between consumptive users and environmental flows (Pittock et al., 2015). In the sMDB where water availability is expected to be reduced (CSIRO, 2008a; Grafton et al., 2014), this raises considerable issues regarding how water will be managed to achieve set environmental objectives.

State and federal co-ordination required to develop the *Basin Plan* provided an opportunity to confront the impacts of climate change identified in the Basin by the Sustainable Yields Project (CSIRO 2008). However, the opportunity to mainstream climate change risk-analysis into national decision-making as part of the *Basin Plan* was not fully realised (Pittock et al., 2015). Adamson et al. (2015) show that if the *RtB* acquisition strategy does not adequately incorporate climate change risk into decision-making, then the *RtB* program will not likely be able to meet the *Basin Plan* objectives. If climate change is realised through increased droughts and floods as is expected, the CEWO will have to carefully and actively manage the held water portfolio, to maximise net social benefits, and meet *Basin Plan* environmental objectives beyond the acquisition of entitlements as the sole market recovery mechanisms.

3.11 Expansion of the Environmental Water Market

The previous section outlined some of the challenges and issues faced in recovering environmental water through entitlement purchases. One way to overcome these issues is further judicious expansion of environmental water markets and policy development to meet both social and environmental objectives. The following section discusses the possible expansion of the MDB environmental water market to include additional environmental water products.

3.11.1 Water Products for the Environment

Various water products for the environment are distinguished by the physical, temporal and contractual characteristics of the acquired water right. The nature of the acquired water product contributes varying influence on the hydrological regime and flow-dependent environmental outcomes, and therefore influences the management of environmental water over spatial and temporal scales, including the EWH's watering strategy and planning ability. To examine the characteristics of various water rights for the environment it is useful to draw on a case study of the western US environmental water market, where EWHs employ a wider range of environmental water products. Table 3.15 highlights key water transaction tools used in the western US water markets and comparisons in the MDB case.

Table 3.15 Water Rights used for the Environment in western US and Australia

Contract Duration	Western US	Australia
Permanent	Water right purchases Conserved water Groundwater-surface water source switch Change in point of diversion	Entitlement purchases Improving on and off-farm efficiency of infrastructure
Temporary (e.g. one to five year leases)	Leases Split-season leases Groundwater-surface water source switch	Allocation trading Long-term lease agreements
Options	Dry-year trigger agreements Minimum flow agreements Rotational pools	Options contracts

Source: Adapted from Wheeler et al. (2013a), p. 430.

Although there are a number of issues with entitlement recovery (as previously detailed), there are considerable ecological advantages of the permanent acquisition strategy. Namely, permanent acquisition means that water is reallocated to the environment in the truest sense, as it can be used by the environment every year (subject to allocation announcements due to changes in water availability). This provides long-term ecological security and allows EWHs to adopt long-term watering plans. Entitlement acquisitions are particularly useful when working to re-establish or restore an environmental asset that requires ongoing maintenance, such as main channel recovery requiring an increase in base-flows. In particular, high security entitlements (or senior water rights) provide higher reliability during drought compared to other acquisition approaches. By permanently acquiring water, the administrative burden of the transfer occurs only once at the time of sale, however these transaction costs may be higher than other market-based water transfers due to the cost of entitlements relative to shorter-term arrangements.

An alternative option is short-term reallocation, such as annual allocation purchases or entitlement leasing for the environment. In lease arrangements, the water right holder legally retains the ownership of the water right but allows it to be used instream for a specified duration. Annual allocation trade has the benefits of reallocating water when it is most needed by the environment (Kirby et al., 2006; Connor et al., 2013), however this approach is also subject to annual market fluctuations (such as extreme drought), which may reduce the amount of allocations in the consumptive pool able to be purchased on an annual basis. As a result, annual reallocation strategies may not provide a reliable supply during drought, where competition for allocations from permanent cropping irrigators is high and water prices increase in response to scarcity. Considering the opportunity costs of permanent crop irrigators, it therefore may not be socially beneficial for the EWH was to acquire allocations during drought. However, temporary arrangements have lower negative community impacts due to the short-term nature of the transaction.

A number of temporary options involve contractual agreements to change the characteristic of the water right, such as split-season leasing or dry-year agreements. Split-season leases are agreements for consumptive water rights holders to use water for part of the season and lease water to the environment for the other part of the season. An ideal arrangement of this kind meets consumptive water demands during all or part of the irrigation season, and meets environmental water demands during critical ecological times, such as during fish migration or spawning. An additional consideration is dry-year trigger arrangements, which stipulates lease arrangements to take effect when triggered by a contractually agreed upon hydro-climactic threshold. Options contracts such as these can provide reasonable long-term security for the environment and drought reliability, by compensating the user to forgo water usage during years in which additional water is required for the environment. However, the management of options contracts requires ongoing monitoring of hydrological conditions to trigger the

contract, as well as enforcement mechanisms to ensure extraction is forgone, therefore requiring ongoing (but moderate compared to entitlement sales) transaction costs.

Typically it is surface-water rights that are purchased for the environment, but there are cases where it is ecologically beneficial and legally possible to transfer groundwater. Examples of this approach include paying a consumptive user to cease pumping operations or through a groundwater to surface water source switch, such that the previously used surface water is made available for the environment. This approach has limited traction in the MDB but has been used widely in international cases. For example, in the hydrologically linked upper Deschutes Basin, a Groundwater Mitigation Program (GMP) has been implemented where new groundwater use must be offset by a mitigation project (or purchase of mitigation credits from established mitigation banks), resulting in a quantity of water protected instream for environmental use (Water Resources Department, 2008). Payments for ceasing pumping operations may gain greater traction as institutions governing groundwater markets mature and environmental water purchases become more common (Wheeler et al., 2016). The ecological impact and reliability during drought of groundwater transfer tools is highly dependent upon the connectivity between surface and groundwater hydrology, which influences the overall impact on local water availability due to groundwater activities.

In addition to water products listed in Table 3.15 it is also possible for water to be donated to the environment from a consumptive user on a permanent or temporary basis. Incentivising donations to the environment can be achieved by making water tax deductable, akin to a donation to a charity, as achieved in Australia in 2010 (Wheeler et al., 2014a).

3.11.2 Environmental Allocation Trade Opportunities in the MDB

A small volume of literature has looked at the use of alternative water products for the environment in the MDB (e.g. NWC, 2004, Kirby et al., 2006, Cummins and Watson, 2006; PC, 2010; Wheeler et al., 2013a; Connor et al., 2013; Ancev, 2014). Of this literature, environmental allocation trade has been most commonly considered. Kirby et al. (2006), Conner et al. (2013) and Ancev (2014) provide hydro-economic modelling results to suggest there is potential ecological benefit of environmental allocation trade with low costs of adopting this strategy. The modelling approach and comparison of results within these models is discussed further in Chapter 6, which provides a comprehensive literature review of hydro-economic modelling in the MDB. This section highlights results specific to opportunities for annual reallocation in the MDB, summarised in Table 3.16. Overall, there is consensus that the addition of water allocation trade for the environment may improve the efficiency and cost-effectiveness of reallocation programs.

Table 3.16 Methods, Key Findings and Future Research Required of Literature Examining Allocation Trade for the Environment

Study	Method	Key findings
Kirby et al. (2006)	Hydro-economic model	Counter-cyclical trading can improve seasonal environmental availability at no net cost to EWH, provided strategic buy/sell behaviour is implemented by the EWH to buy allocations when most needed by the environment and sell allocations when most needed for irrigation.
Loch et al. (2012)	Qualitative irrigator interviews	Irrigator motivations to sell allocations may facilitate EWH allocation purchasing for the environment.
Connor et al. (2013)	Hydro-economic model	Annual EWH trading can reduce the period between flooding of moderately large magnitudes (>1,700GL in the Murrumbidgee) and reduce ecological degradation under some drought conditions.
Wheeler et al. (2013)	Scenario analysis, irrigator survey	Alternative acquisition methods for environmental water can improve structural adjustment in irrigation communities and increase irrigator willingness to participate in the buy-back.
Ancev (2014)	Hydro-economic model	Ecological benefits and net social gain can be improved with wider active participation if the CEWH trades in seasonal environmental allocations.

Source: Own Table.

A key focus of the empirically based literature is the idea of *counter-cyclical* trading (Kirby et al., 2006; Connor et al., 2013), which involves an actively trading EWH who is able to buy and sell allocations from the EWH water portfolio. The advantages of counter-cyclical trading are best explored through an example: consider a floodplain wetland that requires inundation every three years to maintain optimal ecological health. In this example counter-cyclical trading is beneficial by allowing the EWH to lease allocations from the EWH entitlement to irrigators, two in every three years, and deliver the allocations to the floodplain in the third year, which can be supplemented by additional allocations purchased with funds raised by the previous year's leasing. In essence, when water is not needed for environmental watering, the allocation can be placed onto the market to be purchased by a consumptive user. In this way, the environment generates an income from the sale of water when it is not needed, in order to buy water when at more crucial times (Kirby et al., 2006; Connor et al., 2013).

Environmental allocation trading is a relatively new concept in the MDB and a number of important policy relevant questions remain unanswered. Specifically, questions pertain to the financial and acquisition strategy of maximizing environmental outcomes, and minimizing public expenditure and third-party impacts. These questions are raised and discussed in greater depth in Chapter 7, which provides evidence of the potential for a self-financing EWH to fund water allocation purchases through strategically timed generation and sale of carbon credits created by overbank flooding. The following section details the current methods of environmental allocation trading used in the MDB at present.

3.11.3 CEWH Annual Water Trading Framework

As of 2014, the federal portfolio of water entitlements acquired for the environment is actively managed by the CEWH, in line with their Environmental Water Trading Framework (EWTF) (CEWO 2014, revised 2016). The EWTF furthers the objectives of the CEWH *Water Act* (2007) trading rules and the *Basin Plan* objectives. The EWTF stipulates that, each year, allocations from the held environmental water entitlements owned by the federal government can be managed by the CEWH to (CEWO 2014; 2016):

a) deliver water to meet environmental outcomes;

b) carry-over to the following year to meet future environmental outcomes; or

c) trade for equal or higher environmental value if certain circumstances are met

The conditions placed on CEWH allocation trade are provided by the *Water Act* (2007), which stipulates that the CEWH can only trade or dispose of allocations from the federal portfolio of environmental water if:

a) environmental water is not required in the current water accounting period (e.g. environmental watering requirements fully met) and water cannot be carried over (section 106(2)), or keeping an allocation will result in the reduction of future allocations; or

b) environmental outcomes can be improved by selling water allocations and using proceeds to undertake alternative water purchases or environmental activities (section 106(3)).

The conditions and the resulting options for allocation trade are shown graphically in Figure 3.16.

In addition, CEWO portfolio management decisions are made with due consideration to the principles of environmental water delivery stipulated by the *Basin Plan*. The *Basin Plan* lists seven key principles[21] that are to be considered when delivering environmental water (*Murray-Darling Basin Plan*, 2012, Chapter 8, Part 6, Division 1). Of particular note for the economic considerations of environmental water management are Principles 5 and 6:

> Principle 5: Priorities for applying environmental water are to be determined having regard to: (a) limitations on the effectiveness of environmental water, (b) cost effectiveness [….] and (f) optimising economic, social and environmental outcomes;

> Principle 6: Priorities for applying environmental water are to be determined having regard to: [….] (b) ecological opportunity costs of using water for a particular environmental outcome instead of another environmental outcome.

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[21] i) consistency with ecologically sustainable development; ii) consistency with objectives; iii) flexibility and responsiveness; iv) condition of environmental assets and ecosystem functions; v) likely effectiveness of related matters; vi) risk and related matters; and vii) robust and transparent decisions.

Figure 3.16 CEWH Environmental Water Trading Framework
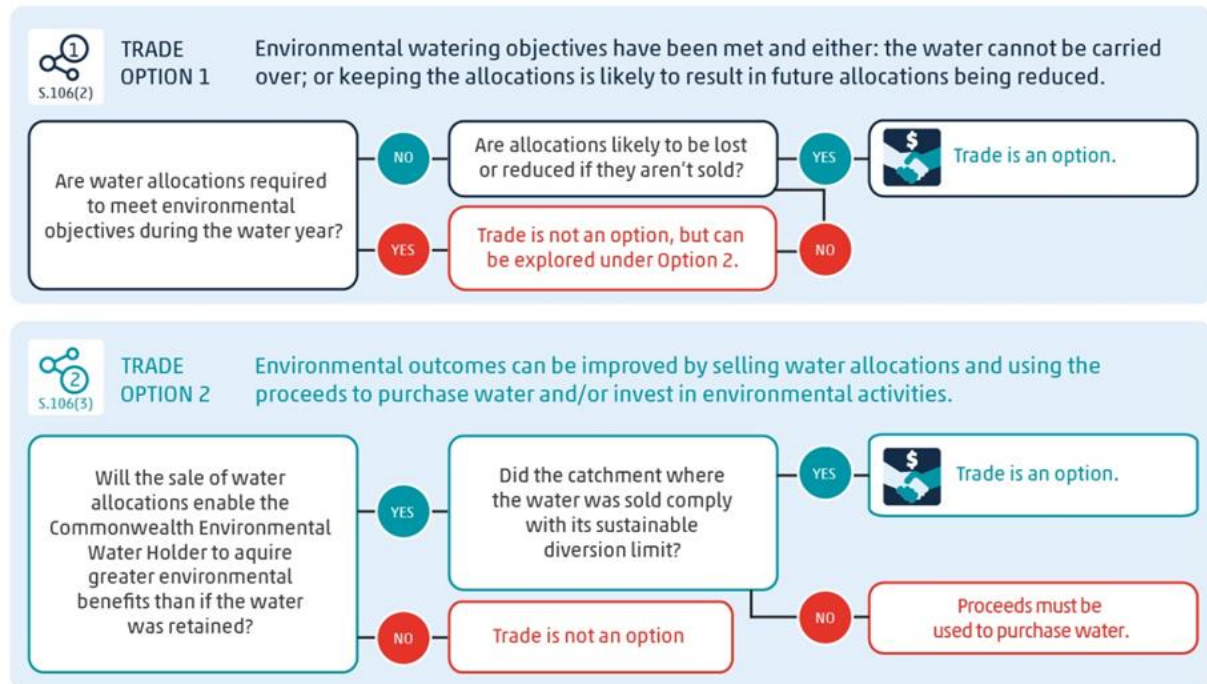
Analysing the conditions for selling water allocations and the principles guiding environmental water delivery, it becomes clear there needs to be a means of estimating the value (and opportunity cost) of benefits derived from environmental water deliveries. For example, trade option 2 (Figure 3.16) requires both the value of benefits derived from allocation delivery in each water year and also the benefit that could be derived from selling the available allocation to be determined. This is a corner-stone of the EWTF, but remains relatively under-analysed in the hydro-economic literature, due to an under-representation of environmental dynamics and economic valuation of the changes potentially induced by an actively trading EWH. This highlights a large gap in the reallocation literature. Further exploration of this gap is provided in the literature review and is addressed in the modelling Chapters 5, 6 and 7.

3.12 Conclusion

Water markets are an institutional innovation that can help allocate scarce water between consumptive water-users by facilitating the trade of water rights. When the legal and institutional conditions allow for it, water markets can also be a useful mechanism to purchase water for the environment. In river basins, where water rights are fully appropriated, water markets are more politically palatable than administrative reallocation, as consumptive users are compensated for forgoing the use of their water right.

Although water markets exist in varying degrees of formality around the world, there are few instances where they have been used to reallocate water to the environment. The most prominent example is the MDB in Australia, where the federally led buy-back of consumptive water rights provides a leading

example of how markets can be used to achieve environmental water objectives. However, there remain a number of challenges pertaining to the methods used to acquire water for the environment in the MDB. As highlighted within this chapter, one alternative method to overcome these issues is the judicious expansion of the environmental water market, to allow flexible annual allocation trade between the environment and the irrigation sector. This possibility is explored further using a hydro-economic modelling simulation approach in Chapters 6 and 7.

Before presenting the hydro-economic literature review, methodology and results in Chapters 5 to 7, the following chapter presents a case study of the Coorong, Lower Lakes and Murray Mouth (CLLMM), which explores the limitations of supply-based management of water scarcity and applies the concepts presented in this chapter to CLLMM case study.

CHAPTER 4: A Century of Intervention in a Ramsar Wetland: the Case of the Coorong, Lower Lakes and Murray Mouth

This chapter presents a paper published in the *Australian Journal of Environmental Management* (2017). Background material covered in previous chapters is removed from the original paper to avoid repetition. The paper discusses the ecological issues faced in the CLLMM region during the drought and the role of infrastructure in the management of the ecological crisis which ensued. This paper demonstrates that the management of the Coorong, Lower Lakes and Murray Mouth has been characterised by a sequence of active and reactive infrastructure interventions, first as active interventions to supply consumptive water demands and more recently as reactive emergency drought responses. It is shown that the use of infrastructure has been both the cause (e.g. barrages) and solution (e.g. environmental works and measures) to environment degradation in the CLLMM. However, infrastructure solutions are not necessarily synonymous with achieving sustainability. The cost of intervention in the CLLMM is calculated and shown to have occurred at significant public expenditure and high opportunity cost. Future possible interventions are identified and categorised, as they are likely to become more relevant when the MDB returns to drought scenarios.

It is argued that greater attention to demand-based management strategies, including time-limited environmental water acquisitions and state-based environmental water holdings, provide an alternative or supplement to future infrastructure reliance. There is also considerable scope for greater provision of cultural flows and engagement with traditional owners to improve ecological condition. The case study also reveals that the path from development to sustainability is not necessarily linear and is impacted by situations requiring emergency management, such as those during the Millennium Drought. Overall, this chapter provides further richness to the background of environmental water management and water markets in the MDB through case study analysis, as well as providing a greater explanation of the severity of environmental problems and challenges of water management faced in the Basin.

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Contribution to the Paper	Paper conception, writing, tables, figures, discussion, conclusions		
Overall percentage (%)	80%		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
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Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

- the candidate's stated contribution to the publication is accurate (as detailed above);
- permission is granted for the candidate in include the publication in the thesis; and
- the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Sarah Wheeler		
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4.1 Introduction

The importance of wetlands for habitat provision and biodiversity maintenance has long been recognised. The economic value of wetlands is also widely acknowledged, and tidal marshes, mangroves and estuaries are often cited as the most valuable components in ecosystem service assessments (Constanza et al., 1997; De Groot et al., 2012; Constanza et al., 2014). Despite this, wetlands around the world are subject to widespread, long-term and ongoing loss or conversion (Davidson, 2014). In Australia, the degradation of wetlands is a result of apparent ecological causes and a number of less understood institutional and societal factors (Finlayson and Rea, 1999).

Of the 65 internationally important Australian wetlands, 16 are within the MDB (ABS, 2008b). In a national effort to achieve a sustainable balance between consumptive use and the environment, ongoing water reforms have been implemented, culminating most recently in the *Basin Plan*. The reform agenda embeds water resource sustainability in national legislation and is, partly, motivated by a movement from developmentalism to sustainability (Crase 2008), including a focus on demand-based policies aimed at improving resource efficiency through economic instruments such as water markets, pricing and trade. In particular, water markets have been increasingly used in the MDB as a means to buy water entitlements from willing sellers and return water to the environment (Settre and Wheeler, 2016). When fully implemented in 2019, the *Basin Plan* is designed to provide more environmental water for river channels, floodplains and wetlands across the Basin.

This paper focuses on the coastal wetland system at the terminus of the MDB; comprising the Coorong, Lake Alexandrina and Lake Albert (the Lower Lakes), and the Murray Mouth (CLLMM). One of the reasons for the importance of the CLLMM is that it is a hydrologic indicator site for the *Basin Plan* and represents the broader environmental flow needs of ecological assets near the end of the system (MDBA, 2012c). An open Murray Mouth is an expected and important environmental objective of the *Basin Plan* (Commonwealth of Australia, 2012).

The importance placed on the CLLMM as a hydrological indicator, its' recognition at a national and international scale, and the evident impact of human intervention, make the CLLMM's past and future management worthy of detailed consideration. It is clear that attention to this area is still needed because, despite progress in returning environmental water to the region, dredging of the Murray Mouth to maintain connectivity with the Southern Ocean recommenced in 2015. The CLLMM also provides an interesting reflection on the role of infrastructure interventions and the transition between supply-based and demand-based strategies throughout changing water management paradigms. Infrastructure interventions (supply-based strategies) are the construction and/or removal of physical infrastructure or environmental works and measures (e.g. weirs, bunds, regulators, channels, pumps) and are one of the four key management levers, which affect the CLLMM's ecological character. Other drivers include

dredging, environmental flows and Upper South East Drainage (USED) scheme flows (Brookes et al., 2009). One way to classify infrastructure interventions are as (Hobbs et al., 2011):

a) *Active interventions* which deliberately change ecosystem properties;
b) *Reactive interventions* which maintain ecosystem states or halt degradation processes; and
c) *Proactive interventions* which are steps taken to limit human impact on ecosystems.

Reactive solutions tend to place emphasis on system components rather than ecosystem processes and feedbacks. This approach to examining water management and infrastructure interventions is applied in this chapter for the case of the CLLMM.

4.2 Site Details Overview

The CCLMM is shown in Figure 4.1 The Coorong is a 110km long shallow inter-dune lagoon (Kingsford et al., 2011). The salinity gradient varies from hyper-saline in the Southern Lagoon to estuarine conditions in the Northern Lagoon (Lester and Fairweather, 2009). Lake Alexandrina connects to the Coorong through a naturally occurring opening, artificially managed by a barrage separating fresh water in Lake Alexandrina from estuarine conditions in the Coorong. Lake Albert is a terminal lake and receives primary inflows from Lake Alexandrina. Combined, the Lakes cover 648km^2 and make up the largest freshwater body in South Australia (SA) (Seaman, 2003). Fresh water from the River Murray flows into the Lower Lakes, and is discharged into the Coorong and Murray Mouth via barrage operations (Overton et al., 2009).

The confluence of freshwater, marine and estuarine ecosystems in the CLLMM have historically provided biologically diverse habitat for migratory birds (Paton et al., 2009) and native fish assemblage and recruitment (Zampatti et al., 2010). In 1985, the CLLMM was listed as a wetland of international importance under the *Ramsar Convention* for its significant populations of waterbirds (Paton et al., 2009). The CLLMM meets 8 of the 9 criteria for Ramsar listing (Phillips and Muller, 2006). The CLLMM also provides significant regulating, provisioning, and amenity services (Banerjee et al., 2013), as well as cultural ecosystem services for the people of the Ngarrindjeri Nation[22].

The primary ecological drivers of the CLLMM are water and salinity levels (Overton et al., 2009; Brookes et al., 2009), which are influenced by environmental flows, dredging of the Murray Mouth, Upper South East Drainage (USED) flows and infrastructure management levers (Brookes, 2009).

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[22] The Ngarrindjeri are the Aboriginal people of the lower Murray Region, Fleurieu Peninsula and Coorong Region.

Figure 4.1 Coorong, Lower Lakes and Murray Mouth Region
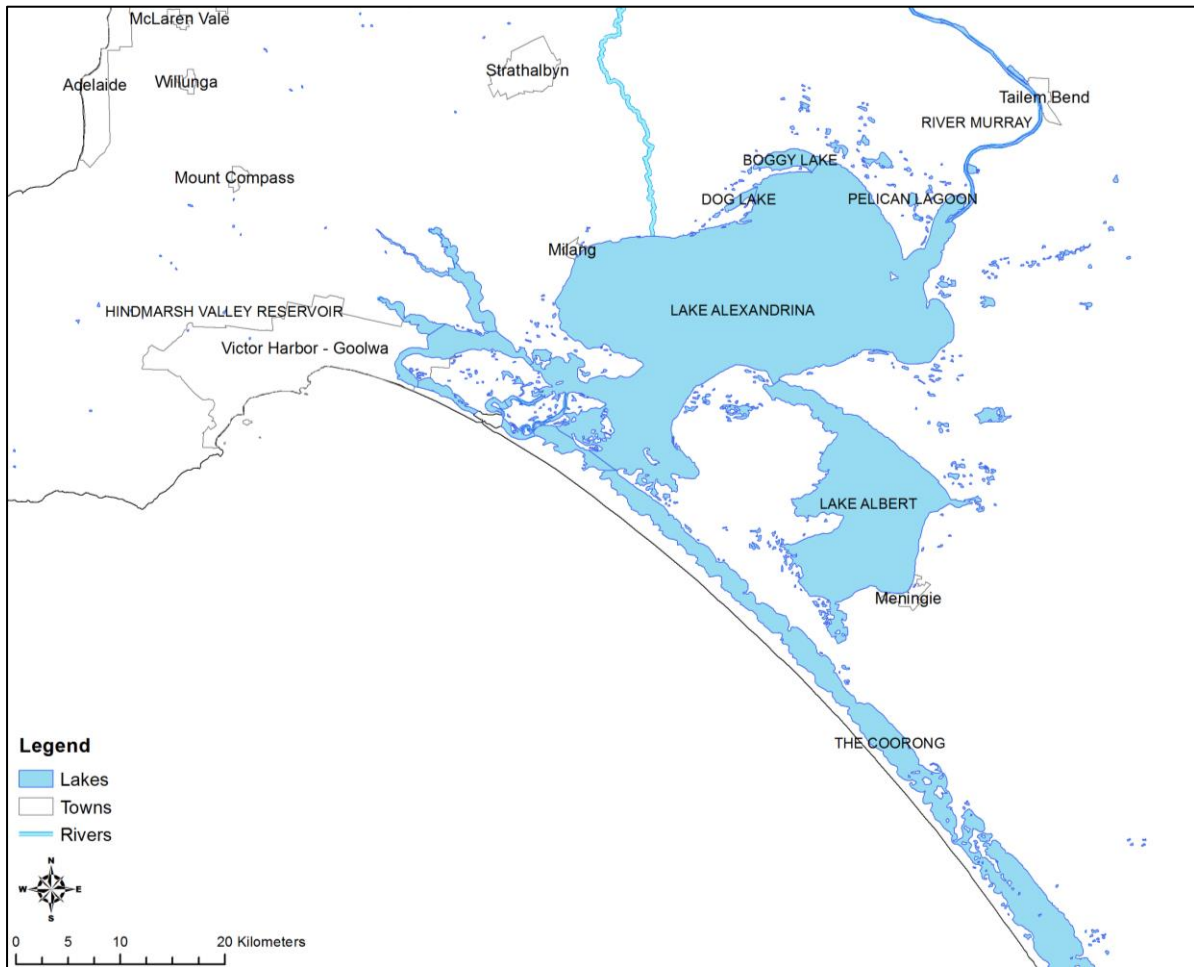
4.3 Water Stakeholders in the CLLMM

CLLMM management is complicated by numerous stakeholder groups interacting across sectorial, geographic and institutional scales, with agendas that are not necessarily compatible. Complexities are compounded by competing motivations of upstream and downstream stakeholders, and the spatially localised perception of benefits. To identify stakeholders we employed an adapted typography of stakeholder groups in wetland functions and values proposed by Turner et al. (2000), as shown for the CLLMM in Table 4.1.

Table 4.1: Stakeholders of CLLMM Wetland Values and Functions

Stakeholder Group		Detail
Water Users		
Water abstractors		Rural townships source water from the Lower Lakes for irrigation, watering stock and potable demand. The capital city, Adelaide, sources 85% of its potable water from the River Murray during years of low rainfall (DWLBC, 2009).Water abstractors' benefit from preservation of freshwater supplies in the Lower Lakes.
Direct extensive users		The people of the Ngarrindjeri Nation who are the traditional owners of the CLLMM use fresh water and saltwater fisheries for subsistence fishing (NRA, 2007). Significant cultural ecosystem services are derived by the Ngarrindjeri and water is regarded as critical to their physiological, material and cultural life (Birckenhead et al., 2011). The Ngarrindjeri benefit from and advocate for CLLMM's ecological health.
Agricultural producers	Irrigation	Irrigated dairy, viticulture, and meat cattle are prominent around the CLLMM. Irrigators benefit from water resource development to provide reliable freshwater supplies for irrigation use.
	Dryland	Land use data indicates a transition from irrigated agriculture to dryland farming in the Coorong District council, partly in response to reduced water availability.
Human settlement close to wetlands		In 2013 local government areas around the CLLMM had a population of 50,989 (ABS, 2013). Human settlements benefit from the maintenance of a reliable supply of potable water in the Lower Lakes and the amenity of a healthy ecosystem.
Non-water Users		
Direst intensive users		Commercial and recreational fishers who intensively harvest fresh and salt water fish. Commercial fisheries benefit from hydrological connectivity in the CLLMM to maintain a salinity gradient required for ecological health of the fishery (Ye et al., 2013).
Indirect users		Individuals and communities across a large spatial scale benefit from carbon sequestration, hydrological stabilization (e.g. flood and tidal surge mitigation) and export of sedimentation through the Murray Mouth. Indirect users benefit from actions taken to increase ecosystem health and resilience.
Nature conservation and amenity groups		Conservation groups and real estate owners benefit from CLLMM ecological health and ecological restoration.
Tourism and recreation		Nature based tourism, recreational boating and fishing substantially contribute to community viability around the Lower Lakes. In 2014 163,000 visitors visited The Coorong local government area (LGA) totalling an annual spend of $29 million (TRA, 2014). Tourism and recreation benefit from improved health, amenity and access of the CLLMM.
Nonusers		Individuals and communities who attribute an intrinsic or existence value to the CLLMM, although they may never directly use it.

Source: Own Table. Categorizations based on Turner et al. (2000).

Stakeholders interact at varying institutional levels including local peak bodies, management groups, district councils and the state and Australian governments. Stakeholder groups are not mutually exclusive and change between or within groups may occur. For example, agricultural producers have undergone a considerable transition from irrigated dairy to dryland cattle farming in the Coorong District Council, owing in part to decreased water levels and water quality issues in the Lower Lakes. Agricultural producers have traditionally benefited from water infrastructure development and have adapted to recent reduced fresh water availability, in part through new pipelines from the River Murray, as described further in Table 4.2. Agricultural producers may also be classed as water abstractors, as could human settlements close to wetlands. The Ngarrindjeri, direct extensive users, have developed the *Ngarrindjeri Sea Country Plan* (2007), which sets out objectives and strategies to implement their vision for the CLLMM (NRA, 2007). The Ngarrindjeri specifically do not support any infrastructure construction and advocate for the River Murray flowing to the Southern Ocean (NRA, 2013). Recently, an agreement between the NRA and the Commonwealth Environmental Water Holder (CEWH) has been made, to collaboratively manage environmental water in the Lower Murray during the 2015-2016 period (CEWH and NRA, 2015). The tourism industry, including nature conservation and amenity groups, has historically contributed to the local economy (DoEH, 2009) and benefits from the continued generation of recreation and amenity ecosystem services resulting from CLLMM ecological health.

4.4 Impact of Water Resource Development on CLLMM

The health of the MDB is contingent on large, variable and specific environmental flows (Overton et al., 2009). Prior to development of water resources, mean flow to the Murray Mouth was estimated to be 12,233GL/year (CSIRO, 2008a); with peaks of 40,000GL/year and lows of almost zero (Newman, 2000). Various approaches to water management, such as the national focus on water resource development and abstraction (see Chapter 2), have significantly altered the natural flow regime in the lower MDB. Figure 4.2 reflects modelled and recorded barrage flows in the CLLMM, overlain with the changing phases of Australian water management and key events (e.g. drought, construction, legislation etc.) Dates in Figure 4.2 indicate year of infrastructure completion and periods of drought affecting south-eastern Australia.

Figure 4.2 Flows over the Barrages and Key Climactic, Policy and Infrastructure Events
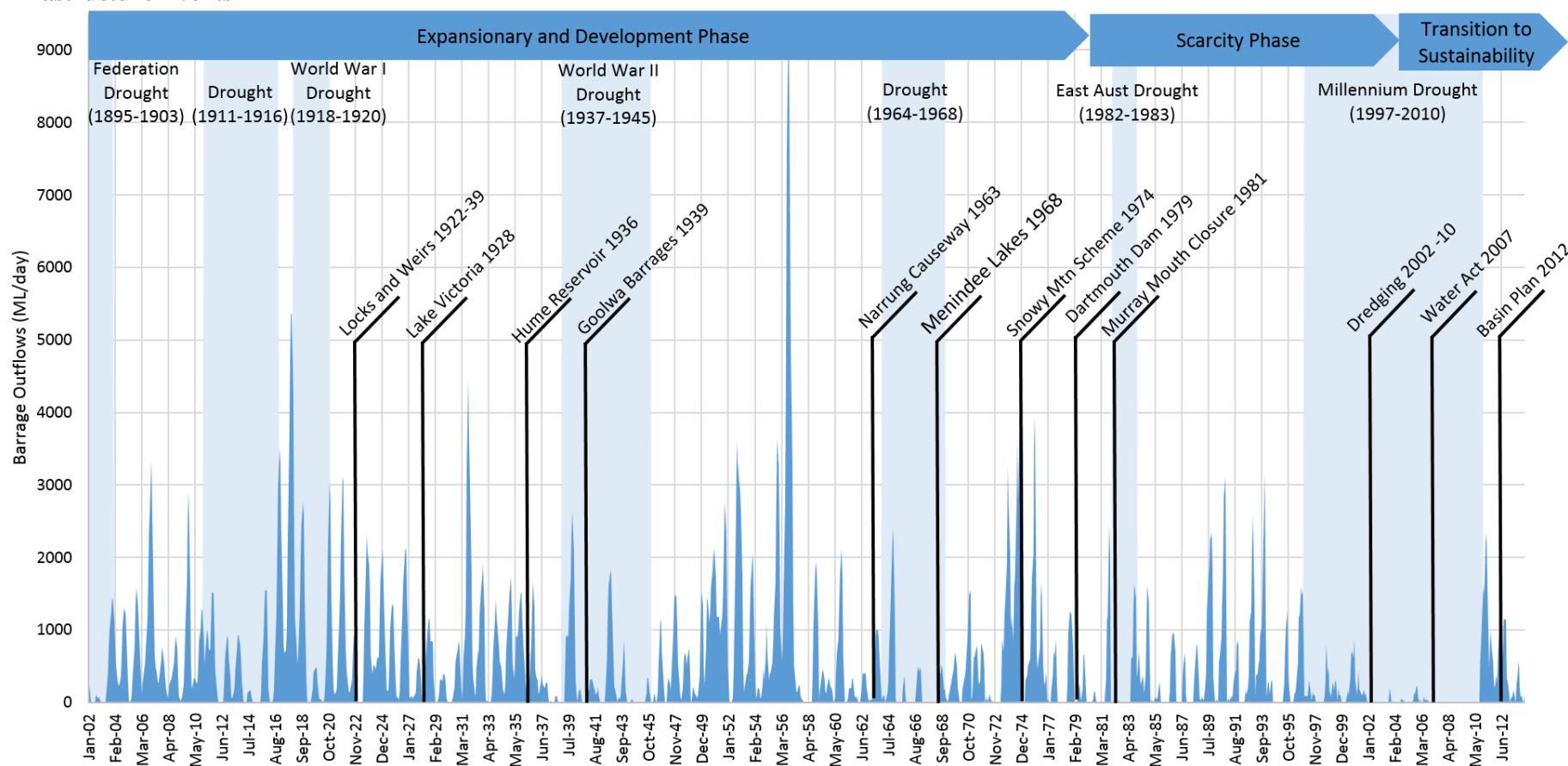
Source: Own Figure. Unpublished Barrage Flow Data provided by SA Department of Environment, Water and Natural Resources; Drought Data obtained from ABS (2012).

Notes: Flows over the barrages from 1970-2014 are estimated using the Murray Simulation Model (MSM) (MDBA, 2010b), flows 1902-1970 are based on a regression of later barrage flows and flows to South Australia ($R^2 > 0.95$).

The first phase of water management, illustrated in Figure 4.2 is the expansionary or development phase characterised by a command-and-control approach to resource management and rapid infrastructure development (NWC, 2011). Infrastructure was constructed to provide benefits for river navigation, water storage, diversion for consumptive use and hydro-electricity. Both the regulatory effects of dams and the impact of upstream diversions reduced water available to MDB wetlands (Kingsford, 2000), while end-of-system flows have been reduced by 61% to 4733GL/year on average, relative to the modelled pre-development case (CSIRO, 2008a). This expansionary period saw the construction of five barrages in the CLLMM to separate freshwater pools in the Lower Lakes from the Coorong's estuarine conditions.

Modelling by CSIRO (2008) suggests that river regulation has increased cease-to-flow occurrences at the Murray Mouth from 1% of the time under modelled natural flow, to 40% of the time with water resource development. This was evident in 1981 when the Murray Mouth closed for the first time due to increased siltation associated with reduced outflows. From the 1980s onwards (the scarcity phase in Figure 4.2), water scarcity initiated a transition from infrastructure development to an increased emphasis on demand-based policies (Tisdell et al., 2002; NWC, 2011). Unsustainable rates of water extraction resulted in the implementation of a cap on diversions to limit abstractions at 1993-94 levels. The implications of the scarcity phase of water management became evident between 1997 and 2010, when Australia experienced a prolonged drought (Millennium Drought), and MDB flows were the lowest on record. In 2001-2002, the Murray Mouth neared closure for a second time following 630 consecutive days of no flows through the barrages (Phillips and Muller, 2006). The reduction in hydrological connectivity between the estuary and the ocean resulted in the salinization of the Coorong Lagoon, and the salinities in the Murray Mouth and Coorong were marine and hypersaline in 2002-2003 (Geddes, 2003). Dredging to re-establish tidal exchange and mitigate hypersaline conditions was re-implemented and continued intermittently until 2010. Lower Lake levels dropped below sea-level in 2009 (Kingsford et al., 2011), resulting in acid sulphate soils and surface water acidification (Mosley et al., 2014). Salinity increased in the Southern Lagoon of the Coorong to 200g/L (2.5 times the modelled natural salinity of 80g/L) and water quality in the Northern Lagoon degraded (Kingsford et al., 2011). Policy responses during the Millennium Drought consisted of a number of infrastructure interventions, particularly temporary infrastructure to prevent further water quality reductions (Table 4.2). Water purchases were also made by the SA government to meet critical human and ecological water needs (Banerjee et al., 2013).

The severe loss of MDB ecosystem services (Banerjee and Bark, 2013) catalysed the most significant reform in Australian water reform history and continued the policy transition to sustainability in the MDB (NWC, 2011), shown in Figure 4.2. The *Water Act (Cwth)* was passed in 2007, at the height of the Millennium Drought, and furthered the push for sustainable water-use in the Basin, as well as an increasing emphasis on demand-based instead of supply-led solutions (Settre and Wheeler, 2016). In

the ongoing transition to the sustainability phase, the CLLMM has seen considerable institutional movement from a state obligation to a MDB environmental objective. Water purchases by the CEWO to be delivered for environmental benefit in the CLLMM demonstrate this growing commitment (Wheeler, 2014). However, as can be seen in Table 4.2, the prolonged Millennium Drought was predominately managed by infrastructure intervention. Further, the 2014 *Lake Albert Scoping Study Options Paper* (the Scoping Study) developed to investigate future CLLMM management strategies, reported six possible options, four of which were infrastructure-based (DEWNR, 2014a). This indicates that infrastructure remains a significant policy focus. The following sections discuss the use and costs of CLLMM infrastructure interventions in the context of active and reactive management, along with the future transition to sustainability.

4.5 History of Infrastructure Interventions in the CLLMM

Management of the CLLMM has been characterised by a sequence of active and reactive infrastructure interventions, first as active interventions to supply consumptive water demands, and more recently as reactive emergency drought responses. Table 4.2 summarises key CLLMM infrastructure interventions, purpose of intervention, date of construction/removal, current status and costs. CLLMM infrastructure intervention began in 1862 with the construction of drains in the south-east of South Australia. The drain construction, an active intervention, was rooted in the expansionary paradigm of Australian water management, and was implemented to remove saline groundwater and excess surface water by redirecting overland flows from the Coorong to the sea. Directing overland flows to sea has meant the partial loss of fresh water inflows into the Coorong and a reduction of water quality in the Southern Lagoon (Gell and Haynes, 2005). To mitigate salinity in the Southern Lagoon, a reactive intervention of altering the existing drainage network was approved in 2014, to direct 26GL/year of drainage into the Coorong Southern Lagoon (Hunter, 2014a)

Alongside the drainage scheme was the construction of five tidal barrages (active intervention) near the Murray Mouth, designed to impound fresh water supplies for irrigation and prevent seawater ingress. Flows to the Murray Mouth pass through a series of barrage gates (Lester and Fairweather, 2009) and are managed through seasonal operations, which play a large role in system hydrodynamics (Webster, 2010). The barrages actively changed the CLLMM's ecological character by extending the average areal extent of the freshwater lakes by 6% (CSIRO, 2008a) and maintaining lake levels at 0.75m Australian Height Datum (AHD) (Kingsford et al., 2011). Isolating the barrages' impact from upstream regulation, water abstraction and overall reduced water availability is a complex task. However, comparing the long-term median flow between 1963 and 2008 of 3,271GL/year, to the median flow through the barrages between 1999 and 2008, Lester and Fairweather (2009) find barrage flows in the latter period were 208GL/year (6% of previous total median flow). Both the drainage network and the barrages were active CLLMM interventions.

Table 4.2: Infrastructure Interventions in the CLLMM

Type of Intervention	Infrastructure Intervention	Purpose	Installed/ Completed	Current Status	Construction Cost ($ million)	Removal Cost ($)
Active	South-East Drainage Scheme	Drain excess water to the sea to allow agricultural expansion	Ongoing construction 1862 - 2011	Drains remain, partial redirection	Upper South East Drainage Program (1990 – 2011): $73 million (DEWNR, 2013); previous costs not available	N/A
	Five barrage structures	Preserve permanent freshwater pools in Lower Lakes for consumptive use	1939	Structures remain	Estimated $1.5 million in 1940 dollars (Kingsford et al., 2011)	N/A
	Narrung ferry causeway	Enable vehicle transport across the Narrung Narrows	1963	Structure remains	Not available	N/A
Reactive	Dredging the Murray Mouth	Maintain hydrological connectivity between Coorong and Southern Ocean. 5.6 million m^3 of sand removed between 2002-2010	1981 2002-2010 2015-2016	Ongoing	1981 cost not available; $6 million p/a between 2002-2010 (DEWNR, 2014b), estimated $48 million in total; $4 million allocated in 2014 (Birmingham and Hunter, 2014)	N/A
	Barrage fish passages	Rock-ramp and vertical-slot fishways	2001-2006	Remains	Part of $45 million Sea to Hume Dam program (Barrett and Mallen-Cooper, 2009)	N/A
	Potable and irrigation water pipelines	167km potable water pipeline; 110km irrigation water pipeline, 11km and 12.6km potable water pipeline extension	2008-2009	Remains	$127 million (Gross et al., 2012)	N/A
	Temporary Flow Regulators	Low flow regulator at Currency Creek and Goolwa Channel near Clayton	2009	Removed 2012-2013	$26 million (Gross et al., 2012)	$3.8 million (DoE, 2011b)
	Narrung bund and pumping to Lake Albert	Maintain higher water levels in Lake Albert, freshwater exchange between Lake Alexandrina and Lake Albert via pumping	2009	Pumping stopped in 2009, bund removed 2011	$14 million (Gross et al., 2012); Pumping costs unavailable	$2.5 million (DoE, 2011a)
	Lakes Cycling	Using barrage operations to manipulate lake levels by 200-300GL/year to manage salinity in Lake Albert	2014	Ongoing	<$5 million (Beal et al., 2014). Operational costs expected to be funded by SA Water	Ongoing
	South East Flows Restoration Project	Redirecting 26GL/annum to the Coorong Southern Lagoon via the existing Upper South East Drainage Network	2014	Ongoing	$60 million committed as part of the CLLMM Restoration Project (Hunter, 2014a)	Ongoing
	Total				$403.5 million	$6.3 million

Source: Own Table. References cited within.

Recent modern CLLMM infrastructure interventions have their genesis in *The Living Murray* program (2004), which committed $150 million in capital expenditure (Pittock and Finlayson, 2013). The political preference for infrastructure interventions was sustained during the Millennium Drought, which initiated a 'crisis management' approach to interventions in order to mitigate the threat of ecosystem collapse (Gross et al., 2012), many of them temporary in nature. It is evident that modern CLLMM infrastructure interventions are largely reactive solutions, which are implemented either in emergency response to the threat of ecosystem change (e.g. drought); or to maintain the social-ecological states created by existing active interventions (e.g. new pipeline to replace Lower Lakes consumptive water supply). A key example of emergency drought infrastructure is the construction of a bund, a reactive intervention, in 2009 between Lake Albert and Lake Alexandrina (Narrung Narrows) designed to prevent further acidification of Lake Albert during a time when water levels were the lowest on record (Mosley et al., 2014). Exchange between the two lakes was managed by pumping operations (DEWNR, 2014a). The bund achieved the objectives, but also contributed to further ecosystem fragmentation (Pittock and Finlayson, 2013) and significant sediment accumulation in the Narrung Narrows. The bund was implemented and removed at a cost of $16.5 million dollars. Another key example are the temporary flow regulators at Clayton and Currency Creek (reactive intervention). Upon return to drought conditions in the future, if flow cannot be provided to prevent ecological decline, these temporary flow regulators will be reconstructed to prevent acidification events (MDBA, 2014a). Additional reactive drought interventions include the dredging of the Murray Mouth and construction of a new pipeline to the River Murray.

The legacy of reduced ecosystem resilience (Gross et al., 2012), the considerable costs involved, and future climactic uncertainty make CLLMM management a significant challenge. Future options were canvassed in *The Lake Albert Scoping Study Options Paper,* which was commissioned by the South Australian government in 2014. The management options proposed are within the context of preserving the CLLMM in the current state created by active interventions (e.g. barrage construction) and can therefore be classified as reactive interventions. Six management options were considered in addition to a no-action base case:

a) dredging the channel between Lake Albert and Lake Alexandrina (Narrung Narrows);
b) full or partial removal/modification of the Narrung Narrows causeway;
c) construction of a pipe connection between the Coorong and Lake Albert;
d) construction of a channel connection between the Coorong and Lake Albert;
e) construction of a permanent structure in the Narrung Narrows; and
f) manipulation of lake levels using barrage structures, also known as *lakes cycling*.

Lakes cycling, which involves the strategic raising and lowering of lake levels, using the barrage structures to promote the flushing of saline water, has been selected and implemented (Hunter, 2014b).

Modelling results suggest that lakes cycling would reduce daily mean salinity levels by 160 EC[23] relative to the base case (DEWNR, 2014a). The selection of lakes cycling over the construction of new infrastructure, in addition to being a function of cost, may indicate an increasing political preference for non-infrastructure solutions.

It is evident there is a clear sequence of active and reactive CLLMM infrastructure intervention. This sequence is consistent with the classification of infrastructure interventions as mal-adaptations (Pittock and Finlayson, 2013), owing to their generation of unanticipated costs and subsequent ongoing interventions to mitigate negative impacts. Further, Table 4.2 demonstrates there is substantial cost associated with CLLMM infrastructure investment. There is evidence that hastily implemented drought infrastructure interventions (Gross et al., 2012) have rarely been subject to cost-benefit analysis (Pittock and Finlayson, 2013) and have high opportunity costs owing to the possible investment in alternative long-term no-regrets solutions.

4.6 Unsuccessful Infrastructure Interventions

Lessons may also be learned from considering unsuccessful proposals, as shown in Table 4.3. These interventions were dismissed when inflows returned after the Millennium Drought but nevertheless are likely to resurface under future droughts, so it is important to understand them. One notable unsuccessful intervention is the Pomanda Island weir, temporary or otherwise, proposed in 2009. The proposed weir would separate Lake Alexandrina from the lower reaches of the River Murray, to prevent salinization of potable water supplies in the event of ongoing high salinity levels in the Lower Lakes or barrage removal. The Pomanda weir was a costly reactive intervention ($160 million) and was met with largely negative stakeholder views (Kingsford et al., 2009; Gross et al., 2012). Initial site preparations were undertaken, but later abandoned. Another notable proposal was the Meningie, Narrung and Lakes Irrigators Association Five Point Plan (hereby: MNLIA Plan). The MNLIA Plan proposed a range of infrastructure interventions to restore CLLMM to health, namely: a) removal/modification of Narrung Causeway; b) clearing the Narrung bund remnants; c) dredging the Narrung Narrows; d) connecting Lake Albert and the Coorong with a pipe or channel; and e) redirection of USED scheme flows to the Coorong Southern Lagoon. Redirection of USED flows is ongoing and the remaining MNLIA Plan points are shown in rows one to three of Table 4.3. Constructing a connector between Lake Albert and the Coorong was determined to be the only infrastructure intervention that would improve salinity, albeit at a cost of $19 million (DEWNR, 2014a). Analysis suggested that benefits only justified costs if the area irrigated increased considerably (Beal et al., 2014). Land-use data indicates that there has been a transition from irrigated agriculture to dry land grazing; hence it is unlikely that benefits will ever exceed costs.

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[23] Electro conductivity (EC) is a measurement of soil and water salinity. 1 EC is equal to 1 micro Siemens per centimetre (µS/cm).

Table 4.3: Unsuccessful CLLMM Infrastructure Proposals

Proposed Intervention	Purpose	Type of Intervention	Effectiveness[1]	Community Support[2]	Costs (qualitative and $)
Removal/ modification of Causeway[3]	Increased exchange between Lake Albert and Lake Alexandrina	Reactive	Causeway removal and dredging to promote connectivity will have negligible effect on the salinity of Lake Albert (DEWNR, 2014a)	62% support, 16% against	Negative impact on ferry operations, increases travel times. Alternative measures to replace ferry (e.g. bridge) would incur additional costs
Dredging the Narrung Narrows[3]	Remove impediments to the freshwater exchange between Lake Albert and Lake Alexandrina	Reactive		82% support, 9% against	Estimated $120 million plus ongoing maintenance (Beal et al., 2014)
Pipe or Channel Coorong Connector[3]	Reduce salinity levels in Lake Albert to 1,800 EC by pumping 3000 GL/year over 12 months (DEWNR, 2014a)	Reactive	Effective at reducing salinity levels (DEWNR 2014a). Low benefits under low and medium climate change scenarios and detrimental impacts under high scenarios (Gross et al., 2012)	63% support, 16% against	Estimated $19 million (+/- 30%) (DEWNR 2014)
Permanent Structure in the Narrung Narrows	Enhance flow exchange in lower lakes (DEWNR, 2014a)	Reactive	Limited effectiveness in reducing salinity in Lake Albert (DEWNR, 2014a)	45% support, 37% against	N/A
Pomanda Island Weir[4]	Preserve freshwater pools in lower reaches of River Murray if there is permanent saline intrusion in the Lower Lakes	Reactive	Designed as a temporary structure Effective in preventing saline intrusion	Unknown	$160 million (Kingsford et al., 2011)
Southern Lagoon Pumping[4]	Pump hyper-saline water from the Coorong to the Southern Ocean	Reactive	Preliminary assessments suggest project is feasible (MDBA, 2014a)	Unknown	Estimated $27 million. Construction of pumping station, pipeline pumping costs not available
Total					**>$326 million**

Source: Own Table. References cited within.

Notes: 1. Climate scenarios are defined by CSIRO (2008) for the year 2030 relative to 1990. Low change: +1.03°C; Medium change: +1.6 °C; High change: +4°C;
2. Community support levels obtained as part of the Lake Albert Scoping Study (DWENR 2014). Two focus groups and 10 interviews with community members were undertaken. Respondents answering 'do not know' were excluded (SquareHoles 2013);
3. Proposal included in the MNLIA Plan; 4. Not addressed in Lake Albert Scoping Study.

4.7 Alternative Infrastructure Proposals

There are other proposals not included in the *Lake Albert Scoping Study* or subject to feasibility studies elsewhere. These proposals involve alteration to the barrage structures, which would impact on the ecological state of the Lower Lakes (active interventions). Long-term regional planning for the sustainability of fresh water dependent industries and ecosystems must consider future barrages viability, costs of managing the aging structures, and alternative proposals.

Barrage automation is identified as the most significant short to medium-term option to achieve environmental watering and fresh water preservation objectives. Automation would allow the transparent operation of the barrage gates to increase operational flexibility and provide greater sensitivity to ecological needs (Jensen et al., 2000; MDBA, 2013). Transparent barrage operations could include opening the barrages to increase connectivity and promote estuarine mixing (Gross et al., 2012). Partial automation of barrage gates has occurred at Mundoo, Ewe Island and Tauwitchere barrages, but further work is required (MDBA, 2014b). Improvements to existing barrage operating procedures to achieve ecological demands, including gate opening strategies, are also ongoing. Lakes cycling, selected by the SA government to address Lake Albert's salinity issues, could be improved with remotely operated barrage systems (DEWNR, 2014a). Barrage investment automation is estimated to cost $5 million (Beal et al., 2014) and investment involves the risk of obsolescence in future climate scenarios that result in either low flows and/or sea level rises (Gross et al., 2012).

Long-term, there is likely to be considerable difficulty in maintaining the barrages in future climate scenarios causing sea level rises (Gross et al., 2012) and it is unlikely the barrages will be able to be preserved in their current state (Matthews, 2005). Possible changes to the structures include opening, removal, relocation or fortification of the barrages. Opening the barrages or the complete removal of the structures has been raised as a long-term option (Jensen et al., 2000). The resulting unrestricted flow from the Coorong to the Lower Lakes would likely change the Lower Lakes from fresh to estuarine conditions (Kingsford et al., 2009), which would breach the Australian governments' Ramsar Convention obligations. If the barrages were removed, the construction of Pomanda Island weir would be inevitable to prevent saline intrusion into the lower reaches of the River Murray (Kingsford et al., 2009; Gross et al., 2012). The relocation of the barrages further upstream to enlarge the estuarine area and create evaporative savings is an alternative option (Jensen et al., 2000). Little attention has been given to this proposal owing to considerable construction difficulty and expense. Coastline fortification to maintain freshwater supplies and prevent barrage obsolescence due to rising sea levels has been proposed (Matthews, 2005; Gross et al., 2012). A step in this direction was taken when the barrages were sealed for 630 consecutive days during the Millennium Drought (Phillips and Muller, 2006). The construction of a single 13km long high barrage may be possible (Gross et al., 2012), but has received little attention and would be expensive. Training walls at the Murray Mouth to stabilise and maintain

the channel entrance (DoEH, 2009), by preventing coastal movement and sand build-up, is an alternative that has been given little attention.

4.8 Discussion and Possible Alternatives

Through the progress of recent MDB water legislation, there has been a back and forth between policy preferences for infrastructure-based management and those for market-based water management (Loch et al., 2014b). In the CLLMM, a preference for infrastructure intervention has dominated, initially to provide supply for consumptive use and more recently as a drought management approach. This has occurred in an observable sequence of active and reactive interventions, which have been implemented at considerable expense and opportunity cost.

Active interventions were motivated by the developmentalist ethos of early Australian water management, while more recent reactive solutions have been deployed to prevent unwanted ecosystem change (e.g. acidification), to improve ecosystem states created by active interventions (e.g. barrage fishways installed) or to maintain the ecosystem services provided by the active interventions (e.g. pipeline to maintain irrigation supply). Unanticipated costs and perverse impacts by active interventions confirm CLLMM infrastructure interventions as overly narrow (Gross et al., 2012) and inflexible mal-adaptions (Pittock and Finlayson, 2013; Miloshis and Fairfield, 2015).

The sequence of interventions from active to reactive is clear, for instance barrage installation and the USED scheme are the genesis of CLLMM infrastructure, and have spurred significant reactive interventions. For example, barrage construction (active), allowed limited mixing between estuarine and freshwater conditions, resulted in the need to install fish passages to the sea (reactive). Reduction of flows into the Coorong due to the USED scheme (active), resulted in redirection of drainage flows into the Coorong Southern Lagoon (reactive). Options for future management also follow this sequence. For example, removal of the barrages (active) would likely cause salinization of the Lower Lakes and would require an additional structure to prevent sea-water ingress (reactive) into the River Murray. The adverse ecological impacts from the Pomanda Island weir would likely require future physical intervention.

Many of the reactive infrastructure interventions during the Millennium Drought were implemented to attempt to maintain ecological health. One such example is the dredging of the Murray Mouth to keep it open. In 2015 dredging recommenced. The approach of reliance on interventions such as dredging raises wider questions: if physical measures are used to maintain the channel opening, is the indicator being met? This research suggests that this is not the case, but rather furthers the trend of addressing symptoms of ecological decline and overshadowing causes (Kingsford et al., 2011; Pittock and Finlayson, 2013). Under the full implementation of the *Basin Plan* in 2019, the risk of Mouth closure is expected to be reduced (Bark et al., 2013) and dredging only required five out of 100 years (DEWNR, 2013), but this remains to be seen.

CLLMM infrastructure has occurred at considerable cost. Table 4.2 depicted a total infrastructure cost greater than $410 million since initial interventions began. Over $266 million was spent between 2002 and 2013 (44% of this was part of ongoing projects predating the Millennium Drought). The estimated cost of proposed/abandoned infrastructure interventions is greater than $326 million. Our results provide an update of previous estimated CLLMM infrastructure costs. For example, Kingsford et al. (2011) and Banerjee et al. (2013) estimate infrastructure expenditure of $211 and $216 million, respectively, but do not account for the cost to remove temporary infrastructure, ongoing projects (e.g. Sea to Hume fishway program) and recent Murray Mouth dredging in 2015 and 2016.

CLLMM infrastructure interventions also have high opportunity cost. This is particularly true for temporary infrastructure, which incurs expenditure for both construction and removal. For example, when drought returns, the Clayton and Currency Creek temporary regulators will be reconstructed if flows are not provided (MDBA, 2014a). The opportunity cost therefore rises as there are more cost-effective/efficient demand-based policies available. Further, the significant attention to infrastructure options in the *Lake Albert Scoping Study* and the recognition of temporary flow regulator reconstruction when drought returns, highlights that infrastructure intervention will likely continue, indicating lessons not yet learned from previous droughts.

4.8.1 The Transition to Sustainability

As part of the *Basin Plan*, the CLLMM has seen considerable institutional movement from a state obligation to a national objective. A greater focus on sustainability is clear in the commitment to deliver environmental water to the CLLMM. However, we argue that the ongoing focus on infrastructure interventions and the 'crisis management' approach to decision-making during drought periods has limited the transition to sustainability in the MDB.

Lakes cycling, which has been selected from the *Scoping Study* options, may be seen as a step towards more CLLMM adaptive management, as it does not require new infrastructure construction. The practice relies on the barrages' operational capacity and design of the structure, and is expected to deliver salinity mitigation benefits under normal flow conditions. However, DEWNR (2014a) reported that high salinity levels are unaffected during drought conditions, even with lakes cycling, as salinity is predominately dictated by freshwater availability. This raises considerable questions regarding the validity of implementing an option that fails to deliver during drought periods. We suggest that lakes cycling could be more suitably implemented when paired with demand-based management strategies available to state and federal governments.

As Baldwin et al. (2015) succinctly summarised: it is important to 'not drop the ball on water'. The Basin's water problems are not solved and the transition to sustainability has not been fully realised, nor is it yet clear if the realised phase will be necessarily sustainable. As part of the transition to sustainability, the health of the CLLMM has been established as a basin-wide environmental watering

objective, though it is not the only Ramsar listed wetland in the MDB. With continued (and as we argue, necessary) focus on CLLMM ecological health, the degree to which CLLMM environmental management is detracting from other internationally important Basin assets remains an unanswered question.

It is argued that demand-side management options are more in line with the transition to sustainability (Lane-Miller et al., 2013). As well as returning additional base-flow to the river through buying water entitlements from willing sellers (Wheeler, 2014), opportunity exists to expand the use of time-limited water products such as annual (allocation) environmental water trade, also known as counter-cyclical trade. Modelling results of annual trade suggest it is cost-effective in delivering environmental water (Kirby et al., 2006). Modelling also suggests it allows capacity to alter moderate flood event timing to prevent ecological decline (Connor et al., 2013). To date, no research has investigated the direct benefits of counter-cyclical trading to the CLLMM. An alternative time-limited product is entitlement leasing, which could be purchased by the government on a fixed-term contract. Wheeler et al. (2013) suggested that time-limited water products have helped structural adjustment and increased flexibility, and that in 2010 there was considerable irrigator willingness to engage in such products. This indicates significant scope for alternative water products to be considered as a means to acquire CLLMM environmental flows, as an alternative to further emergency drought infrastructure interventions, temporary or otherwise. Although there has been great debate over the impacts of water trade and returning water from consumptive to environmental purposes, evidence of social and economic harm is scarce (e.g. see Wheeler, 2014; Crase and Cooper, 2015 and Grafton et al., 2016 for greater discussion).

A perhaps more controversial strategy is the expansion of the South Australian environmental water holder. If severe drought conditions return, water will be needed to maintain lake levels above 0.0 AHD (MDBA, 2014a). Held environmental water is provided to the CLLMM by the CEWH, the *Living Murray*, a non-government organization (Nature Foundation, SA), and the SA government (DEWNR, 2015). Environmental water held by the SA government is available through the SA Minister for Water and the River Murray's managed wetland licence of 34.7GL administered by DEWNR. The active use of an independent statutory EWH is evident in Victoria (VEWH), which delivered 136.7GL in 2014-2015 (VEWH, 2015). Collaborating with the CEWH, state-based EWHs have the ability to provide water above minimum deliveries to key ecological priorities. It is acknowledged that the expansion of a SA-EWH as an independent statutory body would require considerable reform, in addition to interstate co-operation regarding the upstream storage of South Australian environmental water.

An additional option that is gaining greater attention is the recognition, delivery and co-management of flows to the Ngarrindjeri Nation to generate cultural and ecological ecosystem services. Although the *National Water Initiative* (2004) provided for improved Aboriginal involvement in water planning, in practice Aboriginal values have rarely been incorporated or considered (Tan and Jackson, 2013;

Jackson et al., 2015). Integration of cultural values in water planning presents a significant challenge to current management paradigms. For example, Bark et al. (2015) highlighted Aboriginal preference for system holism and connectivity, which is counter to widely employed site-specific management strategies and Eurocentric paradigms for perceiving and assigning value. The discussion around meeting cultural water demands is not isolated to the CLLMM, and comparable case studies exist more widely in the MDB. One example is the Yorta Yorta Nation who, as with the Ngarrindjeri Nation, advocate for allocation and delivery of cultural water, in their case to the Ramsar listed Barmah-Millewa Forest (Robinson et al., 2015). Although this presents a significant challenge, an example of successful participatory water planning is evident in the Tiwi Islands (Northern Territory) (Hoverman and Ayre, 2012). The development of an agreement between the CEWH and the NRA is a significant development, which formally provides for the transfer, delivery and monitoring of Commonwealth environmental water in the Lower Murray by the NRA (CEWH and NRA, 2015). The agreement also enables NRA participation in state water planning. Although Australia has seen a huge shift in the past decade in giving increased precedence to environmental flows, the next paradigm shift may be to recognise, both in law and practice, the importance of Aboriginal participation in water allocation decisions to maintain social, cultural and ecological health needs.

Regardless of future CLLMM management directions, all decisions must consider if the Lakes will be managed as fresh or estuarine pools. This predominately depends on the barrages future management in the face of rising sea-levels, reduced rainfall and surface water availability. To date, restoration interventions have rarely taken climactic uncertainty into account (Bark et al., 2013), inducing considerable risk in CLLMM management. The *Basin Plan* offered an opportunity to mainstream climate change risk-analysis into national decision-making (Kirby et al., 2014a; Pittock et al., 2014), but this was not fully realised and is instead to be managed through state water management plans (MDBA, 2012d). Further, no emphasis is given to barrage management in the *Basin Plan* beyond setting targets for annual flow objectives to the Murray Mouth (Commonwealth of Australia, 2012). Expenditure committed to infrastructure without due consideration of climate change impact and uncertainties may represent a sunk cost. Given the significant costs of CLLMM infrastructure intervention, the ongoing implications of mal-adaptations during (and after) drought, and the uncertainties of a changing climate, very careful scrutiny regarding the future adoption of various active and reactive infrastructure strategies is required.

4.9 Conclusion

This chapter has provided a critical review of CLLMM management, through the various phases of Australian water management. It is demonstrated that management has been dominated by infrastructure interventions, first as supply infrastructure embedded in the expansionary phase of Australian water policy, and more recently as reactive emergency interventions in response to the Millennium Drought. Infrastructure interventions have occurred in a sequence of active and then reactive measures, which have led to perverse impacts and unanticipated costs. This supports existing classifications of infrastructure interventions as overly narrow or mal-adaptations. Given that low flow conditions will become more frequent in the sMDB, it appears to be widely accepted that when drought returns temporary infrastructure solutions will be implemented if additional flows are not provided. Further, based on the demonstrated high cost of infrastructure and implications for ecological health, it is suggested that maintaining an infrastructure-based strategy in the future requires critical consideration and thorough attention to benefits and costs, impact assessment and uncertainty analysis – precautions that were not previously evident.

This chapter has argued that infrastructure investment has a high opportunity cost, owing to alternative pathways that can be pursued. Possible future options include expanding demand-management strategies, such as the water entitlement purchase program for cultural flows, and exploring time-limited environmental water acquisitions (e.g. allocation trade and short-term entitlement leases, or a formalised and proactive SA-EWH). The opportunity to engage and expand the role of traditional owners, the people of the Ngarrindjeri Nation, in future water planning as well as providing additional cultural flows has considerable merit, however will not be without its difficulties. Although water and salinity levels have improved since the end of the Millennium Drought, the dredging that restarted in 2015 indicates ongoing issues remain. Failure to implement interventions, which address the root cause of CLLMM ecological decline, will result in the hasty repetition of previous emergency responses when drought returns. Irrespective of future policy directions, intervention in the CLLMM must occur within the context of the ecological state of the Lower Lakes, which is intrinsically linked with barrage management and climate change impacts, two key aspects which have been under represented in management decisions to date.

CHAPTER 5: MDB Hydro-economic Modelling Literature Review

This chapter presents an expanded version of a published paper in *Water Economics and Policy (2017)*. The majority of this chapter appears as it was published, with the exception of additional material in the review of eco-hydrological and ecosystem service modelling, along with a more specific discussion of the 'gap' in hydro-economic modelling of ecosystem services. The original paper was prompted by a growing yet uneven literature base of hydro-economic models in the MDB, which have been increasingly used to analyse basin-wide water policy. Over 60 hydro-economic models in the MDB were analysed and discussed to identify key findings, innovations and avenues for advancement.

Through a critical review of ecosystem service modelling in the hydro-economic framework, two key gaps are identified. First, it is evident that water reallocation and *Basin Plan* modelling has focused disproportionately on the cost of reallocation to irrigated agriculture, while only a small amount of literature has addressed the ecological and economic benefits of reallocating water to the environment. Second, the existing HEM literature does not include adequate representation of ecological responses and ecosystem service dynamics, and in particular provides limited representation of inter-temporal linkages, biophysical thresholds, non-linearity and path-dependency for processes, other than those pertaining to irrigation. To address these gaps, a considerable advancement in the integration of ecological processes into hydro-economic modelling is required in order to provide a complete picture of environmental opportunity costs faced in water allocation decision-making.

Overall, this chapter is important in showing how policy questions discussed in Chapters 2-4 have been modelled in the MDB to date, identifying innovations and the limitations of current approaches, and demonstrating the gap that the subsequent HEM presented in Chapters 6 and 7 attempts to address.

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Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
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By signing the Statement of Authorship, each author certifies that:

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- the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

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5.1 Introduction

Hydro-economic modelling[24] (HEM) is the formal mathematical integration of relevant hydrological processes and the economic concepts of supply and demand for the purpose of informing effective integrated water resource management (IWRM). Hydro-economic modelling is a broad term that encompasses bio-economic approaches originating from the economics literature, which typically seek to quantify the value of human use of ecosystems for consumption and production (Braat and van Lierop 1987). HEM provides a unified approach for researchers to investigate the relationships and feedbacks between natural hydrological processes and human decision-making, which are inherent in all water resource systems. Importantly, the integration of hydrology and economics into a single methodological approach allows researchers and policy makers to address the question of economic efficiency in relation to water allocation (Elbakidze and Cobourn, 2013). HEM therefore addresses the non-trivial question of: *who is allocated how much water and when?*

Modern HEM has its roots in a range of theoretical advancements in the 1950s which led to the combination of water resource planning and welfare economics (Booker et al., 2012). Krutilla and Eckstein (1958) introduced the basin as the natural scale for HEM, challenging previous planning methods divided by sectors. This conceptual shift formally recognises that, in hydrologically connected systems, the actions of one user group affect the quality and quantity of water available to another user group (Bouwer et al., 2000). Since the 1980s, HEM has expanded in technical ability and scope (Booker et al., 2012), mirroring a range of industry trends including advancements in integrated systems, science and information technology.

In Australia, HEM has been spurred by significant water reform and increasing scarcity pressure due to drought. As previously discussed, water policy in the Murray-Darling Basin (MDB) has been characterised by a series of national water reforms and an increasing focus on achieving a sustainable balance between extractive use and the environment. These reforms cumulated in the passage of the *Basin Plan,* enacted in 2012 (Wheeler, 2014), which sets an upper limit on the amount of water that can be extracted for consumptive use from 2019. The reform process and current policy process are detailed comprehensively in Chapters 2, 3 and 4. While the MDB has received international attention for its innovative reforms and environmental policy (Garrick et al., 2011), HEM in Australia is not necessarily as well developed as it is elsewhere. This chapter summarises the current state of HEM in the MDB, identifies key findings and innovations, as well as providing suggestions for possible avenues for advancement.

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[24] In this chapter HEM is used to refer to both hydro-economic models and hydro-economic modelling.

5.2 Types of Hydro-economic Models

When developing a HEM, a number of decisions are faced regarding how the model is formulated and solved. Model formulation and decision rules implemented based on the modeller's perception of the problem (i.e. bounded awareness) (Grant and Quiggin, 2013) warrant considerable attention as they can contribute to divergent results from similarly parametrised models under the same scenarios. When such divergence occurs, model structure uncertainty is likely to be large (Uusitalo et al., 2015).

HEMs can be formulated either as rules-based simulation algorithms or objective driven optimization models. Simulation algorithms are useful for representing existing system rules and for examining the impact of possible events relative to baseline conditions where optimal solutions can be identified from the simulated feasibility space using scenario analysis (Harou et al., 2009). For example, a simulation of the regional and national economic impact of drought relative to baseline was employed in the MDB by Horridge et al., 2005 and Wittwer and Griffith, 2011. Conversely, optimization models are driven by a maximization/minimization function bounded by constraints, and are typically used to inform system improvements or locating the optimal feasibility space for close simulation analysis (e.g. optimization of regional water allocation across the MDB as used by Mallawaarachchi et al., 2010). These model formulations can be used in isolation or simultaneously. Simultaneous uses of simulation and optimization approaches are common for groundwater analysis, incorporating sub-flow simulations and pumping optimization models (e.g. Khan et al., 2003). Multi-criteria optimization formulations to manage trade-offs are applicable in HEM (e.g. Szemis et al., 2013; Szemis et al., 2014). When assessing change, partial equilibrium (PE) models can be constructed to assess changes at the regional or sectorial level, often driven by a focus on changes to production outputs and farming revenue. A tabulated review of PE models in the MDB is shown in Table 5.1 and discussed in later sections. Wider second round impacts of change are simulated using computable general equilibrium (CGE) models, which address regional and national indicators of welfare. CGE models and results are represented in Table 5.2. Within models, the representation of time can be either static or dynamic. Static models typically optimise or simulate the net present value or annualised benefit of a management decision in a static time frame, prior and post a disturbance (e.g. policy change). Inter-temporal effects and interdependencies over multiple periods are represented in dynamic models (Harou et al., 2009). HEMs developed specifically to inform policy can be classed as decision-support systems (DSS).

5.3 Hydro-economic Modelling of the MDB

Hydrological modelling has a long history in the MDB, beginning with traditional rainfall and runoff, dam storage and water usage demand models which were required by engineers during the development phases of Australian water resources (Randall, 1981, Chapters 3 and 4). More recently, national water policy reforms and the transition to sustainable water-use have motivated the development of HEM to inform and question water policy at the basin scale. HEM in the MDB is a relatively new phenomenon.

61 out of 66 models reviewed were published since the year 2000, and publications have significantly increased following the *Water Act* in 2007 and leading up to the *Basin Plan* in 2012. The increased rate of HEM studies in the MDB is shown in Figure 5.1.

Figure 5.1 MDB Hydro-economic Publications by Year
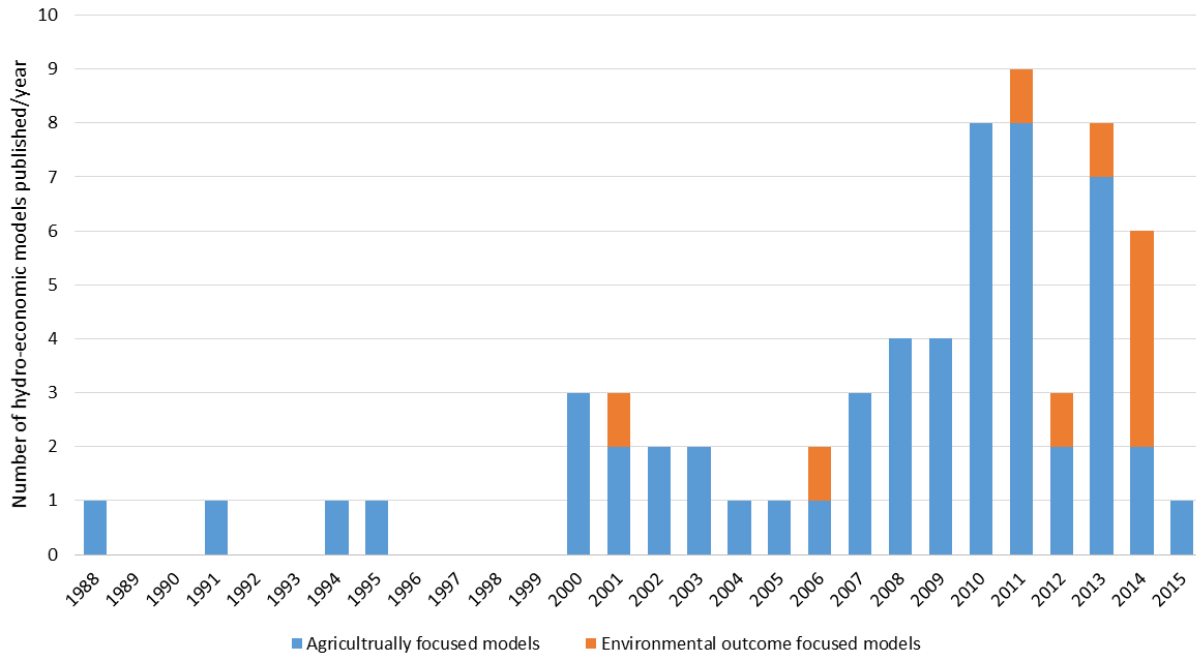
This chapter seeks to review all existing MDB HEMs by providing an overview of the current state of the literature and possible avenues for future advancement. For brevity and quality, the review is limited to published academic papers, government reports and commissioned studies (excluding working papers and consultancies due to a lack of publically available reporting).

The tabulated results of the structured literature review are provided in Table 5.1,Table 5.2 and Table 5.3. The first column in each of these tables is the broad modelling sector (agriculture or environment), noting that some multi-criteria optimization models could reasonably fit into either category (e.g. Grafton et al., 2011b; Ancev, 2014). The second column lists the key issue addressed in the model. Model name, if available, is reported in column three. Column four lists author and publication date. The fifth column describes the model type based on the solution algorithm, acknowledging there may be significant overlap or various means of classification in some cases. The sixth column provides a brief description of the model purpose. For optimization models, the objective function is listed in column seven. The geographical scopes of the models are listed in column eight.

As agriculture is the primary consumptive user of water in the MDB, it is no surprise that agriculturally focused models dominate the MDB HEM literature (56 out of 66 models reviewed). Agricultural issues have predominately been modelled using partial equilibrium optimization, as shown by comparison of

the number of CGE and PE models listed in Table 5.1 and Table 5.2. However, a significant proportion of studies commissioned to inform *Basin Plan* policy were CGE models (Wheeler, 2014) and, as such, have been reasonably influential in influencing Basin policy. Although CGE models are typically weak in hydrology and may not strictly classify as HEMs, these models have been included in this review when they have been used to address changes to water policy or climactic conditions. It was found only a small portion of MDB models are environmentally focused, and are concerned with proposing or assessing management strategies to improve the health of flow dependent ecosystems. Though environmentally focused models are fewer, an increasing trend in these types of models in recent publications can be seen in Figure 5.1.

5.4 Agricultural Focused Modelling

This review identifies four key themes of agriculturally focused HEM and discusses these aspects in the following sections. A cross-cutting theme in all agricultural models is the challenge of obtaining reliable time-series data of sufficient length and quality for calibration and validation. Ideally, calibration of physical system models is undertaken using a long time-series of observed data to capture natural variation. Adequate quality and quantity of data for model calibration and validation is required to increase the reliability and confidence in model outputs. In the MDB, agronomic data is sparse and usually sourced from Australian Bureau of Statistics (ABS) agricultural censuses.[25] In low data cases, positive mathematical programming (PMP) (Howitt, 1995) can be used to calibrate mathematical models to observed behaviour for a historic reference point in time, such as a census year, using calibration constraints that bound activity to the observed reference point level (Howitt, 1995). The use of PMP does not require a series of observations to reveal economic behaviour (Frahan et al., 2007) and as such has been employed in case of the MDB (e.g. ABARE-BRS, 2010; ABARES, 2011; Connor et al., 2012; Qureshi et al., 2013b; Whitten et al., 2015). However, PMP calibration introduces an optimization bias to baseline condition and suffers from reduced output reliability when applied outside the calibration baseline (Doole and Marsh, 2013). As the past may no longer be representative of the future due to a changing climate, calibration of historical data without due consideration for a stochastic future may introduce risk to the implementation of decisions and policies informed by results of HEMs. Further calibration difficulties arise owing to the complexity of natural resource systems, differing spatial and temporal scales across datasets, and policy drivers that cannot be easily validated (Letcher and Jakeman, 2003). In cases where validation data is available, reliability of model outputs can be assessed and parameters iterated if required. Fifteen percent of MDB models report undertaking a validation process. The following sections detail four key themes in agriculturally focused HEMs in the MDB.

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[25] Agricultural censuses are currently undertaken every five years in Australia.

Table 5.1: Partial Equilibrium Agriculturally Focused MDB Hydro-economic Models

Focus	Issue	Model	Author and Date	Model Type	Description	Objective	Scale
Agriculture	Ground-water salinity	SWAG-MAN Farm	Khan et al. (2003); Khan et al. (2008a); Khan et al. (2008b); Khan and Hanjra, (2008); Khan et al. (2009); Khan et al. (2010)	Dynamic salt and water balance	Determines optimal land uses (Khan et al., 2003) to maintain acceptable levels of salinity (Khan et al., 2008a); evaluates the impact of on-farm interventions (Khan and Hanjara, 2008), permanent water transfers (Khan et al., 2009) and property right regimes (Khan et al., 2010) on groundwater salinity and farm profitability. Links paddock scale salinity to regional targets (Khan et al., 2008b)	Maximise agricultural returns	Coleambally and Murrumbidgee (Khan et al., 2003); MDB (Khan et al., 2008a), Coleambally (Khan et al., 2008b; Khan and Hanjra, 2008, Khan et al., 2010); Murrumbidgee (Khan et al., 2009).
		-	Tisdell (2000)		Investigates groundwater induced soil salinization and farm-scale production under various policy scenarios		Murrumbidgee Catchment
		SALSA	Heaney et al. (2000); Bell and Klijn (2000); Heaney and Beare (2001); Heaney et al. (2001)	DSS	Assesses targeted reforestation (Heaney et al., 2000) and improved irrigation efficiency (Heaney et al., 2001) as salinity management tools; assesses impact of water trade and efficiency upgrade (Heaney and Beare, 2001) and land use options (Bell and Klijn, 2000) on river flows, salinity and agricultural returns	N/A	Macquarie-Bogan Catchment (Heaney et al., 2000); Riverland (Heaney et al., 2001); MDB (Bell and Klijn, 2000; Heaney and Beare, 2001)
	Water trade		Bell and Blias (2002); Heaney et al. (2004)		Evaluates cost of water entitlement trading impediments and the salinity cost of water trade (Bell and Blias, 2002); estimates the impact of water charges on gains from trade (Heaney et al., 2004)	Maximise agricultural returns	sMDB
	Climate change		Beare and Heaney (2002)		Investigates infrastructure upgrades and water trade as possible climate change adaptations		MDB
		-	Jiang and Grafton (2012)	Static linear optimisation	Estimates the economic impact of climate change scenarios on agriculture with and without inter-regional trade		
		-	Crean et al. (2013)	State-contingent model	Compares state-contingent and expected value model results in modelling climate uncertainty		Central-west NSW
		-	Adamson et al. (2009)		Assesses the impact of climate change and adaptation on irrigated agriculture		MDB
		-	Quiggin et al. (2010)		Analyses effects of climate change on irrigated agriculture for a range of adaptation and mitigation options		

			Reference	Method	Description		Location
		-	Qureshi et al. (2013a); Qureshi et al. (2013b)	Positive mathematical programing	Estimates the economic impact of climate change and adaptation options on irrigated agriculture (Qureshi et al., 2013a) and Australia's food production/exports (Qureshi et al., 2013b).		
		-	Qureshi and Whitten (2014)		Estimates the effectiveness of crop selection and water markets as climate change adaptation options		
		-	Rowan et al. (2011)	Dynamic optimisation	Investigates infrastructure investment and farm viability under climate scenarios at a farm scale		Hypothetical MDB Farm
		-	Connor et al. (2012)		Assesses impacts of reduced supply reliability and increased salinity on irrigated agriculture		Lower Murray-Darling Basin River
	Land use change	Farm model: MIDAS	Finlayson et al. (2010)		Quantifies changes in catchment water yield and agricultural profit in response to land use change		Little River Catchment
		Hydro model: CAT	Nordblom et al. (2010)		Investigates strategic land use change to improve catchment water yield, salt loads and impacts on agriculture		Bet Bet Catchment
		-	Schrobback et al. (2011)	State-contingent model	Investigates impacts of incentivised reforestation on agricultural land use decisions		MDB
	Water allocation	Hydro model: IHACRES	Letcher and Jakeman (2003)	Dynamic optimisation	Simulates economic and environmental trade-offs for a number of water allocation options		Namoi River
	Salinity	-	Quiggin (1988); Quiggin (1991)	Sequential non-linear; Dynamic programming	Investigates how property right regimes (Quiggin, 1988) mitigation works and new technologies (Quiggin, 1999) impact farm water-use and salinity levels		Murray River
		-	Adamson et al. (2007)	State-contingent model	Demonstrates application of state-contingent approach in linear and non-linear models		MDB
	Water trade	-	Hall et al. (1994)	Non-linear optimisation	Estimates the effects of water pricing, trading and banking on irrigated agriculture		sMDB
		-	Qureshi et al. (2009)		Estimates the costs of temporary water trade restrictions and state non-participation in the water market		
	MDB Plan	-	Qureshi et al. (2007)		Examines the impact of environmental water acquisition strategies on agricultural water-use and income		

		-	Qureshi et al. (2010)		Compares impact of infrastructure and market-based reallocation policies on water-use, return flows, and land retirement		Murrumbidgee Catchment
		-	Grafton and Jiang (2011)	Static linear optimisation	Estimates the economic effects of water buy-back on irrigated agriculture		
		-	Mainuddin et al. (2007)	Non-linear equilibrium	Estimates the impact of water buy-back strategies on irrigated agriculture		
		-	Kirby et al. (2013); Kirby et al. (2014a)	Hydrological network simulation	Assesses the impact of water-use reductions and climate change on river flows (Kirby et al., 2013) and flow sensitivity (Kirby et al., 2014a)		
		RSMG model	Mallawaarachchi et al. (2010)		Estimates the regional impacts of SDLs and optimises water allocation across the MDB for SDL scenarios		MDB
		RSMG model	Adamson et al. (2011)	State-contingent model	Assesses impacts of stochastic water variability on the implementation of the *Basin Plan*		
		-	Adamson and Loch (2013)		Examines infrastructure efficiency upgrades on irrigator adaptation and return flows (Adamson and Loch, 2013), consumptive water demands and environmental water availability (Loch and Adamson, 2015)		
		-	Loch and Adamson (2015)				
	Conjunc-tive use	HYDROLOG, AQUIFEM-N	Chiew et al. (1995)		Determines sustainable long-term pumping yields and economic benefits of conjunctive use scenarios	N/A	Campaspe Catchment
	Water allocation	Hydro model: IHACRES	Croke et al. (2006)	Hydrological network simulation	Simulates flow and diversions for assessment of various water allocation options		Namoi Catchment
	Water trade	Hydro: REALM	Zaman et al. (2009)		Simulates short-term and long-term third party impacts of temporary water trading		Goulburn-Broken Catchment
	IWRM	Hydro model: Source IMS	Welsh et al. (2013)		Simulates surface and groundwater flow to inform integrated water management decision-making		Murray River; Namoi River

Source: Own Table. References cited within.

Table 5.2 General Equilibrium Agriculturally Focused MDB Hydro-economic Models

Focus	Issue	Model	Author and Date	Model Type	Description	Objective	Scale
Agriculture	Drought	TERM-H2O	Horridge et al. (2005); Wittwer and Griffith (2011)	CGE	Simulates the impacts of 2002-03 drought (Horridge et al., 2005) and 2006-09 drought (Wittwer and Griffith, 2011) on agriculture and GDP	N/A	MDB, Australia
	MDB Plan		Wittwer (2010); Wittwer (2011); Wittwer and Dixon (2013)		Simulates the impact of a range of SDL volumes and compensation schemes (Wittwer, 2010; Wittwer, 2011) on agriculture and GDP; assesses the investments in water purchases and irrigation efficiency upgrades (Wittwer and Dixon, 2013)		MDB, Australia
			Dixon et al. (2011)		Simulates the impact of Commonwealth government water purchases under temporary drought conditions		sMDB
			Banerjee (2015)		Investigates the regional and national economic impact investment in irrigation efficiency upgrades		Murrumbidgee Catchment, Australia
		WTM, AUSREGION	ABARE-BRS (2010); ABARES (2011)		Estimates the long-run (2010) and short-run (2011) impacts of reduced water availability as a result of the SDLs on agriculture and GDP		MDB, Australia
	Climate change	GCCR Model	Garnaut (2008)		Identifies medium to long-term policies based on climate change impacts on sectors of the Australian economy		Australia

Source: Own Table. References cited within.

5.4.1 Basin Plan Modelling

The reallocation of consumptive water to the environment as part of the *Basin Plan* and the resulting concern of *Basin Plan* impacts to the agricultural sector has been a key HEM focus in the MDB. Around half of studies published after 2007 are focused on estimating the impacts of the *Basin Plan*. Modelling *Basin Plan* impacts is a challenge for HEM modellers, as it requires modelling under significant uncertainty regarding the largely unpredictable behaviour of irrigator adaptation, the stochastic nature of the climate, and the institutional setting in which the water reallocation process will be implemented, including possible SDL revisions and market or infrastructure prioritization.

Models addressing *Basin Plan* impacts can be roughly divided into two categories: whole-of-economy CGE simulation models designed to investigate the regional and national impacts of water recovery and infrastructure investment, and PE models designed to investigate the impact of reduced water availability on the irrigation sector. The CGE group of models converge on the result that there is little change in overall economic activity in the Basin and nationally, with fully implemented SDLs. The predicted small changes to regional economic activity as a result of the SDLs are explained by a number of factors. One of these factors is that the CGE models differ in model structure from PE models by assuming the money earned from the sale of water from willing sellers to the government is an input to the regional economy, as is government funding for investment in irrigation infrastructure. A caveat to the CGE findings is that *Basin Plan* impacts on smaller regional communities with high reliance on irrigation expenditure could be more acute (ABARES, 2011).

The long-term time horizon adopted by ABARE-BRS (2010) CGE model has been criticised for underestimating the short-term impacts of the *Basin Plan* (Jiang, 2011). It is reasonable to expect that the economic effects of the *Basin Plan* on regional communities may vary temporally, such that the short-term impact is larger where production variables remain unchanged. However, the ABARES (2011) short-run results are largely consistent with the ABARE-BRS (2010). The results derived from the CGE group of models are further supported by Mallawaarachchi et al. (2010) along with a number of other related studies, and suggest that the reduction in consumptive water availability is likely to induce a less-than-proportional reduction in irrigation revenue.

The results from the PE optimization models are comparable, though indicate slightly larger economic impacts of the *Basin Plan* than the CGE modelling predictions. Grafton and Jiang (2011) develop a linear optimization model to estimate the impact of various SDLs levels. A key difficulty of linear optimization models of production is that they do not permit factor substitution within a given activity, resulting in assumptions of fixed water-use coefficients. This is a simplified abstraction of reality, which leads to an only partially representative model structure and a corresponding reduction in result reliability to inform policy. Non-linear optimization approaches with greater consideration of irrigator adaptation used to estimate the impacts of the *Basin Plan* were undertaken by Mainuddin et al. (2007);

Qureshi et al. (2007); Qureshi et al. (2010); Mallawaarachchi et al. (2010); Adamson and Loch (2014) and Loch and Adamson (2015). Modelling results converge on the conclusion that the availability of adaptation strategies, such as water trade, can reduce the decline in regional farm profits which might otherwise occur under the SDLs, although impacts differ significantly across regions. The value of water trade in reducing the impacts of water reallocation is subject to institutional factors, such as trade restrictions between regions and hydrological barriers (Mainuddin et al., 2007), which are difficult to account for due to high data requirements. Using a non-linear partial equilibrium optimisation model, Qureshi et al. (2009) determined that MDB trade-limiting regulations imposed an annual cost of AUD$17 million. Qureshi et al. (2007) and Mainuddin et al. (2007) further show that target reallocation can significantly reduce water reallocation opportunity cost.

Investment in on-farm infrastructure is an additional mechanism to reallocate water to the environment as part of the *Basin Plan.* Investment in irrigation infrastructure is widely criticised in the academic literature, as discussed in Chapter 3, although is less contentious at the political level compared to market-based recovery. Infrastructure investment has received comparably less attention in the HEM literature, with the exception of Adamson and Loch (2014); Loch and Adamson (2015); and The Enormous Regional Model-H2O (TERM-H20) modelling (e.g. Wittwer and Dixon, 2013; Banerjee, 2015). Adamson and Loch (2014) show there are potentially negative feedbacks from investing too heavily in irrigation efficiency, and Loch and Adamson (2015) argue infrastructure investments are at odds with the *Basin Plan* objectives to improve environmental outcomes, because of reduced return flows from improved on-farm water-use efficiency. However, Wittwer and Dixon (2013) show that during severe drought, the performance of infrastructure upgrades increases due to the increased price of water, but would also slow farm investment over time. A more comprehensive discussion of the issues associated with infrastructure-based reallocation and the rationale for market-based reallocation are provided in Sections 3.5 and 3.10.

5.4.2 Water Trade

Water trade changes the spatial and temporal pattern of water demand and is necessarily a focus of HEM. More than half of the models reviewed gave some consideration to water trading. A partial spatial equilibrium model developed by Hall et al. (1994) is an early example of water trade modelling, banking, pricing and purchasing for environmental needs in the MDB. Key findings from the trade literature are that water trade increases total gross farm margins (Hall et al., 1994) and increases the volume of water used in high value activities (Adamson et al., 2007).

In the *Basin Plan* focused models, discussed above, it was shown that inter-regional water trade can reduce the amount of irrigation income foregone when water is reallocated from agriculture to the environment. In the models investigating the benefits of inter-regional trade as an adaptation to the *Basin Plan*, tangible costs such as delivery constraint violations and salinity impacts are not wholly

accounted for when measuring costs and benefits. Water trade that increases salinity may not be viable in the long-term (Khan et al., 2009) due to reduced agricultural productivity and remediation costs. Water trade externalities are typically localised, varying in magnitude, site dependent and temporally lagged, posing a significant challenge to modellers concerned with estimating aggregated trade benefits.

Concern over third party impacts of water trading, including surface and groundwater salinity, has been the focus of a number of studies (e.g. Tisdell, 2001; Heaney and Beare, 2001; Bell and Blias, 2002; Khan et al., 2009; Zaman et al., 2009). Using a dynamic salt and water balance model, Khan et al. (2009) show that if water is traded out of mature irrigation areas, minimum irrigation intensities to flush salt from root zones are unlikely to be met, resulting in reduction of resource quality and agricultural production in high salinity impact areas.

As with *Basin Plan* modelling, water trade modelling also requires multiple assumptions, including free and open trade of water in hydrologically connected catchments, and profit maximizing behaviour at the farm scale which forms the objective function. These assumptions are requisite for ease of modelling and provide a reasonable approximation of observed trading patterns[26]. However, the nuance of irrigator decision-making factors including social constraints, institutional barriers to trade such as transaction costs and trading rules, are missed in these assumptions. For example, assuming rational profit maximizing behaviour and simplification of institutional complexity is necessary, considering both the epistemic uncertainty (e.g. incomplete knowledge of irrigator motivation) and the natural variability of human behaviour. The challenges in modelling water trade is consistent with wider literature, which only approximates the operation of water markets in a mathematical framework (Booker et al., 2005).

5.4.3 Groundwater

Limited datasets representing the complex and heterogeneous non-linear surface-water and groundwater connections and boundary conditions, result in an incomplete knowledge base about MDB groundwater resources. As a result, a number of underlying assumptions are required, specifically when considering the complex economics of groundwater management (Katic and Grafton, 2012). Due to these data limitations, groundwater modelling and management is an inherently uncertain science. These uncertainties and assumptions, specifically those pertaining to spatial aspects and temporal dynamics, are detailed comprehensively in Katic and Grafton (2012).

Generally speaking, agricultural sector models addressing climate change, water trading, water allocation and the *Basin Plan* give a limited account of agricultural groundwater use. Eleven out of the

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[26] Farmers' and institutional water holders (e.g. EWHs, bulk irrigation water etc.) are heterogeneous in their response to Basin Plan and reasons for water trading. Some of these reasons may not necessarily reflect profit maximising behaviour. For example, a farmer may wish to maintain 50% production of the farm each year, or choose only to sell a percentage of the water allocation available each year. In HEM, these considerations can be built into the model through scenario analyses. A similar approach could be used to test the impact of institutional barriers to trade, such as the 4% cap on entitlement sales out of Northern Victoria (removed 2014).

66 models reviewed incorporate groundwater resources. However, groundwater has been comprehensively addressed specifically by dedicated models e.g. Salt Water and Groundwater Management (SWAGMAN) Farm model, Salinity and Land-Use Simulation Analysis (SALSA). SWAGMAN Farm is a key example of groundwater modelling in the MDB and differs from the majority of hydro-economic models due to its focus on farm-level analysis, which can be used to either simulate salinity effects or optimise cropping choice. Similarly, the SALSA model incorporates relationships between land use, vegetation, surface and groundwater hydrology, and agricultural returns (Heaney and Beare, 2001), to provide guidance on improved salinity outcomes.

Although traditionally managed as separate resources, surface-water and groundwater resources interact across the MDB to varying degrees, based on the hydro-geological properties of the region (Parsons et al., 2008). Where the two resources interact, the use of one resource affects the quality and quantity of the other (Winter et al., 1998). For example, in modified basins irrigation and chemical application along with changes in the runoff and infiltration characteristics of the catchment result in a change in quality and volume of groundwater recharge. If hydrologically connected, this in turn influences surface-water quality and availability. When groundwater and surface-water interactions are not adequately modelled, this can lead to failure to anticipate perverse impacts, which may arise from exploitation of one resource without due consideration for the other. Groundwater use has been increasing steadily in Australia because irrigators often use groundwater in dry periods to augment surface-water supplies (NWC, 2011), or as substitution when farmers sell entitlements to the government (Haensch et al., 2016). Concerns over changes to the hydrological regime from increased afforestation and the risk of double-counting (Young and McColl, 2009), as well as changes in localised recharge due to climate change (Barron et al., 2011), further highlight the need for greater inclusion of groundwater resources in HEM.

5.4.4 Climate Change

In the MDB, the impact of climate change will be spatially and temporally diverse, with water availability more likely to decrease than to increase (Leblanc et al., 2012), causing the increased probability of dry states of nature (Adamson et al., 2009). More than half of the studies reviewed provide a discussion of climate change and 14 studies model climate change impacts directly.

The focus of climate change impact modelling has predominately been the impact of reduced water availability on irrigated agriculture, with results broadly indicating that dry to extreme climate scenarios result in reductions in available water, irrigated land and agricultural profits (Quiggin et al., 2010; Jiang and Grafton, 2012; Qureshi et al., 2013a; Qureshi et al., 2013b; Qureshi and Whitten, 2014). Climate change may impact agriculture through changes in water quality, input prices, growing and harvesting periods, and crop yields (Troost and Berger, 2014). Climate change impacts in the MDB have also been modelled using the state-contingent approach (detailed fully in Quiggin and Chambers (2006) with

regard to modelling under uncertainty), by modelling 'dry' and 'wet' states of nature to represent potential future inflow scenarios. The state-contingent approach has been employed to model irrigator behavioural response to various possible states of nature (Quiggin et al., 2010; Crean et al., 2013; Adamson and Loch, 2014; Loch and Adamson, 2015). Using this approach, Adamson and Loch (2014) argue modelling that does not account for the role of this uncertainty may misrepresent the benefits of public investment to return water to the environment. In the MDB, the impact of reduced water quality as a result of climate change has been sparsely addressed, with the exception of Beare and Heaney (2002), Adamson et al. (2009) and Connor et al. (2012).

Employing a whole-of-basin hydrological simulation, Kirby et al. (2014a) demonstrate that the estimated river flows and diversions in the MDB are more sensitive to climate change projections than to various *Basin Plan* reallocation scenarios. Further, the enhancement of flows due to reallocation is beneficial only under scenarios of no climate change or wet climate change scenarios. The proposed *Basin Plan* did not directly account for climate change impacts (Pittock et al., 2015). Historically, the naturally high variability of basin inflows were not fully considered during the development phases of the MDB (Leblanc et al., 2012). Neglecting future hydrological variation and stochastic uncertainty is therefore a risk of repeating past omissions. When modelling climate change, improving the certainty of impacts, or assessing and modelling which impacts may be considered highly uncertain, will assist with planning and implementation of adaptation measures and contingency plans (GCCR, 2008).

5.5 Environmental Outcome Focused Modelling

Table 5.3 summarises environmentally focused MDB hydro-economic models. Comparing Table 5.3 with Table 5.1 and Table 5.2 in the previous section it is clear that MDB HEM has predominately focused on agriculture, although there is evidence of a growing emphasis on the ecological and economic benefits of environmental water reallocation.

Generally speaking, there are two methodological approaches to modelling change in water resource systems: hydro-economic modelling and eco-hydrological modelling. Eco-hydrology is an interdisciplinary field concerned with analysing the interactions between water and ecosystems, and particularly focuses on feedbacks between hydrology and ecology. As previously outlined, hydro-economics is the field concerned with linking hydrological drivers with economic concepts of supply and demand. The choice between these two approaches depends on the objectives of the modeller and the data available. As previously described, hydro-economic modelling involves the integration of economic concepts into hydrological analysis of water resources. Conversely, eco-hydrological analysis involves simulating ecological responses to hydrological changes.

Overall, HEM in the MDB has given relatively small attention to the inclusion of representative ecological functions when modelling the environment. Similarly, eco-hydrological modelling has rarely incorporated the economic costs or benefits derived from changes in the ecosystem. The combination

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of hydro-economics and eco-hydrology can provide substantial methodological and policy insights with regards to water resource decision support. One area in which these two approaches overlap is the field of ecosystem service (ES) modelling. The overlap of hydro-economic and eco-hydrological disciplines is shown in Figure 5.2. ES modelling is the domain in which biophysical modelling and economic valuation combine to provide scientifically defensible estimates of benefits generated by the environment (Barbier et al., 2009). As such, the combination of economic valuation and biophysical modelling of flow-dependent ecosystem services is a powerful way to evaluate the costs and benefits of water reallocation, and inform the use water allocation in a way which would most benefit society.

Figure 5.2 The Intersection of Hydro-economic Modelling and Ecological Response Modelling
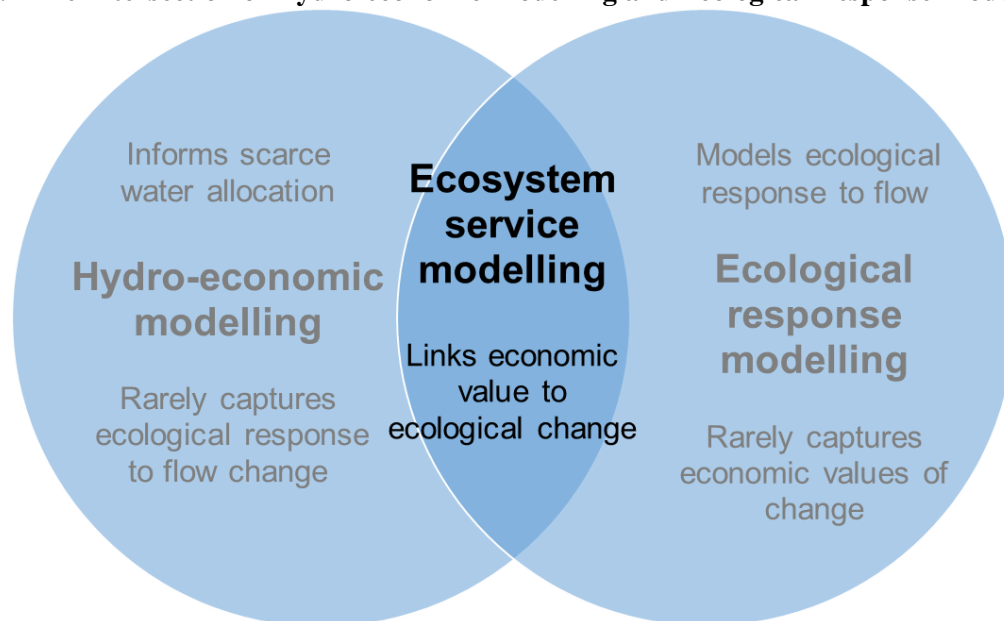
Source: Own Figure.

Although HEM is the dominant modelling approach in the MDB, eco-hydrological modelling is growing in prevalence and sophistication (e.g. Shenton et al., 2012; Szemis et al., 2014; Sawada et al., 2014). However, the combination of process-based ecological responses (eco-hydrology) with economic valuations of environmental and farming values (hydro-economics) is comparatively rarer. This is a significant gap in HEM for the MDB. To further explore this gap, the following sections review the current state of ecological and ecosystem service modelling in the MDB.

Table 5.3: Environmentally Focused Hydro-economic Models in the MDB

FOCUS	ISSUE	MODEL	AUTHOR AND DATE	MODEL TYPE	DESCRIPTION	OBJECTIVE	SACLE
Environment	Environmental water acquisition	ARIS	Kirby et al. (2006)	Hydrological network simulation	Examines potential environmental and economic costs and benefits of counter-cyclical trading and banking	N/A	Generic
		-	Ancev (2014)		Estimates potential environmental benefits of annual environmental allocation trade	Maximise aggregate returns	Basin wide
	Economic valuation	-	CSIRO (2012)		Quantifies ecological and economic benefits of an additional 2800GL environmental water	N/A	MDB
		Hydro model: IQQM; DSS: IBIS	Akter et al. (2014)	DSS	Links stated preference valuation to hydrology model to estimate marginal value of environmental water	N/A	Macquarie Marshes
	IWRM	-	van Emmerik et al. (2014)	Coupled non-linear ODE	Simulates interaction between population dynamics, resource availability, and ecosystem health using ODE	N/A	Murrumbidgee Catchment
	Water allocation	-	Grafton et al. (2011b)	Dynamic optimisation	Optimal water allocation model between irrigated agriculture and consumptive use	Maximise net social returns	Murray River
	Environmental water acquisition	-	Connor et al. (2013)		Estimates the potential environmental benefits of annual allocation trade in environmental water	Minimise environmental damage	Murrumbidgee Catchment
	Environmental water management	Ecological model: MFAT	Szemis et al. (2013); Szemis et al. (2014)	Evolutionary ant colony optimization	Develops optimal trade-off options between ecological response, and environmental water allocation for upstream constraints and infrastructure conditions (Szemis et al., 2013) and operations schedule changes (Szemis et al., 2014)	Maximise ecological benefit, minimise allocation; minimise schedule description	Murray River
	Water trade	-	Tisdell (2001)	Multi-objective optimization	Investigates impacts of water trade on flow regime, relative to natural flows and economic impacts on irrigated agriculture	Minimise difference between flow regimes, maximise agricultural returns.	Border Rivers Catchment

Source: Own Table. References cited within.

5.5.1 Ecological Response Modelling

Ecological response models are used to evaluate how an ecosystem will respond to changes in environmental or policy conditions. This approach has been integrated into HEMs to assess the impact of expanded environmental water trading (Connor et al., 2013; Ancev, 2014) and optimal water allocation and management (Grafton et al., 2011b; Szemis et al., 2013; Szemis et al., 2014).

In the MDB experience, conceptual ecological response functions, e.g. hypothesised response to drought (Grafton et al. (2011b)), have been used to overcome the high data requirements and epistemic uncertainties, resulting from a dearth of ecological time-series data and the complexity of ecological responses (e.g. Grafton et al., 2011b; Connor et al., 2013; Ancev, 2014). For example, to investigate the ecological benefits of expanding environmental water trade, Connor et al. (2013) developed a stylised ecological response function, to represent exponentially increasing ecological damage as time exceeds the recommended floodplain inundation frequency. Modelling results indicate the introduction of active EWH annual trading can increase frequency of moderately large floods (>1,700GL/year) to reduce ecological damage.

A similar approach is adopted by Grafton et al. (2011b) who uses a constructed drought indicator to measure ecological damage. Using this approach, it is found that a pulse flow is a useful means to deliver flows to achieve environmental benefit and, furthermore, water markets can reduce the costs associated with reallocating water to the environment by allowing dynamic trade between users (Grafton et al., 2011b).

In a multi-objective optimization approach, Ancev (2014) integrates the economic value of environmental water into a benefit function, by summing the benefit from productive use and ecological response. Economic value is derived by matching approximate expected environmental conditions for a range of environmental water availability with willingness-to-pay values for improved environmental conditions. Because environmental conditions are approximated on water availability, the integration of the hydrological and biophysical is coarse in this case and there appears to be no direct linkage between ecological change and economic value. Overall, results presented in Grafton et al. (2011b), Connor et al. (2013) and Ancev (2014) are consistent with wider non-environmental focused literature (e.g. Kirby et al., 2006; Brennan, 2006), which concludes that net benefits to society are greater when the CEWO actively participates in the water market by allowing water to be secured when it is most needed by the environment.

Further to conceptually based representation of the environment, evaluating the trade-offs between management options can be more accurately informed when the composite environmental benefit function is articulated. An example of this approach is Szemis et al. (2013), who uses the Murray Flow Assessment Tool (MFAT) to construct a weighted ecological response function of environmental assets and species across multiple years, to inform dam manipulation for environmental flow releases. In this

case, the function was employed as an objective within a multi-objective hydro-economic model, designed to maximise environmental benefit while minimizing releases to the environment (Szemis et al., 2013); and minimising disruption to management schedules (Szemis et al., 2014). This highlights the complicated trade-offs of management decisions. The MFAT model has also been used more widely in the MDB and is the primary decision support tool used in modelling ecological responses to hydrological change. The tool models changes in habitat condition for a number of bird and fish species through the specification of preference scores, for a variety of hydrological events at different points in the lifecycle (Young et al., 2003). For example, a preference curve may specify Murray Cod (*Maccullochella peelii*) preference for overbank flooding in juvenile life period to provide nutrients for growth. Scores are awarded from zero (unsuitable flow regime) to one (ideal) to provide an indication of the predicted habitat condition for any specified flow pattern (InsideMFAT, 2004). MFAT also models ecological responses of floodplain vegetation, wetland vegetation and algal growth (Young et al., 2003).

One limitation of ecological response modelling is that, as with groundwater modelling, ecological response modelling suffers from significant epistemic uncertainty resulting from deficiencies in data describing boundary conditions, forcing processes and system response (Beven, 2015). Ecological data is observed incompletely, resulting in an unknown amount of unknowns in captured and missed data, leading to an incomplete knowledge of the system (Cressie et al., 2009). Further, within the MFAT model, habitat condition is used as a proxy for species populations, which implies linearity between population abundance and the habitat suitability index. This is done because, in general, direct quantitative modelling of population dynamics is a complex and computationally prohibitive task. Debate regarding the reliability of habitat suitability scores as a proxy for species population is ongoing in the ecological literature (Roloff and Kernoham, 1999; Favata et al., 2015), but will not be elaborated here.

5.5.2 Ecosystem Service Modelling

A subsequent step in environmental integration of HEM is the economic valuation of changes in ecological state, with respect to changes in the hydrological regime. In the MDB, considerable literature has been developed in establishing a baseline of MDB ecosystem services (CSIRO, 2012; Banerjee et al., 2013), particularly in the Coorong, Lakes Alexandrina and Albert Ramsar Sites (Colloff et al., 2015).

The state of the art in ecosystem service modelling in the MDB is CSIRO (2012) and Bark et al. (2016)[27], which provide the first whole-of-MDB estimate of ecosystem service supply value. These examples extend the scope of the literature by considering a superior range of ecosystem service

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[27] Bark et al. (2016) is a summary of results from an integrated ecosystem service assessment undertaken by CSIRO (2012) to understand the ecological and economic impact of the *Basin Plan.*

indicators, their spatial partitioning, interaction, and importance for policy and incentivising strategic management (Banerjee and Bark, 2013). In these studies, the value of the supply of ecosystem services is modelled under a 'without development', 'baseline' and *Basin Plan* '2800GL scenario' based on expected ecological and biophysical responses, to estimate the improvement in ES supply values achieved through improved environmental water availability. It was estimated that a total value would be in the order of billions, derived predominately from habitat services (between $3 and $8 billion), aesthetic value (>$330 million) as well as improvements in carbon sequestration (between $120 million and $1 billion) (CSIRO, 2012).

A notable modelling study of ES values is by Akter et al. (2014) who integrate hydrological modelling with ecological responses and willingness-to-pay estimates, to evaluate the economic benefits of increased environmental water availability. The simulation is run for three flow scenarios, 'baseline', '104GL increase' and '135GL increase' (Akter et al., 2014). The model is applied to the Macquarie-Marshes and uses WTP estimates for bird habitat (Morrison et al., 1999) and native vegetation (Whitten and Bennett, 2000). A benefit of $0.15million and $1.7million was estimated for an increase of 104GL and 135GL river flow, respectively (Akter et al., 2014). The highly integrated model demonstrates the insight that can be gained through combining economic valuation with ecological modelling, to inform strategic reallocation between competing uses.

The method applied by CSIRO (2012); Akter (2014) and Bark et al. (2016) is essentially a static comparison model as it runs the simulation for each scenario and then compares the result. One limitation of this approach is that it does not capture the dynamic nature of ES provision and the annually variable costs and benefits. As such, static comparison models are not fully applicable to evaluating the inter-annual trade-offs between consumptive and environmental water-use on an annual basis, or for use in aiding the understanding of the marginal benefit of temporally variable increases (e.g. annual reallocation) of environmental water. The limits in methodological approach is are partially reflected in the stakeholder survey from the CSIRO (2012) ES study. The survey revealed a stakeholder perception of reduced creditability due to the lack of consideration of alternative flow scenarios or potential to achieve better environmental outcomes through alternative management of environmental water (Hatton-MacDonald et al., 2014).

An additional issue faced by ES modellers is the difficulty regarding the validity of economic valuation techniques applied. For example, there exist a number of criticisms regarding stated preference methodologies (Hausman, 2012), which have been widely used in the MDB case (Morrison and Hatton-MacDonald, 2010). Additional challenges arise from the fact that attributes included in stated preference surveys may not be easily quantified in ES models. An example of this is 'biodiversity' as an attribute in stated preference experiments, which can only be reasonably modelled with indicators (e.g. species population) in ES models. However, there is now a large and growing (see Morrison and

Hatton-MacDonnald, 2010 for a comprehensive review) volume of valuation studies from a number of environmental assets in the MDB, which allow the careful selection of studies (as is required for benefit transfer, for example) to be cautiously undertaken to minimise valuation concerns. Additional discussion of the economic valuation methodologies applicable to ES modelling, as well as the potential for bias in ES modelling, are outlined in Section 6.1.

5.6 Summary of Findings of MDB Modelling

It is clear that MDB HEM has been primarily focused on agricultural production. Assessing *Basin Plan* impacts has been approached using a number of methods, resulting in differing but comparable estimates of irrigation income forgone with the introduction of the SDLs. Modelling results indicate that reduction of irrigation water availability, following the introduction of the SDLs, is likely to result in less-than-proportional reduction in regional farm profit, and a minor impact on the overall national economy. Modelling further indicates that targeted reallocation can reduce the opportunity cost of water reallocation. The prevalence of MDB water trade has been mirrored in modelling efforts, and consensus has been built regarding the positive economic benefits of water trade in reducing the cost of adapting to decreased water availability, caused by climate change or the reallocation of entitlements to the environment. The reduction in farming revenue can be reduced with irrigator adaptation (e.g. transition to dry land farming) and the availability of inter-regional trade. Regulations limiting inter-regional trade have significant annual costs. Localised externalities from trade are difficult to account for and have not been wholly modelled, potentially affecting the magnitude of estimated water re-allocation and trading benefits. Model results indicate water traded out of regions may cause a reduction in resource quality and agricultural production in high salinity impact areas if minimum irrigation intensities cannot be met. Climate change has been widely discussed, through less commonly modelled, and the majority of effort has been focused on addressing water supply variability. Model results indicate climate change will likely negatively affect irrigation revenue, by both reduced water availability and increased salinity impacts. Realised climate change impacts are likely to also affect the trajectory of MDB water policy, as flows in the basin are more sensitive to climate change than reallocation strategies as part of the *Basin Plan*, thus requiring a more robust incorporation of climate change risks into future policy design.

The environmentally focused modelling demonstrates that additional environmental water availability generated under the *Basin Plan* is likely to have considerable ecological and economic benefits through an increase in ES functions. Improvements in ES supply are expected to be predominately generated by improvements in non-market values (e.g. habitat and biodiversity) as well as carbon sequestration. At a state based level, it is shown that optimised infrastructure operations and environmental water trade can improve ecosystem health and locally water availability, which in turn can measurably increase native bird and fish habitat. Environmental water trading models indicate that counter-cyclical trading involving the sale and purchase of water allocations for the environment can be implemented at no net cost (except administration) and can have a number of hydrological and environmental

outcomes. In hydrologically connected floodplains, trading allocations for the environment can reduce the rate of ecological degradation and reduce drought costs (conversely, improve ecological benefits) by improving the frequency of moderately large floods to the floodplain. Large flood events are mostly untouched by annual allocation trading.

5.7 The Gap in MDB Hydro-economic Modelling

Modelling the supply of flow-dependent ES requires the integration of biophysical models of ecosystem responses with economic estimates of ES values. In the MDB to date, HEMs that incorporate some form of ecological representation do so through the use of conceptual representations rather than process-based ecological models (e.g. Kirby et al., 2006; Grafton et al., 2011b; Connor et al., 2013; Ancev, 2014). This is largely because of the complexity of ecological responses which drive ES service supply and a dearth of ecological time-series data available for HEM modelling, although ES data is growing in standard and breadth (CSIRO, 2012). Put simply: economists undertaking hydro-economic modelling are not ecologists and *vice versa*. This effectively means that the first two aspects of the ecosystem services cascade[28] ("biophysical structure or process" and "ecosystem function") (Haines-Young and Potschin, 2010) are simplified to conceptual functions, which do not necessarily fully capture ecosystem dynamics. Conceptual response functions also do not represent the reality that within a management area there are generally multiple and sometimes competing assets (e.g. river reaches, wetlands and floodplains) containing multiple indicators of ecosystem health. This issue is heightened in cases where ES benefits are assumed to be linearly proportional to volumetric increase in environmental water, as is frequently assumed in the policy debate (Crase et al., 2011). A discussion on the implications of adopting volume as a metric of success for reallocation programs is presented in Section 3.10.7.

However, there are a number of models which do include a more sophisticated representation of ecological change (e.g. CSIRO, 2012; Akter et al., 2014; Bark et al., 2016). This group of literature models the impact of permanent reallocation of water on the supply of ES. As previously mentioned, these approaches are in essence static comparison models which use a range of scenarios to estimate ES changes. These models therefore do not give an indication of ecological response or ES change with regards to annually dynamic anthropogenic actions such as annual environmental water trading. To reiterate, models which do investigate the impact of annual trading do so using conceptual response functions and do not provide estimates of ES or their values. As a result, there are currently no MDB HEMs which adequately address the temporal dynamics of ecological response, ES supply and economic value, in response to variable changes in environmental water availability (e.g. caused by allocation trading for the environment) on an annual time step.

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[28] A full explanation of the ecosystem service cascade and related concepts are presented in Section 6.1.

This review and the subsequent findings confirms results from existing international literature reviews, which identify ecological modelling and simulation methodologies as a deficiency in current ES valuation approaches (Seppelt, 2011). This review also confirms recommendations by Fisher et al. (2008) and Haines-Young and Potschin et al. (2012) who argue that there remains a pressing challenge for models to estimate the marginal changes in ES resulting from anthropogenic actions, which is crucial in making informed economic decisions about trade-offs and policy alternatives (Fisher et al., 2008).

Overall, models rooted in an understanding of the dynamics of the biophysical system are best placed to address complex issues on non-linearity, path-dependency, ecological thresholds, irreversibility and marginality of ecosystem responses which produce ESs and economic value. By better accounting for the biophysical dynamics of ES supply, a more robust picture of environmental benefits can obtained and used to inform strategic and cost-effective annual water reallocation decisions.

To overcome this gap and generate models which are best placed to inform environmental watering policy, HEM in the MDB faces significant challenges regarding the integration of hydrological, biophysical and economic components. To demonstrate the level of integration required to link these components a conceptual model is shown in Figure 5.3, which demonstrates the components of a HEM of ES supply developed for informing trade-offs, including the required links between hydrology, ecology and economics. Figure 5.3 also provides examples of the assumptions required in ES modelling, such as proportionality between methods of valuation (e.g. marginal WTP estimates of population increase) and ecological indicators (e.g. habitat suitability indices).

Figure 5.3 Conceptual Representation of the Model Linkages between Hydrology, Ecology and Economics in Ecosystem Service HEM
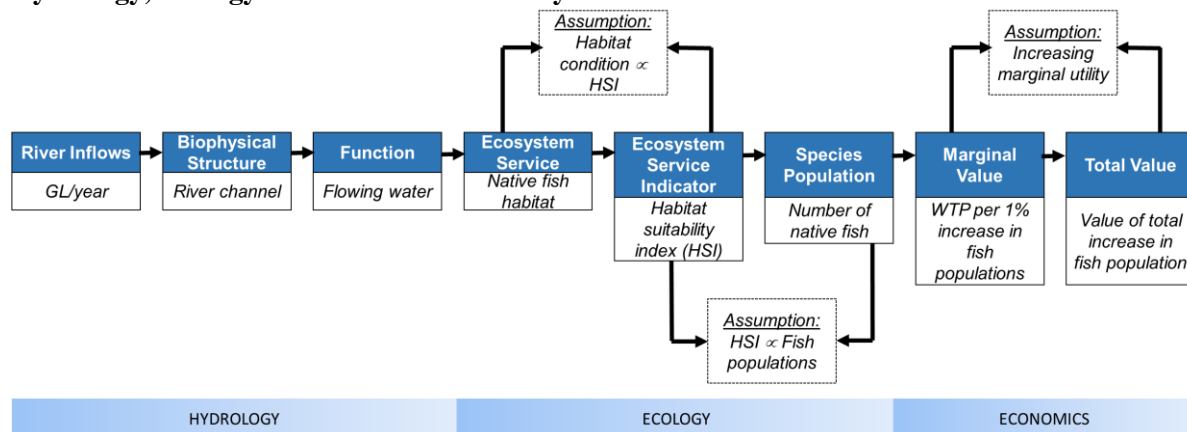
Source: Own Figure.

5.8 Conclusion

A considerable amount of HEM literature has been written to query and inform water policy in the MDB. This chapter has presented a comprehensive review of over 60 of these HEMs in the MDB. It is evident that water reallocation and *Basin Plan* modelling has focused disproportionately on the cost of reallocation to irrigated agriculture, while only a small amount of literature has addressed the ecological and economic benefits of reallocating water to the environment. There is therefore a pressing need to construct environmentally focused models, which are able to account for the other side of the balance sheet and effectively evaluate and inform water reallocation policy. Within the environmentally focused models, this review has also highlighted that existing literature does not include adequate representation of the biophysical processes that drive ES supply dynamics and, in particular, provides limited representation of inter-temporal linkages and non-linearity for processes other than those pertaining to irrigation. This is a considerable gap in the MDB HEM literature, the closing of which could provide a number of methodological and policy insights. This gap is partially addressed through the construction of an integrated HEM applied to the Murrumbidgee catchment, comprehensively detailed in Chapter 6.

CHAPTER 6: Theory, Data and Methodology of Hydro-economic Modelling of Carbon Storage Dynamics

This chapter presents the background, theory, methodology and data used to develop a hydro-economic simulation model. The model is used to simulate the annual water reallocation decisions of an environmental water holder (EWH) who acts to improve floodplain ecosystem health by annually reallocating water to the environment when it is economically and ecologically justifiable to do so. The model is coded in the General Algebraic Modelling System (GAMS) and consists of four integrated sub-models: a) the hydrology model; b) the water allocation price model; c) the carbon dynamics model including biophysical modelling and economic valuation; and d) the reallocation decision algorithm. Methodologically the model is a dynamic simulation model as it is computationally prohibitive to optimise an integrated model such as the one described in this chapter. The data and methodology sections of this chapter is extracted from the current working paper *"Water and carbon market opportunities for environmental water re-allocation to improve floodplain forest carbon storage"* which is split between Chapter 6 and 7 in this thesis to present in detail the methodology and results, respectively.

Firstly, an introduction to ecosystem services (ES), focusing on the classification flow-dependent ES and their role in water allocation decision-making. In particular, the carbon storage and sequestration ES generated by floodplain River Red Gums is introduced as the primary focus of the modelling. This section provides a logical defence for the focus on carbon storage and sequestration as of River Red Gums as an indicator of ecosystem health and a marketable ES commodity. This chapter also outlines the carbon sequestration valuation and provides a brief discussion on valuation methodologies and potential bias in ecosystem service valuations. A detailed description of the Murrumbidgee case study to which the model is calibrated and applied, including a description of the existing water market and potential carbon market that is in operation in Australia. Lastly, a comprehensive description of the hydro-economic GAMS model is presented and explained, including a description of all model data sets and assumptions.

6.1 Introduction

In many river basins around the world water is fully allocated for consumptive purposes and flow-dependent ecosystems are subsequently degraded. In particular, wetlands, including floodplains, swamps and marshes, are subject to long-term, widespread and ongoing degradation (Davidson, 2014). Inland wetlands generate a disproportionately high level of biodiversity and benefits to human wellbeing relative to their global extent (Russi et al., 2013). As the health of water dependent ecosystems declines due to natural or anthropogenic factors, so do the environmental benefits provided by these ecosystems (Banerjee et al., 2013).

In a growing number of cases water markets are being used to facilitate the compensated reallocation of water from consumptive to environmental users to prevent continued degradation (Wheeler et al., 2013b). However, identifying reallocation strategies that maximise environmental value and minimise water cost expenditure without large costs to the irrigation community is a considerable challenge for EWHs. In this study, it is hypothesised that the actions of an EWH who has the ability to annually reallocate water to the environment through trade can sufficiently influence the overbank flooding regime in order to promote improved carbon uptake or prevent carbon decay on a vegetated floodplain.

The MDB is relatively unique as it is one of the few areas where there are functioning markets for both water and carbon. The existence of these dual markets allow the proposition to be tested: could it be possible to generate carbon credits of sufficient value through strategic floodplain inundation in a way which can offset the cost of environmental water purchases required to cause the inundation?

A number of previous hydro-economic modelling (HEM)[29] studies have sought to integrate catchment hydrology, ecological change and economic impacts of water reallocation strategies, as described in the literature review in Chapter 5. However, no previous studies assessing the joint water and carbon market opportunities to enhance environmental flows have been identified. Further, as highlighted in Chapter 5 while there are a number of MDB HEM studies that have evaluated environmental flow strategies, very few have included adequate representation of floodplain forest carbon storage ecosystem service dynamics. More generally, relevant HEM studies to date have limited representation of inter-temporal linkages, biophysical thresholds, non-linear responses, and path-dependencies for process other than system hydrology and crop production. Similarly, hydro-ecological models with high levels of ecological process detail rarely account for the economics of environmental improvements and opportunity costs of irrigation water use. The result is a significant gap in the literature and a limited basis with which to quantify ecosystem service values and evaluate water allocation trade-offs.[30]

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[29] HEM is used to abbreviate both hydro-economic modelling and hydro-economic models.
[30] A literature review of existing studies, methodological and subject gaps, and possible avenues for advancement was discussed at length in Chapter 5.

To address this gap, a highly integrated HEM is developed incorporating sub-models representing: a) catchment hydrology; b) the water allocation price model; c) the carbon dynamics model including biophysical modelling and economic valuation; and d) the reallocation decision algorithm. This chapter details the development of the HEM and provides a discussion of the theory and process of integrating carbon storage dynamics into a HEM framework.

6.2 Ecosystem Services

The ecosystem service (ES) framework is used in the HEM to guide the identification and quantification of the benefits generated when water is reallocated to the environment. Ecosystem services are the direct and in-direct benefits to human survival and well-being that are derived from the natural environment (Daily, 1997; Costanza et al., 1997; MEA, 2005; TEEB, 2010; de Groot et al., 2012; Costanza et al., 2014).

Ecosystem services are derived from a number of overlapping structures and processes within the environment. It is therefore pertinent to employ a systematic method of classifying types of ES. For the purpose of this thesis, the Economics of Ecosystem Services and Biodiversity (TEEB) framework is employed. The TEEB framework classifies ecosystem services into four categories: a) provisioning services which are derived from ecosystem material or energy outputs; b) regulating services which are provided by the ecosystem acting as a regulator; c) habitat services provided to plant or animal species to survive; and d) cultural services derived from the direct or indirect cultural use of an ecosystem (TEEB, 2010). The TEEB framework is chosen for the advantages of simplicity of application, comprehensiveness of ES categories and the incorporation of 'supporting services' into other first-order categories to help to avoid double counting. The TEEB categorization of ES is shown in Table 6.1. Discussion of other commonly used frameworks of ES classification and their relative merits and differences are provided in Appendix C.

Table 6.1 The Economic of Ecosystems and Biodiversity (TEEB) Ecosystem Service Categories

Ecosystem Service	Example
Provisioning Services	
Food	Agricultural food production; fresh water; managed and wild fisheries; hunting and game; wild food collection.
Raw materials	Wood for construction, reeds for weaving and baskets, biofuels, plant oils.
Fresh water	Potable water for human and animal consumption, irrigation water for agriculture.
Medicinal resources	Traditional medicines and raw materials for pharmaceutical production.
Regulating Services	
Local climate and air quality	Regulation of local climate through shade from trees, local rainfall influence, air purification.
Carbon sequestration and storage	Regulation of global climate by long-term storage of greenhouse gasses, terrestrial and marine carbon sinks.
Moderation of extreme weather events	Moderation of storms, tsunamis, avalanches and landslides by creating buffers between natural disasters and human settlements.
Waste-water treatment	Purification of natural and anthropogenic pollutants by biological activity of microorganisms in soil and natural water purification systems (e.g. wetlands).
Erosion prevention and maintenance of soil fertility	Vegetation cover preventing the mobilisation of soil and sediments, trees maintaining soil structure for natural and agricultural ecosystems.
Pollination	Insect and animal pollination of natural vegetation and food-crops.
Biological control	Regulation of pests and diseases through predators and parasites.
Habitat Services	
Habitat for species	Habitat for the survival of animal species including food, water and shelter during all stages of life-cycle.
Maintenance of genetic diversity (biodiversity)	Genetic diversity of genes within and between animal and plant species contributing to species health, development and adaptation capacity.
Cultural Services	
Recreation, mental health and physical health	Providing green space for physical activity, recreation and mental relaxation.
Tourism	Attracting visitors to engage in nature-based activities.
Aesthetic appreciation, cultural inspiration, art and design	Providing inspiration for artistic representation of the environment, aesthetics for human activities and settlements and intrinsic cultural appreciation.
Spiritual experience and sense of place	Sacred, spiritual or religious places or aspects of nature, continuation of traditional knowledge, nature-based customs or rites, continuation to a sense of belonging.

Source: TEEB (2010). Available: http://www.teebweb.org/resources/ecosystem-services/, [Accessed August 4th, 2017]

6.2.1 The Ecosystem Service Cascade

The way in which ecosystem services are generated, perceived and valued is described by the ES cascade (Haines-Young and Potschin, 2010). The ES cascade was adapted by de Groot et al. (2010) to produce the TEEB diagram shown in Figure 6.1. The ES cascade demonstrates that biophysical structures or processes in natural ecosystems carry out functions that individuals and society perceive

as services. As shown in Figure 6.1, ecosystem services maintain or generate an increase in human well-being and therefore have a value that can be valued in economic terms. The generation of ecosystem services is influenced by the sum of natural and anthropogenic pressures acting upon the ecosystem and can also be influenced by human institutions, such as ecological restoration and management. The perception of ecosystem services are mediated by interaction with institutions and human judgements (Costanza et al., 2014).

Figure 6.1 The Ecosystem Service Cascade.
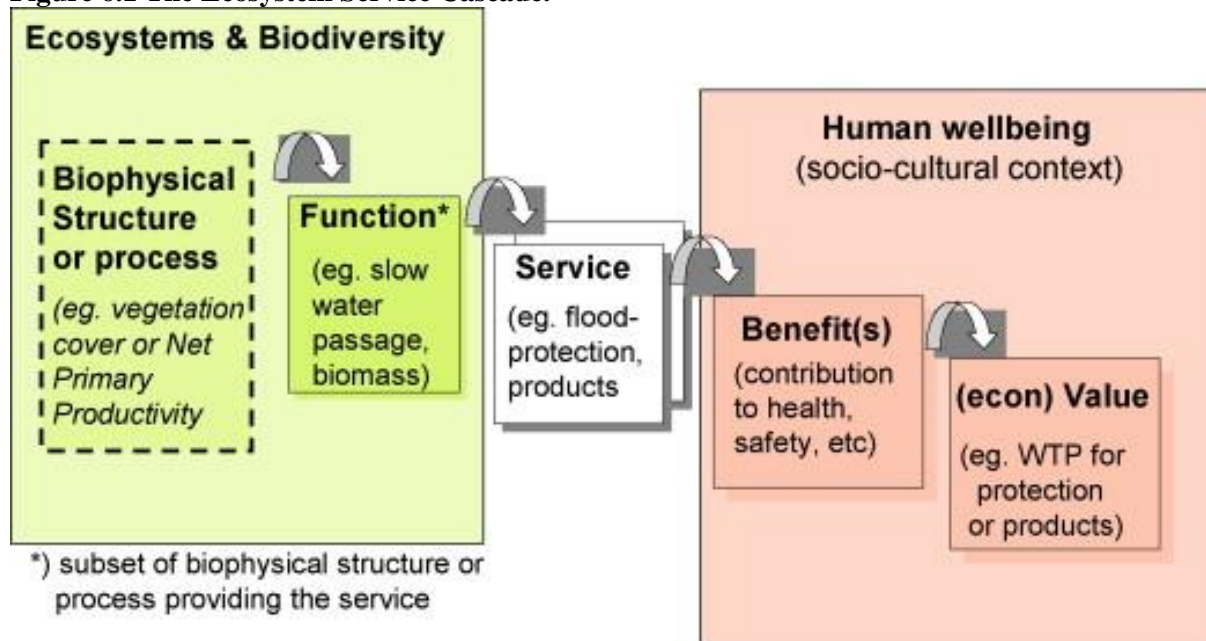
Source: de Groot et al. (2010), adapted from Haines-Young and Potschin (2010).

6.2.2   Flow Dependent Ecosystem Services

Flow dependent ES are services which are influenced by hydrological events and characteristics such quantity, quality, location, duration and timing flow (Brauman et al., 2007). Flow-dependent environments such as floodplains, wetlands and rivers generate a disproportionately high level of biodiversity and benefits to human wellbeing relative to their global extent (Costanza et al., 1997; Russi et al., 2013). Figure 6.2 demonstrates how flow-dependent ES are generated in freshwater environments.[31]

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[31] In Figure 6.2, hydrological services are an alternative name for flow-dependent ecosystem services (CSIRO, 2012). TEEB classifications for the hydrologic ecosystem services in Figure 6.2 are: Diverted water supply=freshwater; in situ water supply=habitat for species, recreation and mental and physical health, food (e.g. fish), waste water treatment; water damage mitigation=moderation of extreme events; spiritual and aesthetic=tourism, aesthetic appreciation and inspiration for art, culture and design, spiritual experience and sense of place; supporting=maintenance of soil fertility, local climate and air quality, maintenance of genetic diversity. Note that Figure 6.2 does not provide a comprehensive list of hydrologic ecosystem services and there is not complete overlap between hydrologic services and the TEEB classifications.

Figure 6.2 Flow Dependent Ecosystem Services

Ecohydrologic process (what the ecosystem does)	Hydrologic attribute (direct effect of the ecosystem)	Hydrologic service (what the beneficiary receives)
Local climate interactions Water use by plants	→	Quantity (surface and ground water storage and flow)	**Diverted water supply:** Water for municipal, agricultural, commercial, industrial, thermoelectric power generation uses
Environmental filtration Soil stabilization Chemical and biological additions/subtractions	→	Quality (pathogens, nutrients, salinity, sediment)	**In situ water supply:** Water for hydropower, recreation, transportation, supply of fish and other freshwater products
Soil development Ground surface modification Surface flow path alteration River bank development	→	Location (ground/surface, up/downstream, in/out of channel)	**Water damage mitigation:** Reduction of flood damage, dryland salinization, saltwater intrusion, sedimentation **Spiritual and aesthetic:** Provision of religious, educational, tourism values
Control of flow speed Short and long-term water storage Seasonality of water use	→	Timing (peak flows, base flows, velocity)	**Supporting:** Water and nutrients to support vital estuaries and other habitats, preservation of options

Source: Brauman et al. (2007), p. 73.

Analysis of flow dependent ecosystems services requires consideration of the eco-hydrological processes (e.g. *'what the ecosystem does'* in Figure 6.2) that drive the production of flow-dependent services. In freshwater ecosystems, river flow governs ecosystem health at a local and regional scale (Poff et al., 1997). In particular, flows of sufficient volume to generate overbank floods of variable volume and frequency is the primary determinant of the ecological health and resilience of rivers, wetlands and floodplains (Junk et al., 1989; Poff et al., 1997; Overton et al., 2009). The relationship between flow characteristics and the generation of flow-dependent ES generation is shown in Figure 6.3. Figure 6.3 is annotated with an example of the carbon storage and sequestration ES and is discussed below.

Figure 6.3 The Generation of Flow-dependent Ecosystem Services
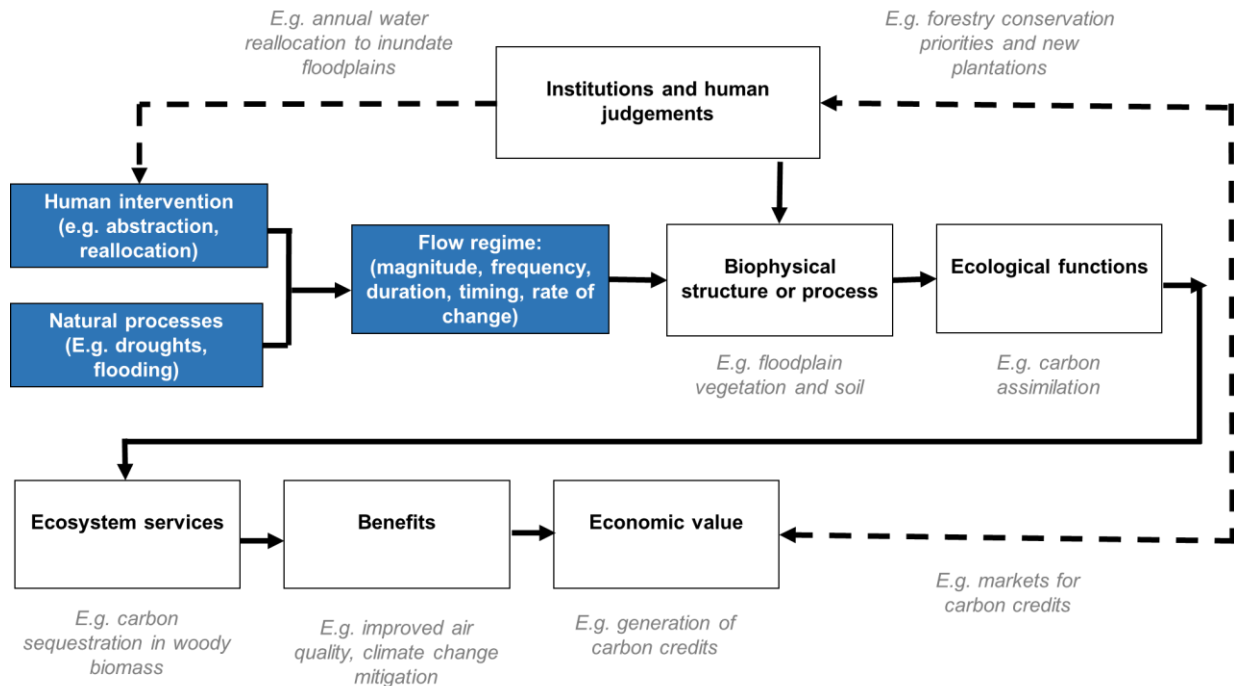
Source: **Own Figure. Adapted after Haines-Young and Potschin (2010).**

As shown in Figure 6.3 hydrologic characteristics impacting the generation of flow-dependent ecosystem services can be influenced by natural processes and human interventions. For example, ecosystem health in the MDB is contingent upon natural hydrological variability, including high flooding interspersed with periods of drying (Overton et al., 2009). The flow regime can also be altered by human intervention including water abstraction, river regulation and reallocation programs. Human intervention on the flow regime is influenced by institution and human judgements that can directly and indirectly impact the hydrological regime (e.g. laws and policies set water management priorities) and biophysical processes (e.g. deforestation and land clearing). However, the relationships between hydrology and ES generation in socio-ecological systems, such as shared water resources, are rarely linear (Haines-Young and Potschin, 2010; Levin et al., 2013) due to the existence of complex non-linear ecological responses, biophysical thresholds and ecosystem irreversibility, among others. Figure 6.3 is therefore a simplification of the relationship between hydrology and flow-dependent ecosystem services.

6.2.3   Informing Decision-making with Ecosystem Services Analysis

As the climate changes, populations grow and society places new demands on supporting ecosystems, there becomes an increasing need to be able to quantify the ecological and economic impacts of the changes that are occurring. Quantifying change in the environment has typically fallen within the realm of ecology, and as such the impacts of environmental change is often reported in the form of ecological metrics or eco-hydrological modelling outputs. However, ecological metrics and modelling outputs may

not necessarily be stakeholder relevant, easily accessible to decision makers, or useful in evaluating the cost-effectiveness of environmental projects. As a result, there is a risk that the full suite of environmental costs and benefits are unaccounted for in public and private decision-making.

ES analysis, which includes a typology for identifying benefits derived from the environment, provides a platform to link environmental change to economic outcomes in order to better assess trade-offs and inform decisions. Because the ES cascade is in essence an economic technique, the outputs expressed in monetary terms produces stakeholder relevant information which can be incorporated into decisions weighing the costs of ecological degradation against the benefits of development (Garrod and Willis, 1999). The ES approach therefore provides the needed bridge to communicate complicated environmental outcomes to decision-makers in a common language (Braat and de Groot, 2012). By doing so, the valuation of ES provides a means of mainstream the consideration of environmental values in natural resource decision-making (Farber et al., 2002; Cowling et al., 2007; Daily et al., 2009). Further, it is argued that quantitatively enumerating the value that the environment contributes to society also provides an effective method of debunking the *environment vs. economy* perception inherent in public policy debate (Costanza et al., 2012). Overall, ES analysis provides a means of making the environment visible and quantifiable in economic accounting, which therefore allows environmental impacts to be more robustly incorporated into mainstream decision-making.

6.2.4 Carbon Storage and Sequestration Ecosystem Service

The MDB environment produces an array of ecosystem services. These are comprehensively detailed in CSRIO (2012) and Bark et al. (2016). In this analysis, the single carbon storage and sequestration ES is modelled. A description of additional ES modelling is provided in Appendix A, which details model development for native fish habitat, blue green algae prevention and erosion prevention.

Terrestrial carbon storage and sequestration is the capture and storage of atmospheric carbon by vegetation and soils. Terrestrial forests act as carbon sinks due to their ability to uptake atmospheric carbon dioxide (CO_2) via the process of photosynthesis[32]. During photosynthesis, the carbon (C) from atmospheric CO_2 is converted into carbohydrates used to generate plant biomass growth, effectively trapping the C in the wood, foliate and roots of the plant. Long-lived vegetation sequesters C annually as the plant grows and can achieve the long-term storage of C in vegetation.

In addition, carbon taken up by plants can be stored in soils organic matter component via root transport into the soil or decomposition of vegetation into the soil, making soil-sinks a significant carbon storage component, especially in high above-ground carbon storage regions such as native forests (Smith and Reid, 2013). Floodplain and wetland ecosystems are particularly good at storing carbon because the soil is either submerged or periodically inundated by floodwaters which slows the rate of carbon

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[32] The overall balanced chemical equation for photosynthesis is: $6CO_2 + 6H_2O \rightarrow C_6H_{12}O_6 + 6O_2$

decomposition due the saturated wetland soils (NOAA, 2017). Figure 6.4 provides an overview of carbon pathways in terrestrial ecosystems. Terrestrial carbon sinks influence the global carbon cycle and are classified as regulating ES in the TEEB framework (TEEB, 2010). As a regulating service, carbon storage and sequestration offer the prospect of mitigating rising atmospheric CO_2 concentrations through reforestation and management of existing forests to promote carbon storage (Schlamadinger and Marland,1996).

Figure 6.4 Carbon Pathways in Vegetated Floodplains.
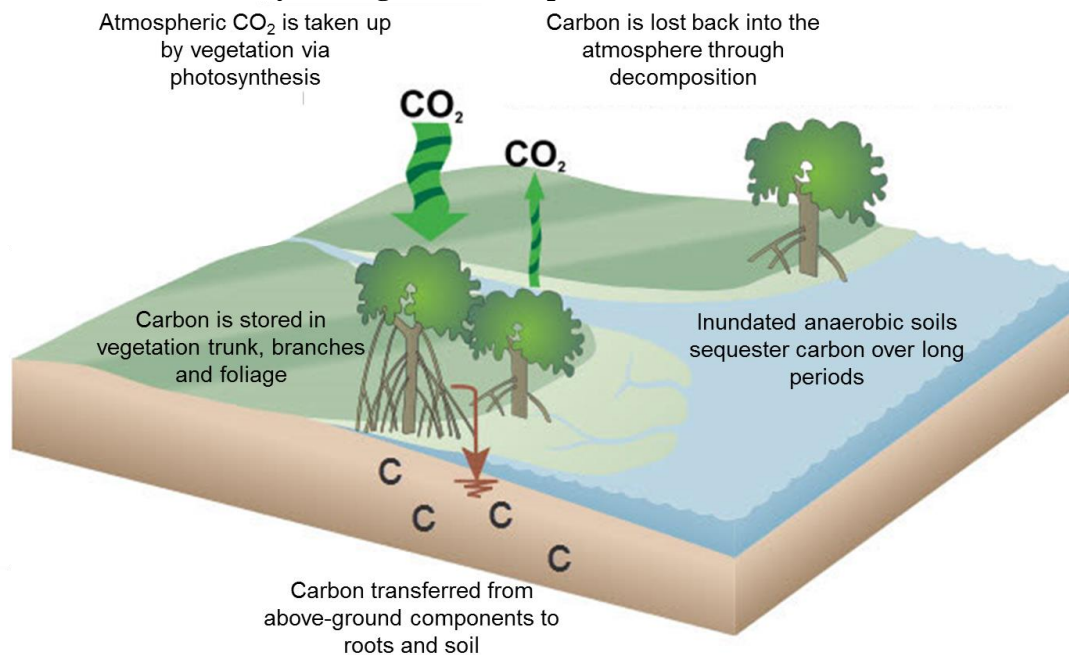
Source: Adapted after NOAA (2017). Available at:
http://www.habitat.noaa.gov/coastalcarbonsequestration.html [Accessed June 17, 2017].

Drought disturbance and water stress affects the condition of floodplain vegetation (Kingsford, 2003; Overton et al., 2009). Under drought conditions, River Red Gums reduce transpiration through stress-induced defoliation to conserve water and reduce heat load (Roberts and Marston, 2011). This behaviour was observed extensively during the Millennium Drought on the floodplains of south-eastern Australia (MDBC, 2003; Doody et al., 2015). The structural and biophysical changes in response to water scarcity have implications for carbon assimilation by reducing photosynthetic capacity and therefore reducing the rate of carbon uptake (Chaves, 1991; Gower, 2001; van der Molen et al., 2011). Under moderate water stress conditions the rate of photosynthetic CO_2 assimilation in tree leaves can decline to values close to zero due to a reduction in stomatal conductance (Chaves 1991; Faria et al., 1998; Lawlor and Cornic, 2002). The relationship between water deficit and declining carbon uptake has been observed in a number of case studies. For example, Gatti et al. (2014) observe a decrease in CO_2 sequestration was following a prolonged drought event in the Amazonian rainforest.

Similarly, there has been a hypothesised relationship between increases in water availability, tree health and the rate of carbon uptake. In a study of ES in the MDB generated by the *Basin Plan*, CSIRO (2012) find that the carbon sequestration is one of the largest contributors of ES improvements, generating a benefit in the order of $120 million to $1 billion (2012 dollars). The CSIRO (2012) report highlights two important findings. First: the rate of carbon storage and sequestration in forests can be improved through the increase of environmental flows (e.g. carbon volume stored can be modelled as a function of water availability); and second: there is considerable monetary value in increasing water available to floodplain forests. For these reasons, carbon storage and sequestration is selected as the ES to be included in the modelling.

In Australia, above-ground River Red Gums (*Eucalyptus camaldulensis*) store more carbon than other components of the ecosystem in native vegetated floodplains when present (Smith and Reid, 2013). River Red Gums therefore provide the best estimate of the volume of carbon stored on a floodplain without modelling additional functional groups. Consideration of additional carbon stores (e.g. soil, grasses) adds complexity but no more explanatory power to the model for the current purposes, and are therefore omitted. River Red Gum carbon storage is also selected due to the comparatively higher degree of data available due to monitoring efforts in response to the widespread decline of River Red Gums during the Millennium Drought (MDBC, 2003) and their icon status in the Australian environment. Further, due to their reliance on the delivery of specifically timed floods of variable volumes River Red Gums make an excellent indicator for measuring the success of environmental flow management (Catelotti et al., 2015). The condition of River Red Gums may also provide a reasonable indicator of overall ecosystem health (Catelotti et al., 2015), and hence ES generation, in the case study (detailed in Section 6.6.).

6.3 Economics of the HEM

This section discusses the economic foundations of the model, including the concept of value, the conceptual economic framework, method of ES valuation, and the valuation of River Red Gum carbon storage.

6.3.1   Conceptual Framework

It is economically desirable for scarce resources to be efficiently allocated among high value uses and time periods such that benefits are equalised across uses, or equivalently, the principle of equimarginality. For catchments within a basin this is typically modelled for a small water additions and subtractions from point $k = 1....K$ connected to flows. The net economic benefit of using water across points within a single period, t can be written $NB_k(x_k) = B_k(x_k) - C_k(x_x S)$, where $B_k(x_k)$ is the benefit of water-use and $C_k(x_k, S)$ is the cost of water use, subject to water supply S.

Because of the dynamic nature of the environmental watering and irrigation planning there are temporal trade-offs of using water. These are typically represented with discrete temporal variables for a horizon of $t = 1...T$ years. In such models, the period and point-specific net benefit of water-uses can be represented as $NB_{k,t}(x_{k,t}, S_t) = B_{(k,t)} - C_{k,t}(x_{k,t}, S_t)$ where the flows of net benefits generated in future periods are appropriately discounted at rate, r. An efficient allocation of water $x_t^* = (x_{1t}^*....x_{kt}^*)$ for each period t maximises the present value of the net benefits over the decision horizon:

$$\max \sum_{t=1}^{T} \left(\frac{1}{1+r} \sum_{k=1}^{K} B_{k,t}(x_{k,t}, S) \right) \qquad \textbf{(Equation 6.3)}$$

For HEM practice the maximization equation is subject to a range of period- and location-specific constraints. For example, in models with temporal linkages water-use in period one may affect water-use in subsequent periods by changing the water availability. This is partially the case in groundwater depletion, where water extracted in one year may not necessarily be replenished quickly enough by natural processes to ensure the same extraction rate in the following year. In the model described within this thesis, there are modelled temporal linkages between the environmental water demand of the River Red Gums and the subsequent environmental benefits which are influenced by the pattern of water delivery and hydrological conditions. For example, failure to meet the environmental watering delivery targets for an extended period has implications for location specific rates of carbon stock recovery on the floodplain.

If in Equation 6.3 $T \rightarrow \infty$ this then becomes the optimal control problem. Optimal control theory of dynamic optimization is used to identify control policies for dynamic systems and is often applied to fisheries and forestry harvest management (Sethi, 1973; Chen and Ahmed, 1982; Cohen, 1987). However, tractability in optimal control problems is a challenge, especially in natural ecological systems which are inherently complex, non-linear and incompletely observed (and hence highly uncertain) (Loehle, 2006). Subsequently, to model River Red Gum carbon and overbank flow dynamics (as described fully in Section 6.4) a rolling horizon approach is used. In rolling horizon decision-making, the decision maker makes the most immediate decision (e.g. decisions in the first period) based on knowledge of conditions in the present period and a forecast of the remaining years in the decision horizon (Sethi and Sorger, 1991). In the next period, the second period decisions become the most immediate and the decision horizon is pushed out further into the future, and is thus 'rolled over' (Sethi and Sorger, 1991). A rolling horizon approach provides a reasonable approximation to model the EWHs decision-making and it is shown in the sensitivity testing (Section 7.7) that there is no significant change in the reallocation outcome for a decision horizon over three years. To model the decisions of a forward-looking EWH in the rolling horizon framework, it is assumed that the modelled EWH understands the future probabilities of all future states of nature.

6.3.2 Ecosystem Service Valuation Methods, Challenges and Potential for Bias

Economic valuation of the environment is the process of attributing economic values to environmental goods and services to represent the value the environment provides to society. Economic valuation of ecosystem services supply relative to changes in environmental conditions is crucial in making informed economic decisions about trade-offs and policy alternatives (Fisher et al. 2008).

Broadly, there are two types of economic value of ecosystems: use values and non-use values (Krutilla, 1975). Use values are derived from direct, indirect or consumption use (de Groot et al., 2010). For example, consumptive use values encompass the value of food and fibre, freshwater, or timber. Non-consumptive direct uses are, for example; recreational use and aesthetic appreciation. Indirect uses include erosion prevention and local climate regulation. The other type of values are non-use or passive use (Randall 1993) which relate to the value attributed to an environmental attribute despite having no intention of using it directly or indirectly. One aspect of non-use is existence value which defines an individual's willingness-to-pay for the continued existence of a resource (Madaraga and McConnell, 1987). Another is bequest value which relates to the value of knowing future generations will be endowed with the resources of a healthy environment (Walsh et al., 1984). An additional category is options value, which includes the value assigned to the possible future use of the resource (e.g. future groundwater use) (Kurtilla and Fisher, 1975; Walsh et al., 1984). In uncertain conditions (e.g. adjudication of groundwater rights), additional quasi-options values (Rolfe, 2010) are sometimes included and represent the insurance value of possible future use (Walsh et al., 1984). The combination of use and non-use economic values provide an estimate of total economic-value (TEV).[33]

Table 6.2 provides a summary and example of valuation methodologies and approaches which can be used to elicit the economic value of ecosystem services. However, valuing ecosystem services involves challenges and potential for bias. This section discusses some important valuation challenges relevant to ES valuation.

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[33] The TEV diagram of water resources was shown in Chapter 2.

Table 6.2 Economic Techniques for ES Valuation

Valuation methodologies	Valuation approaches	ES examples
Market valuation	• Market price based approach • Cost based approach (avoided cost, replacement cost, mitigation or restoration cost) • Production function approach	• Market value of provisioning services sold in agricultural markets (e.g. the value of farming produce sold on the market). • Replacement cost (or avoiding of damages in the absence) of regulating ecosystem services with artificial technology (e.g. the cost of replacing a wetland with flood drains). • The value of enhanced marketed commodities due to improvements in input (e.g. increased farm productivity from improved soil fertility).
Revealed preference	• Travel cost approach • Hedonic pricing approach	• The direct expense and opportunity cost of time to visit an environmental asset (e.g. the time taken to drive to a recreational site). • The value of implicit demand for an environmental attribute of marketed commodities (e.g. house values derived from proximity to ecological asset).
Stated preference	• Contingent valuation approach (willingness-to-pay, willingness-to-accept) • Choice modelling approach • Group valuation	• Survey respondents' willingness-to-pay or -accept for changes in environmental conditions (e.g. willingness-to-pay for 1% increase in bird habitat). • Survey respondents' choice of attributes from a set of alternatives for a given context (e.g. willingness-to-pay for 1% increase in bird habitat vs. 1% increase in fish habitat) • Combined stated preferences and deliberative processes to uncover aspects such as value pluralism

<u>**Source: Adapted after Pascual et al. (2010).**</u>

Market valuation is the exchange value that ecosystem services have or would have if traded for currency (de Groot et al., 2002). It can be assessed using data from existing market, such as markets for carbon credits or water allocations, where relevant markets exist. Using market based approaches which rely on real price data reflecting the demands of individuals or societies, the value of the ES is reflected in market price. Market valuation provides an accessible means to estimate the exchange value of ecosystem services when traded on the market.

For services such as regulation services (typically not able to be traded in the market) avoided cost or replacement cost methods are able to be employed (Pascual et al., 2010). However, this approach has a number of challenge because it requires the fully effective least-cost option to be identified (Bakermann et al., 2008). In complex water resource systems with variable conditions and a number of decision variables at play, identifying the least cost effective solution can be a considerable challenge. For example, the treatment of cyanobacterial blooms (Blue green algae) may be treated using a variety of tools (e.g. in-situ treatment, flushing flows), the cost of which is state-dependent (e.g. the cost of

flushing flows increase during drought periods due to increased water prices to scarcity). As a result, Heal et al. (2005) argue that replacement costs or avoided cost methodologies are justifiable only under a limited set of conditions.

However, because many ecosystem services are public or common pool resources, they are not often traded in private markets (Hanemann, 1994; Costanza et al., 2014) and goods and services with no market price can still have economic value (Hanemann, 2006). Market pricing can overly emphasis the exchange value of an ES and rarely provides a full account of the value of open access, public good services and typically does not reveal society's willingness-to-pay for an ES. Overall, market pricing can sometime lead to an inappropriate and erroneous measure of value. Where public markets for ecosystems do exist, issues arise when markets are thin, absent or incomplete. When markets for non-excludible and non-rivalrous goods are missing or inappropriate for use, the valuation of public environmental goods, such as the existence of native habitat and species populations are more difficult to quantify (Hanemann, 1994). In such cases stated and revealed preferences or stated preference approaches can be employed (Bateman et al., 2002).

Revealed preferences approaches assume that individuals reveal their preferences through choices, such as where to live or what product to buy, and therefore have the useful characteristic of reflecting actual consumer preferences (Booker et al., 2012). In the valuation of public goods, recreational demand models, such as the travel cost method, or the hedonic value model are applicable and widely used (Morrison and Hatton-MacDonald, 2010; Booker et al., 2012). However, reliable data for revealed preference approaches (e.g. property prices for hedonic valuation; travel expenditures for the travel cost method; least cost action for avoided cost methodologies) can be rare or incomplete, and as such also has a range of challenges associated with this approach (Barkmann et al., 2008).

A third approach to ES valuation is the use of non-market valuation to elicit an individual's non-use values with stated preference approaches. An abundance of literature regarding stated preferences approaches and is only very briefly summarised here (see Bateman et al., 2002 for a thorough exploration of the approach). Stated preferences approaches elicit value by presenting survey participants with hypothetical payment scenarios for change in ES provision or environmental condition to derive their willingness-to-pay (WTP) (or –accept (WTA)). A discussion on the divergence between WTP and WTA estimates is provided in Hanemann (1991) who suggests that income and substitution effects could be the cause of the frequently observed differences between WTP and WTA.

To derive WTP and WTA estimates, contingent valuation is commonly used globally to estimate value (Hanemann, 1994; Nunes and van den Bergh, 2001). However, stated preference methodologies to measure non-use values remains controversial (Barkmann et al., 2008) and has historically been heavily criticised for lack of creditability and sources of bias (e.g. Diamond and Hausman, 1994). One source of bias in stated preference methodologies is information bias, which suggests that the information

provided (or not provided) to survey participants who may not have a comprehensive understanding of the issue may be a form of persuasive communication (Ajzen et al., 1996; Barkmann et al., 2008). Information bias may influence in cases where the non-expert surveyed individuals undervalue ecosystem services and environmental functions because they have little or no awareness of the benefits that are provided (Barkmann et al., 2008). Information bias is particularly relevant for the valuation of regulating services, and critics of this approach argue that survey participants will likely lack sufficient understanding of ecosystem functions (e.g. nutrient cycling, carbon cycling, water cycling) in order to make meaningful preference statements (Nunes and van den Bergh, 2001). Ajzen et al. (1996) find that when the environmental asset or service being valued has low perceived personal relevance to the survey group, subtle contextual clues can bias the value estimates. As a result, Nunes and van den Bergh (2001) argue that stated preference results can give a lower estimate of the value of ecosystem services. An additional source of bias in stated preference approaches of ecosystem services is hypothetical bias which is the difference between what individuals report they are willing-to-pay (or -accept) and what they would actually pay if money were to be collected (Loomis, 2014). This is typically assumed to produce an overestimate of willingness-to-pay, and has resulted in suggestions that such estimates should in fact be halved unless they could otherwise be validated (Yadav, 2007). Additional sources of bias may include the starting point bias, which occurs from the choice of initial hypothetical payments leading surveyed individuals to think around those price points (Boyle et al., 1985).

In addition to technical challenges described above, there are a number of considerations which arise from some societal sectors' and authors' hesitancy to *'put a price tag'* on nature. For example, Kosoy and Corbera (2010) argue that payment for ecosystem services is a form of commodity fetishism which they argue leads to: a) implications for the way humans perceive nature; b) the denial of multiplicity of values; and c) perpetuating power asymmetries and existing inequalities. Arguments against the valuation (e.g. in favour of the intrinsic value approach) and commodification of ecosystem services are particularly pronounced in flow-dependent systems because due to the idea that freshwater is the last 'bastion against privatization and commodification' (Morgan, 2004, p.2).

6.3.3 Carbon Storage Valuation

Monetizing the value of carbon stores to be comparable with traditional measures of economic value (e.g. water allocation price) enables standardised units of measure to be used in evaluating carbon-water trade-offs to inform water reallocation decisions. The economic benefit of allocating water to the environment to inundate the floodplains is therefore dependent on the price assigned to each tonne of carbon. When deciding to reallocate water to inundate floodplain vegetation to stimulate biomass growth and carbon sequestration, the EWH is faced with two prices of carbon: the local market price and the global social cost of carbon. For the two different carbon prices, the modelled EWH chooses a reallocations strategy that occurs when the marginal of increased carbon storage equal the marginal cost of reallocating water from irrigation.

6.3.3.1   Social Cost of Carbon

The global social cost of carbon (SC-CO_2) is the economic cost caused by an additional ton of carbon dioxide emissions or equivalent being emitted (Nordhaus, 2017). The SC-CO_2 is the amount that would get paid with a complete intergenerational market for carbon. Estimates of the SC-CO_2 vary based in assumptions about emissions trajectories, the impact future emissions will have, climate response and discount rate applied (IWG SC-CO_2). Estimates for the SC-CO_2 are shown in Table 6.3. In this thesis, the 'central' estimate is used, which is approximately AU$48/t$CO_2$ in 2015.[34]

Table 6.3 The Social Cost of Carbon 2010-2050.

Year	Discount Rate			High Impact (95th percentile at 3%)
	5% Average	3% Average	2.5% Average	
	The SC-CO_2 (USD $)			
2010	10	31	60	86
2015	11	36	56	105
2020	12	42	62	123
2025	14	46	68	138
2030	16	50	73	152
2035	18	55	78	168
2040	21	60	84	183
2045	23	64	89	197
2050	26	69	95	212

Source: IWG SC-CO_2 (2015), p. 4.
Note: The SC-CO_2 values are dollar-year and emissions-year specific.

6.3.3.2   Market Price of Carbon

The local market price of carbon[35] reflects the price the Australian government (e.g. the Clean Energy Regulator (CER)) would buy one tonne of carbon dioxide equivalent at auction. Emission Reduction Fund (ERF) auctions for carbon credits are used as a mechanism for farmers and landowners who have engaged in carbon abatement actions can sell the subsequent carbon credits to the CER. The first ERF auction for carbon abatement credits was held in April 2015 and established the average price per tonne of carbon at AU$13.95/t$CO_{2e}$ (CER, 2016). The ERF auctions have since been repeated five times. The price per tonne of carbon resulting from each auction and the average price over the five auctions ($11.79/t$CO_{2e}$) is shown in Figure 6.5.

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[34] Assuming a long-term conversion rate of 1:0.764 (AUD:USD).
[35] In 2011 the Australian *Clean Energy Act* (*Cwth 2011*) set an initially fixed price of AU$23/t$CO_2$, commonly referred to as the 'carbon tax'. The carbon tax was later repealed in 2014.

Figure 6.5 Carbon Prices in Australia
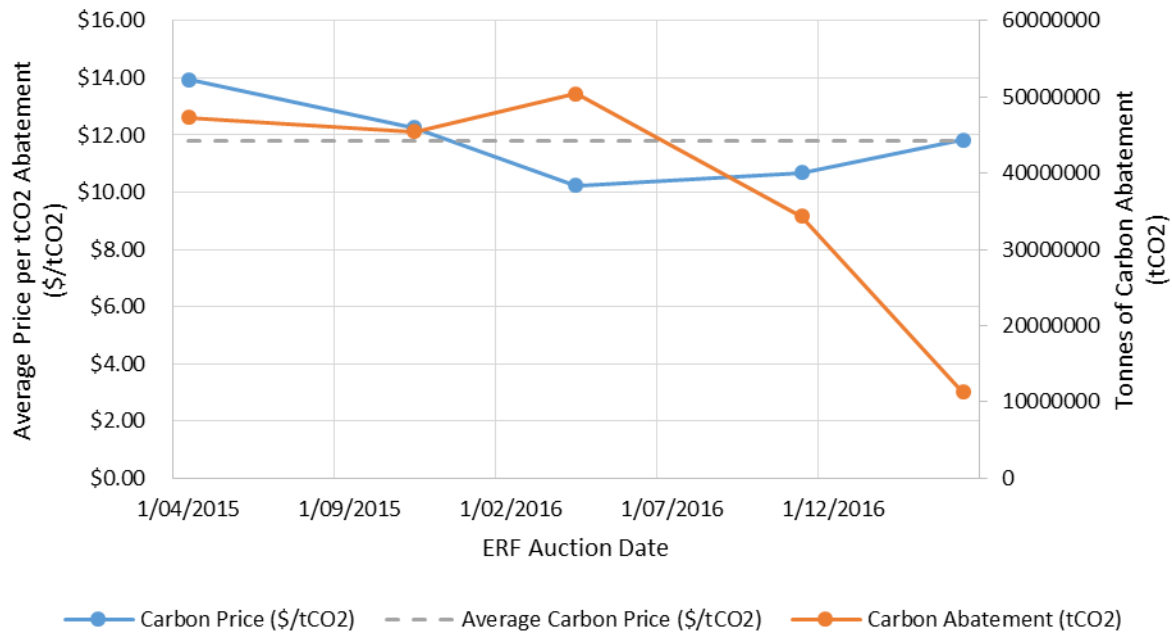
Source: **Own Figure. Data collected from Clean Energy Regulator 2015a; 2015b; 2015c; 2016a; 2016b; 2017.**

6.3.4 Selection of Discount Rates

Discounting reflects the value that society places on the flow of benefits in the future. A discount rate of zero implies that society has no preference for receiving benefits in future periods relative to today. Discount rates of larger than zero are typically applied to reflect society's preference to receive benefits in the current rather than the future periods. The choice of discount rates in climate and environment related policy is specifically important because there is a substantial time lag between cause and effect, such that the carbon abatement costs will be incurred in the present but avoided damaged are received well into the future. The choice of discount rate can lead to variation in climate and environment policy recommendations (Moxnes, 2014).

With public goods, there is an intergenerational equity argument to not completely discount future benefits by applying a high discount rate. As a result low discount rates are typically applied for environmental projects which generate benefits far into the future (Harrisson, 2010). An example of this is the *Stern Review* (Stern, 2006), which applied a discount of 1.4% and the results of which subsequently called for immediate climate change action.

Conversely, some authors have applied higher discount rates at the 3-5% level (e.g. Nordhaus, 2007). Higher discount rates weaken the case for investment in mitigation activities that accrue benefits over long periods, such as climate mitigation climate change mitigation (Gouler and Williams, 2013) and instead favour projects which accrue benefits early in the project life (Harrison, 2010).

To select the appropriate discount rate, existing guidelines are followed. For environmental projects the US-EPA recommends a discount rate of 2-3% (US-EPA, 2000). The PC in Australia suggest the

application of discount rates between 3-10%, including sensitivity testing within this range (Harrison, 2010). When evaluating the costs and benefits of a carbon abatement or sequestration program, it is important to maintain internal consistency between the discount rate applied to the social cost of carbon and the discounting of other costs and benefits considered (IWG SC-CO$_2$, 2015). As discussed, the central SC-CO$_2$ prediction which uses a 3% discount rate is employed. This discount rate is consistent within the bounds recommended by the Australian PC (Harrison, 2010) and US-EPA, and is therefore applied in the model.

6.4 Modelling Case Study

6.4.1   The Murrumbidgee Catchment

The model is developed to simulate the water reallocation decisions in the Lowbidgee wetland area of Murrumbidgee sub-catchment in the MDB. The Murrumbidgee catchment covers 8.2% of the total area of the MDB and accounts for 22% of MDB consumptive water diversions (CSIRO, 2008c). The catchment is bordered by the Great Dividing Range in the east, the Lachlan River catchment in the north and the Murray catchment in the south. The topography differs significantly across the region, varying between alpine mountains to low lying plains (Wen, 2009). The climate in the Murrumbidgee is diverse and average rainfall ranges from 1700mm in eastern side of the catchment and 350mm/year in the western plains (Green et al., 2011). The Murrumbidgee River is 1,600km in length and flows westwards to a confluence with the Murray River. The headwaters of the Murrumbidgee River are the Snowy Mountains Hydro-electric Scheme that is required to release a minimum of 1,026GL of water into the Murrumbidgee River per year, subject to storage and savings diverted into the Snowy River (CSIRO, 2008c).

Agriculture is concentrated in the Murrumbidgee and Coleambally irrigation areas on either side Murrumbidgee River (Wen, 2009; Green et al., 2011). The vast majority of irrigated agriculture in the Murrumbidgee is cereal for grain, including rice, wheat and millet. This accounts for 396,258.9 ha of agricultural area in the catchment (ABS, 2010c).

The Murrumbidgee River is highly regulated by storages to provide reliable year-round supply for agriculture and domestic use (Kingsford, 2003; Wen, 2009). Large-scale catchment modifications to accommodate agriculture and industrial development have reduced the volume and frequency of overbank flooding required to inundate the floodplains and maintain ecological health (Kingsford, 2003; Page et al., 2005).

The Lower Murrumbidgee floodplain (the 'Lowbidgee') is defined as the bounds of the modelling study. The Lowbidgee, shown in Figure 6.6, is located at the south-western end of the catchment between the townships of Hay and Balranald is the most extensive wetland area in the Murrumbidgee catchment (Kingsford, 2003). The Lowbidgee is a particularly good example of Lignum swamps and River Red Gum (*Eucalyptus camaldulensis)* forests (Kingsford, 2003; MDBA, 2012b). The Lowbidgee

is a hydrological indicator site for the Basin Plan and is the focus of environmental flow management (Wen, 2009) and receives environmental watering from water owned by the Commonwealth Environmental Water Office and the NSW Office of Environment and Heritage (DoE, 2016). Concern over the declining health of the River Red Gum forest during the severe 'Millennium Drought' (1997-2010) (BOM, 2015) prompted environmental watering in 2000 and mid-2010 (Doody et al., 2015).

6.4.1.1 The Existing Water Market

Water markets have been present in Australia for a number of decades and are well-established; especially in the southern part of the MDB which is hydrologically connected. A detailed description of the evolution of water markets and their current geographic and institutional scope in the MDB was provided in Chapters 2 and 3. The Murrumbidgee is part of the sMDB water market and participates in two main forms of water trade: water entitlement trade and water allocation trade. A water entitlement is an entitlement to a total volume of water available in a basin in any given year, subject to the rules of that entitlement. A water allocation is the amount of water that is allocated to an entitlement in any single water year, subject to seasonal water availability and entitlement rules. Water allocation trades can occur seasonally or temporarily, while water entitlement trades occur permanently. Annual prices vary considerably in the allocation water market and are driven predominately by water scarcity factors of drought, low rainfall and low water allocations drive higher water prices (Wheeler et al., 2014a). Most of the trades that occur in the water market are water allocation trades and this is the water market that is considered in this model. Further description of water entitlements and allocation markets was provided in Section 3.2.

6.4.1.2 The Potential Carbon Market

As a ratified party to the *Paris Agreement,* Australia has committed to reducing its' emissions by 26-28% by 2030 relative to 2005 levels. Progress towards emissions reductions is governed by the Australian Emissions Reduction Fund (ERF) which provides incentives for emissions reduction activities across the Australian economy (DEoE, 2016). An important ERF funded activity is the generation of carbon credits though activities undertaken to increase carbon storage in forests (the 2011 *Carbon Farming Initiative*). Carbon credits may then be sold on the carbon market to individuals, cities or companies interested in offsetting or reducing their net carbon emissions. One option to generate potentially tradable carbon credits is to actively manage existing forests to promote carbon uptake and prevent carbon loss (Law, 2013).

This study investigates the possibility of actively managing the floodplain forest carbon storage by annually reallocating water to the environment to augment the overbank flooding regime and improve River Red Gum forest condition, and hence carbon uptake. The potential sale of the carbon credits generated by improved the River Red Gum condition is the credits considered in this case.

Figure 6.6 Location and extent of Lower Murrumbidgee ('Lowbidgee') River Floodplain.
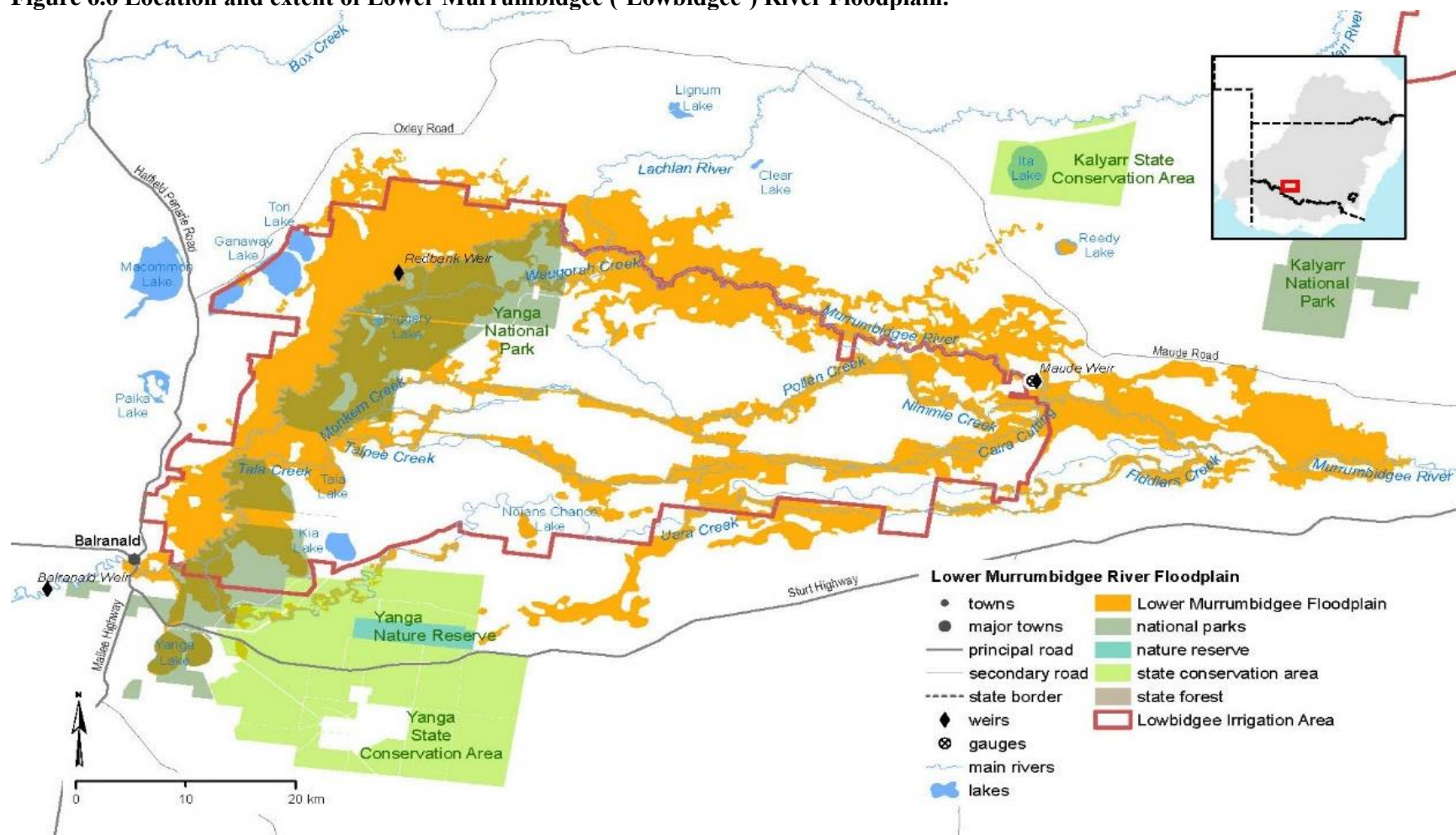
Source: MDBA (2012b), p.4

6.5 Model Development and Methodology

6.5.1 Methodology Overview

To simulate an annually dynamic water reallocation strategy and evaluate the carbon-irrigation water trade-offs faced by an EWH wishing to buy and manage water for the environment, an integrated HEM is developed. The HEM is based on a previous model developed by Connor et al. (2013).

The novelty of the model presented within this thesis as compared to Connor et al. (2013) is the incorporation of more complex ecosystem service dynamics. In particular, it replaces a simple environmental damage function (details and comparisons made to the original function in Section 7.5.4) with a more realistic stock-and-flow model of floodplain vegetation carbon sequestration in response to hydrological inundation events (or absence therefore). The model presented in this thesis also extends Connor et al. (2013) by incorporating a carbon valuation sub-model to assign a dollar value to the carbon stocks in the floodplain vegetation. Lastly, using outputs from the carbon valuation sub-model the model presented in this thesis develops a reallocation algorithm that compares the carbon and irrigation opportunity costs to choose an annually dynamic optimal reallocation scenario. Based on the new construction of the model, a range of new questions are able to be asked, namely: is it possible for water reallocation costs to be offset by the generation and sale of carbon credits derived from improved floodplain vegetation health?

The model is developed in the General Algebraic Modelling System (GAMS) and is shown graphically in Figure 6.7. Methodologically the approach is dynamic simulation. This is an appropriate choice for models such as this with many integrated elements, time steps and spatial units that pose difficulties to optimisation (Harou et al., 2009; Momblanch et al., 2016).

The baseline model began with a 113 year historic (1896 to 2008) sequence of flows. Water allocation to agriculture and flow to the Lowbidgee floodplain were computed with a calibrated catchment hydrology network node model assuming current water-sharing rules and irrigation development levels. With floodplain inflow as an input, inundation modelling computed frequency and area of inundation by zonal areas with differing elevation and river connectivity. Inundation modelling results are input to a forest carbon stock model built on locally calibrated River Red Gum carbon storage growth and decay responses to water deficits and overbank floods. Two economic valuation components follow the biophysical process modelling. For carbon, four exogenous price levels are set ranging from the two most recent Australian ERF average auction prices; the initial price set on carbon by the 2011 *Clean Energy Act*; and the social cost of carbon. The irrigation opportunity cost model used a regression-based function relating historic irrigation water allocation levels to observed water allocation price to estimate the price impact of reallocating an increment of water from irrigation to the environment. Finally, a water reallocation choice algorithm is executed. This involves iterating the entire integrated model across 5% increments of additional water re-allocations from irrigation to the environment each year,

comparing the economic value of carbon improvements with irrigation water opportunity cost (water allocation price) and choosing the highest level of annual water reallocation with carbon credit value equal or in excess of water cost (bounded by zero and 50% water reallocation limits). Modelled annual water reallocations occur for a period of one year and are then returned to the consumptive pool (e.g. comparable to an annual water lease/water allocation sale).

The overall HEM consists of sub-models representing: a) catchment hydrology; b) floodplain carbon storage dynamics; c) carbon credit valuation; d) water allocation price; and e) water reallocation algorithm. The hydrology and water price sub-models are adapted from Connor et al. (2013).

Figure 6.7 Conceptual Model Overview and Key Outputs
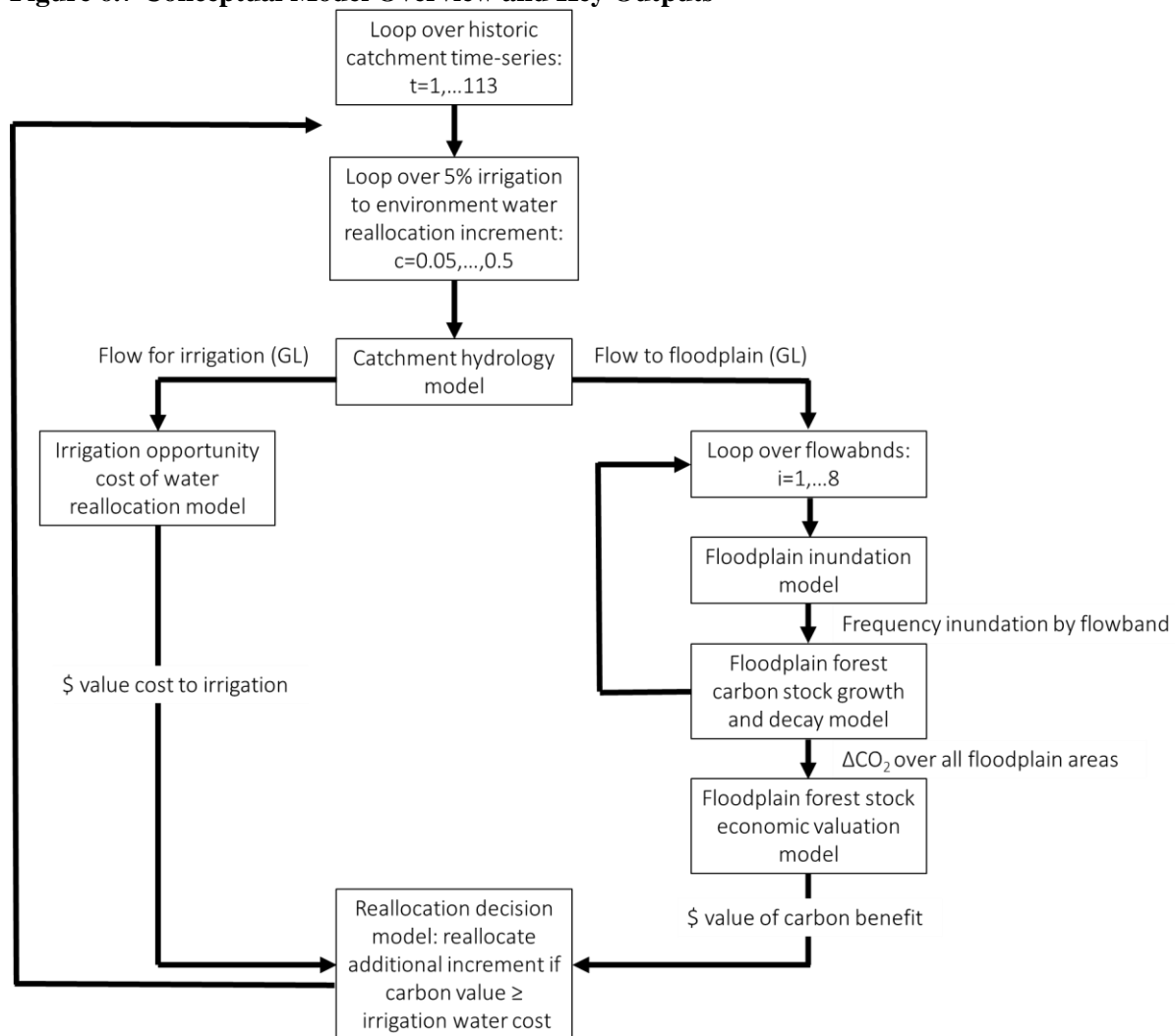
6.5.2   Data and Assumptions

6.5.2.1 Data

Data sets from a range of disparate fields were used to develop and calibrate the model and are shown in Table 6.4. Table 6.4 summarises the data sets used in the model development and calibration. Existing secondary data sets from a number of disparate fields unintended to be used in conjunction with one

another were used in the model development and as such, considerable manipulation was required. Further detail regarding data manipulation is provided in Appendix B.

Table 6.4 Description of Datasets Used in the HEM

Data	Source	Year or period	Sample size
Announced water allocations	MDBA Water Audit Monitoring Reports (1996-2009)	1996/97 – 2008/09	N/A
Water allocation price data	Victoria Water Register; Murrumbidgee Irrigation reports	1996/97 – 2008/09	N/A
Carbon prices	Clean Energy Regulator auction results (2015; 2016; 2017) US – EPA (2016) The Carbon Credits (Carbon Farming) Initiative (2011)	2015-2017 - -	N/A
Hydrological parameters	Kirby et al. (2013) M. Kirby and J. Connor (2014, pers comms.)	N/A	N/A
Hydrology time-series inflows	M. Kirby and J. Connor (2014, pers. comms.)	1896-2008	N/A
Climate change scenarios	CSIRO (2008)	N/A	N/A
Lowbidgee environmental watering requirements	MDBA (2012)	N/A	N/A
River Red Gum environmental watering requirements and annual transpiration rates	Doody et al. (2016)	2010-2011	N=40 trees
River Red Gum health metrics	Catelotti et al. (2015)	1993, 1994, 2004, 2011/12	N = 212 trees
River Red Gum carbon storage capacity	Smith and Reid (2013)	2008	N=61 trees
Native tree growth parameters	CSIRO, unpublished report.	N/A	N/A

Source: Own Table.

6.5.2.2 *Assumptions*

A number of assumptions were required to fully integrate the hydrology, economics, and carbon storage and sequestration sub-models into one cohesive HEM. Before describing the modelling methodology in depth, this section provides an overview of the key assumptions and their necessity.

When developing the carbon sequestration model it is assumed that, considering all else constant, the rate of carbon uptake and storage is proportional to changes in tree health. Tree health is inferred assuming an explanatory relationship between average annual transpiration rates and tree condition. Data analysis demonstrates an approximately exponential relationship between site flooding frequency and average annual transpiration (see Appendix A), which is consistent with existing ecological models of drought response. The assumption linking carbon storage capacity to tree health allows the integration of existing tree growth curves, which capture the influence of disturbances such as floods

and droughts, into the model. This assumption is also required because there is, in general, a lack of existing reliable methods to quantify the carbon sequestered in restored native forests (Jonson and Freudenberger, 2011). Where methods do exist, they rely primarily on tree measurements regarding height, crown extent and diameter at breast height. For the case study examined, such measurements are not available and it would not be feasible or cost-effective to obtain given the extreme heterogeneity of native forests.

Overbank flooding is assumed to be the primary determinant of floodplain tree health, which in turn drives the rate of carbon uptake and storage. This assumption was made because it is known that under low-to-medium water availability conditions overbank flooding and broad area inundation are the main determinant of floodplain condition (Junk et al., 1989; Poff et al., 1997; Page et al., 2005, Overton et al., 2009; Wen, 2011). Additional influences on tree health such as access to groundwater and nutrients, salinity levels and stand competition are omitted from the modelling as their inclusion does not further the aims of this study or provide any further explanatory value to answer the proposed research question.

Further, in absence of more detailed data on the relationship between tree health and carbons storage capacity, it is assumed that the average observed volume of carbon stored per hectare of River Red Gums is the maximum possible volume able to be stored per modelled hectare of River Red Gums. There is no available data describing the maximum carbon storage potential for River Red Gums when maintained under optimal site conditions. Difficulties further arise as the maximum carbon storage capacity is a function of tree age, strand density and forestry management practices, among others. Further, in absence of primary data, it is assumed that at the start of the simulation ($t=0$) all floodplain River Red Gums are non-water stressed and containing the modelled maximum (i.e. mean) carbon storage volume.

In terms of valuation, tonnes of carbon sequestered in the River Red Gums is considered equal value of carbon abatement credits which can be sold on the market. The price of carbon is assumed to be constant throughout the simulation. This assumption was required because one of the aims of the modelling is to determine at which price it is possible to offset environmental water allocation purchases with the sale of carbon credits, and thus constant prices throughout the simulation is required to achieve this. In future research there would be benefit in amending this assumption, as it is known that the social cost of carbon is likely to increase in the future (IWG SC-CO$_2$, 2015) and that the market price of carbon is variable in Australia based on changing laws, policies or auctions. The inclusion of highly variable carbon prices would likely influence annual water reallocation decisions and the profitability of the reallocation decisions.

An additional underlying assumption is that it is possible for the carbon captured and stored in the River Red Gum forests can be converted fully into carbon credits, which can then be sold in the carbon market. However, at present existing native forests are excluded from national carbon accounting schemes, and

there are a number of additional challenges associated with management and sale of carbon credits in native forests, as discussed in detail in Section 7.8.2 which reflects on modelling results and possible applications and challenges.

The hydrology model uses a sequence of 113 years of modelled inflow data. It is assumed that this hydrological series captures the required variability of MDB hydrology required to inform water decision-making. Further, the possible feedbacks between biomass growth on the floodplain as a result of improved overbank flooding and impacts on the runoff characteristics of the catchment are ignored in this study, although it is acknowledged that the potentially negative the impact of floodplain vegetation on stream flow is well documented, especially for new plantations (Young and McColl, 2009; Schroback et al., 2011). However, it is anticipated that because the case study considers only an already existing floodplain forest, the possible negative changes in stream flow due to altered vegetation cover is considered negligible in the current context.

To model the decisions of a forward-looking EWH, it is assumed that the forward-looking EWH has perfect foresight of probabilities of future states of nature arising. As discussed in Section 6.2.1, a rolling decision horizon of three years is used, and sensitivity testing is undertaken around the number of years in foresight. The model does not include any learning rules for the EWH or irrigators, however it is known that both parties become market savvy through trading experience over time.

Lastly, the model is set within the current institutional setting for water trade in the MDB. To simulate this it is assumed that water allocation trade between the modelled EWH and irrigators is frictionless. Further, the model investigates the annual trade of water allocations only and entitlements are not traded between irrigators and the environment in the model. It is initially assumed that the environment is given only a baseline level (i.e. 2009 conditions) of water entitlements, and no permanent water rights are given to the environment on the outset of the analysis. This assumption is later relaxed to investigate the impact of trading after the final implementation of the Basin Plan, which returns approximately 20% of entitlements to the environment (see Chapters 2 and 3 for further detail).

6.5.3   Description of Variables

This section provides a description of the variables and their units used in model as described in Equations 6.4 to 6.25. Variable definitions are grouped by sub-model component in Table 6.6 to 6.9 Before introducing the model variables, Table 6.5 presents the indices which control the range of the variables.

Table 6.5 Name, Definition and Range of Variable Indices

Index	Range	Definition
i	1-8	Flowband
t	1-113	Simulation period
c	0% - 50%	Annual proportion of consumptive rights reallocated to the environment
r	Headwaters, dam, river	Geographic node in the hydrology model

Source: Own Table.

Table 6.6 Hydrology Model Variable Definitions

Variable	Unit	Definition
$S_{(t,r)}$	GL/year	Water storage in a node (e.g. dam)
$I_{(t,r)}$	GL/year	Annual inflow to a node (e.g. dam, river)
$iw_{(t,c)}$	GL/year	Water abstracted from the river for irrigation
$ew_{(t)}$	GL/year	The volume of water permanently allocated to the environment surplus to baseline catchment outflows.
$et_{(t,c)}$	GL/year	The volume of water annually leased by the EWH for the environment
$cw_{(t)}$	GL/year	Conveyance water released in the river representing all other outflows
$l_{(t,r)}$	GL/year	Losses due to evaporation and seepage
S_{max}	GL	Maximum dam storage capacity
$Sp_{(t)}$	GL/year	Dam spill volume. Spill is zero unless the sum of all inflows exceeds the maximum dam storage capacity. When non-zero spill is given as the difference between storage capacity and inflow volume
$O_{(t,r)}$	GL/year	Annual outflow from a node in the catchment water balance (e.g. dam, river)
$ro_{(t,r)}$	GL/year	Catchment runoff into the river
$\Delta S_{(t,r)}$	GL/year	Change in storage volume between years

Source: Own Table.

Table 6.7 Carbon Storage Dynamics Model Variable Definitions

Variable	Unit	Definition
$ER_{(i,t)}$	GL/year	Desired inflow to the Lowbidgee floodplain to meet environmental objectives
$EF_{(i,t)}$	Years	Desired return interval for inflows to the Lowbidgee
$A_{(i)}$	Ha	Area of floodplain River Red Gums inundated
$MaxCarbonStorage_{(i,t,c)}$	tC/ha	The maximum volume of carbon that can be stored in a hectare of River Red Gums per year
$MinCarbonStorage_{(i,t,c)}$	tC/ha	The minimum volume of carbon that can be stored in a hectare of River Red Gums per year
$CarbonDecay_{(i,t,c)}$	tC/ha/year	The annual increment of carbon decay per hectare of River Red Gums
$EC_{(i,t,c)}$	Years	The number of sequential periods past the desired return period for a flooding event
Y	Years	The number of sequential years of no flooding required to cause tree death.
v		Carbon decay coefficient
$CarbonGrowth_{(i,t,c)}$	tC/ha/year	The annual increment of carbon growth per hectare of River Red Gums
D	tCO_2	The asymptotic increment of carbon uptake at forest maturity ($y_{(\infty)}$)
F	N/A	Gompez carbon growth calibration parameter
$EB_{(i,t,c)}$	Years	The number of sequential years that environmental watering requirements have been consecutively met
k	-	Tree growth parameter for a slow growing Eucalyptus species
$CarbonStorage_{(i,t,c)}$	tC/ha	A temporary carbon accounting variable used to account for carbon stored in period t-1
$NetFloodplainCarbonStorage_{(t,c)}$	tC	The total volume of carbon stored across the whole floodplain in each period
$w_{(i)}$	N/A	Unit-less weight parameter proportional to the area between each zonal floodplain area

Source: Own Table.

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Table 6.8 Carbon Valuation, Water Price and Irrigator Opportunity Cost Variable Definitions

Variable	Unit	Definition
$CarbonValue_{(t,c)}$	$	The total value of the carbon stored in the floodplain River Red Gums
$\Delta CarbonValue_{(t,c)}$	$	The marginal difference in the value of carbon stocks potentially achieved for reallocation water to the environment in proportion, c, relative to a reallocation proportion of $c-1$ $\Delta CarbonValue_{(t,c)}$ is a measure of carbon opportunity cost
CP	$/tC	The price per tonne of carbon
$Water\,Price_{(t,c)}$	$/ML	The price of annual water allocations
β_0	N/A	Regression parameter for annual water allocation price model (β_0=2716)
β_1	N/A	Regression parameter for annual water allocation price model (β_1=-798)
$IrrigOC_{(t,c)}$	$/ML	The opportunity cost of irrigation water forgone

Source: Own Table.

Table 6.9 Foresight and Reallocation Model Variable Definitions

Variable	Unit	Definition
$CarbonValue\,Pr_{(t,c)}$	$/year	The present value of the change in carbon storage generated by all possible discrete annual reallocation proportions, c which occur in each period , over the decision horizon $t+n$
n	Years	The number of years in foresight beyond the current time period
δ	Percent	Discount rate applied to the future flow of benefits. δ=3 %
T	Years	The length of the simulation (e.g. number of periods)
$IrrigOC\,Pr_{(t,c)}$	$/year	The present value of irrigator opportunity costs of water foregone for all possible discrete annual reallocation proportions, c, which occur in which occur in each period , over the decision horizon $t+n$
h	N/A	The point at which the marginal benefit of carbon storage improvements are equal to the marginal cost of irrigators forgoing water-use
j	$	The level of tolerance around the point of marginal equilibrium

Source: Own Table.

6.5.4 Hydrology and Floodplain Inundation Model

The hydrology model is an annual water balance for the Lowbidgee catchment and is a component of a whole-of-basin hydrological model described in Kirby et al. (2013). The hydrology component is a modelled as a direct flow network and described using water balance equations to represent inflows, two dam nodes, consumptive water demands, losses, dam storage and spill, baseline environmental

water supply and annual environmental water reallocation. The hydrological simulation runs over 113 years using the historic climate sequence (1896-2008). The baseline water available to inundate the Lowbidgee floodplain is the annual river outflow, $O_{(t,r=river)}$ which is the sum of the dam outflows and catchment runoff less irrigation abstraction and losses (e.g. evaporation, seepage). This section provides an overview of the hydrology component for context.

Inflows into the first dam node are from headwater releases in the Snowy Mountains. Outflows are the addition of consumptive water demands, environmental water supply, leased environmental water and losses from seepage and evaporation. The model is first run with no reallocation to the environment for a baseline result. The basic water balance for each dam node is given in Equations 6.4 to 6.8. In each period, dam storage and dam spill are given by:

$$S_{(t,r=dam)} = S_{(t-1,r=dam)} + I_{(t,r=headwaters)} - iw_{(t,c)} - ew_{(t,c)} - et_{(t)} - cw_{(t)} - l_{(t,r=dam)}$$ **(Equation 6.4)**

$$Sp_{(t)} = max\left[0, S_{(t,r=dam)} - S_{max}\right]$$ **(Equation 6.5)**

Where $S_{(t,r=dam)}$ is the volume of water stored in the dam in period t, where t is an index of years and r is a geographic index representing *headwaters, dam, catchment* or *river*. $I_{(t,r=headwaters)}$ is inflow into the dam from the headwaters, $iw_{(t)}$ is irrigation consumptive water demand, $l_{(t)}$ is all losses from the dam, $cw_{(t)}$ is conveyance water representing all other outflows, $ew_{(t)}$ is the baseline environmental water supply and $et_{(t)}$ is any consumptive water leased to the environment each year and can be zero. $Sp_{(t)}$ is dam spill which is zero if cumulative inflows do not exceed dam capacity, S_{max} and is otherwise given as the volume that inflows exceed dam capacity. Variable $et_{(t,c)}$ can be altered by the EWH decisions by annually reallocating water in 5% increments up to 50% reallocation each year. For baseline conditions $et_{(t,c)}$ is set to zero. The outflow from the dam node is given by:

$$O_{(t,r=dam)} = iw_{(t)} + ew_{(t)+}et_{(t)} + cw_{(t)} + Sp_{(t)}$$ **(Equation 6.6)**

Where $O_{(t,r=dam)}$ is the outflow from the dam node in each period and all other variables are previously defined. The outflows from the dam plus any catchment runoff become the inflows into the river reach:

$$I_{(t,r=river)} = O_{(t,r=dam)} + ro_{(t,r=catchment)}$$ **(Equation 6.7)**

Where $I_{(t,r=river)}$ is the inflow to the river and $ro_{(t,r=catchment)}$ is the catchment runoff into the river. Outflows from the river are the river inflows less any irrigation abstractions, losses due to evaporation and change in river storage. The river outflow is the baseline volume of water available to flow to inundate the Lowbidgee floodplain. River outflows are given by:

$$O_{(t,r=river)} = I_{(t,r=river)} - iw_{(t)} - l_{(t,r=river)} - \Delta S_{(t,r=river)} \qquad \textbf{(Equation 6.8)}$$

Where $O_{(t,r=river)}$ is the outflow from the river, $l_{(t,r=river)}$ is losses in the river channel and $\Delta S_{(t,r=river)}$ is any change in river channel storage, and other parameters are previously defined.

To maintain floodplain productivity and vegetation health, floods of certain volume are required to be delivered to the Lowbidgee floodplains within ecologically appropriate time periods. The desired environmental water flooding volumes and frequencies for the Lowbidgee floodplain River Red Gums are shown in Table 6.10.

Table 6.10: Lower Murrumbidgee Environmental Watering Objectives

Flowband	Total inflow to the Lowbidgee floodplain (GL/year)	Desired return interval (years)	Percent Area of River Red Gum areas inundated	Area of River Red Gum inundated (ha)
(i)	$(ER_{(i,t)})$	$(EF_{(i)})$	$(\%)$	$(A_{(i)})$
1	50	0.95	2	1073.6
2	100	1.1	5	2684.1
3	170	1.33	11	5905.1
4	270	1.43	18	9662.9
5	400	1.67	25	13420.7
6	800	2	47	25231.0
7	1700	4	84	45093.7
8	2700	6.67	96	51535.6

Source: Adapted from MDBA (2012b), p. 13.

The floodplain is divided into zonal areas, $A_{(i)}$, each of which is inundated by a flow $ER_{(i)}$ at a desired return period in years $EF_{(i)}$. Higher flows are required to inundate broader and higher elevation regions of the floodplain, as demonstrated in Figure 6.8. For example, $ER_{7} = 1,700\text{GL}$ is the flow level required to create an inundation of flowband area 7 of 45,093ha and is required at return interval $EF_{7} = 4$ years for the forest in this floodplain region to remain in a healthy condition.

Figure 6.8 Flood Volumes and Inundation Area for a Vegetated Floodplain.
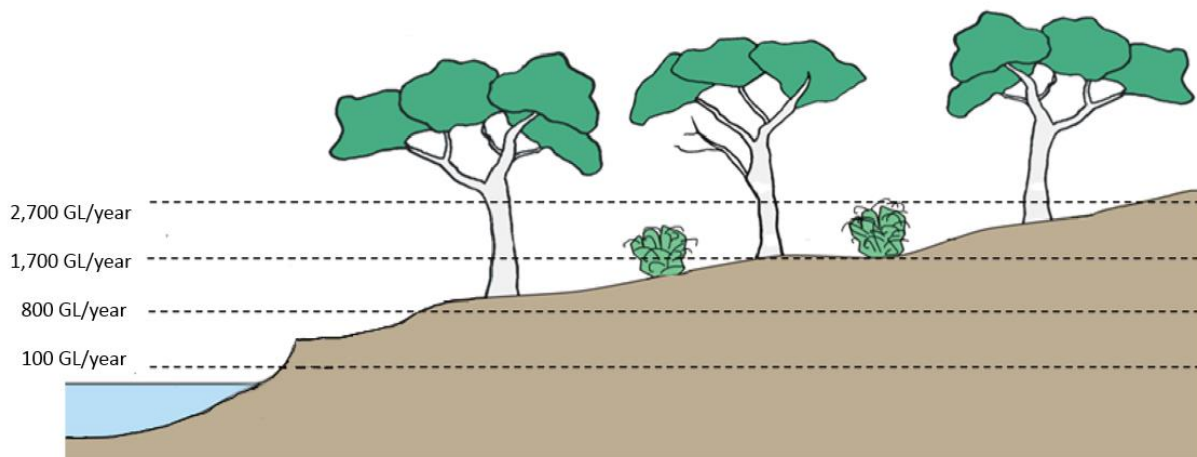
To meet the environmental flooding objectives, the modelled EWH has the option to increase the volume of water flowing to the Lowbidgee floodplains by reducing consumptive extractions in 5% increments and increasing flows to the floodplain accordingly. The proportion of water annually allocated to improve floodplain watering is governed by the variable c which increases in 5% increments from 0% to 50%. To identify periods where reallocation can be most ecologically beneficial, the EWH evaluates opportunities to improve the volume or prevent decay of carbon stored in the floodplain River Red Gums.

6.5.5   Carbon Storage Dynamics Model

This study focuses on the carbon stock dynamics of River Red Gums because: a) they are the dominant native tree species of the Lowbidgee (Doody et al., 2015); b) they store the majority of carbon in natively vegetated floodplains (Smith and Reid, 2013); and c) they are an useful indicator of overall ecosystem health (Catelotti et al., 2015). The mean carbon storage capacity of River Red Gums is 104.4tCO_2[36] per hectare of randomly sampled floodplain forest in NSW (Smith and Reid, 2013). To maintain plant productivity and health, River Red Gums require inundation in ecologically appropriate volumes and frequencies, as outlined in Table 6.10. Floodplain River Red Gum condition degrade when flood volumes and frequencies are not met (MDBC, 2003; Doody et al., 2015).

A stock-and-flow model of annual potential carbon storage was developed based on the biophysical condition of the Lowbidgee River Red Gums in response to overbank flooding and water deficit events. In each year the stock represent the volume of carbon stored in the River Red Gum at any one time

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[36] Additional carbon is stored in other carbon components of River Red Gum trees, but it is a negligible amount compared to the woody component. For example, the herbaceous component of River Red Gums stores a mean volume of 0.7tCO_2/ha; 1.6tCO_2/ha is stored in plant litter and 6.2tCO_2/ha is stored in coarse woody debris (Smith and Reid, 2013). Carbon is also stored in River Red Gum roots and surrounding soil but is not considered within this study due to the lack of process data relating overbank flooding with soils carbon dynamics. Estimates of the total carbon storage and the subsequent monetary value of the forest carbon sink are therefore conservative estimates.

expressed in terms of tonnes of carbon. This is governed by the $MaxCarbonStorage_{(i,t,c)}$, $MinCarbonStorage_{(i,t,c)}$ and $CarbonStorage_{(i,t,c)}$ variables. Conversely, the flow represents the annual change in carbon (either a uptake of carbon into tree biomass, or a release of carbon from tree decay) expressed as a rate of tonnes of carbon sequestered or released per hectare per year. The flow of carbon is governed by the $CarbonGrowth_{(i,t,c)}$ and $CarbonDecay_{(i,t,c)}$ variables.

Four ecosystem states were developed to describe River Red Gum carbon stock dynamics, namely: a) growth state; b) decay state; c) steady state equilibrium; and d) tree death equilibrium. The carbon states are defined in Table 6.11. The definition of variables expressed in Table 6.11 are defined in Table 6.5 to Table 6.9.

Table 6.11 Description of Modelled Ecosystem States and Conditions

Ecosystem State	Conditions	Description
Growth state	$EC_{(i,t,c)} = 0$ $EB_{(i,t,c)} > 0$ $CarbonStorage_{(i,t,c)} <$ $MaxCarbonStorage_{(i,t,c)}$ $CarbonStorage_{(i,t-1,c)} \leq$ $CarbonStorage_{(i,t,c)}$	An area of River Red Gum is in the growth state (e.g. net increase of vegetation carbon stocks) for time t that floodplain area $A_{(i)}$ receives flooding of volume $ER_{(i,t)}$ or greater within a time equal to or less than the desired return interval $EF_{(i,t)}$.
Decay state	$Y > EC_{(i,t,c)} > 0$ $EB_{(i,t,c)} = 0$ $CarbonStorage_{(i,t,c)} <$ $MaxCarbonStorage_{(i,t,c)}$ $CarbonStorage_{(i,t-1,c)} \geq$ $CarbonStorage_{(i,t,c)}$	An area of River Red Gum is the decay state (e.g. net decrease of vegetation carbon stocks) for time t that floodplain area $A_{(i)}$ receives flooding of volume $ER_{(i,t)}$ in an interval that exceeds $EF_{(i,t)}$ but it less than the time past desired return interval to cause tree death Y.
Dead tree equilibrium	$EC_{(i,t,c)} > Y$ $EB_{(i,t,c)} = 0$ $CarbonStorage_{(i,t,c)} =$ $MinCarbonStorage_{(i,t,c)}$ $CarbonStorage_{(i,t-1,c)} =$ $CarbonStorage_{(i,t,c)}$	An area of River Red Gum is in tree death equilibrium for time t that floodplain area $A_{(i)}$ receives flooding of volume $ER_{(i,t)}$ in an interval that exceeds $EF_{(i,t)}$ and the time past desired return interval to cause tree death, Y.
Steady state equilibrium	$EC_{(i,t,c)} = 0$ $EB_{(i,t,c)} > 0$ $CarbonStorage_{(i,t,c)} =$ $MaxCarbonStorage_{(i,t,c)}$ $CarbonStorage_{(i,t-1,c)} =$ $CarbonStorage_{(i,t,c)}$	An area of River red Gum is in steady state equilibrium for time t that floodplain area $A_{(i)}$ receives flooding of volume $ER_{(i,t)}$ or greater within a time equal to or less than the desired return interval, $EF_{(i,t)}$.

The transition between states occurs on an annual basis and is modelled as a function of the severity of hydrological events (e.g. water deficits and overbank flooding), which are represented using separate drought and flood counters. The drought counter iteratively sums the number of consecutive years that a flood volume for each floodplain zonal area exceeds the desired return interval for each flowband. The drought counter reverts to zero when a flood of desired magnitude and frequency is delivered to each flowband. Conversely, the flood counter iteratively sums the number of consecutive years that desired flooding events occur for each floodplain area within the desired timeframe, and reverts to zero when the desired return period is exceeded.

Carbon growth dynamics are maintained for the periods where two conditions are met: a) the flood volumes are delivered within the desired intervals such that the drought counter is zero; and b) the maximum potential carbon storage per hectare is not yet reached. Carbon storage dynamics transition to a steady state equilibrium once: a) the maximum carbon storage potential is established; and b) the drought counter remains zero. When the drought counter exceeds zero this indicates a flood volume has exceeded the desired return period (e.g. a flood has not been delivered on time), and the carbon storage dynamics switch to decay. Tree death occurs in periods where the drought counter exceeds the desired return period by a period beyond which tree recovery is not possible, after which point tree death equilibrium is reached.

Altering the magnitude and temporal distribution of hydrological disturbances can cause transition between ecosystem states (Folke et al., 2004). It is therefore hypothesised that the modelled EWH's decisions to reallocate water to improve inundation volumes and frequency can therefore alter the floodplain carbon dynamics. To model this, the EWH has the option to annually reallocate water to the environment in 5% increments up to 50%. Annual reallocation volumes are controlled by altering the variable, $et_{(t,c)}$, controlled by the choice of variable, c which takes the values one to 11. The possible supplementary volume of environmental water available to inundate the floodplain and the exogenously determined irrigation water supply availability prior to any reallocation ($iw_{(t,c)}$) is shown in Equation 6.9 below and elaborated further in Section 6.4.8 with regards to the treatment of foresight.

$$et_{(t,c)} = \left(\frac{c}{20} - 0.05 \right) iw_{(t,c)}$$

(Equation 6.9)

6.5.5.1 *Modelling Maximum and Minimum Carbon Storage Capacities*

The simulation iterative traces the changes in carbon stock in each floodplain zonal area, $A_{(i)}$, for all possible reallocation proportions, c, in each simulation period, t. In absence of primary data it is

assumed that at the start of the simulation ($t=0$) all floodplain River Red Gums are in a non-water stressed condition and contain the maximum potential carbon storage volume[37] ($\mu \pm 1 s.d$):

$$MaxCarbonStorage_{(i,t,c)} = 104.4 \pm 72.1 tCha^{-1}$$ **(Equation 6.10)**

Where $MaxCarbonStorage_{(i,t,c)}$ is the modelled maximum volume of above ground carbon stored per hectare of River Red Gums inundated by floods of flowband, i, in each year, t, for all possible annual reallocation proportions, c. As this is the modelled maximum, at no point during the simulation does the cumulative carbon storage surpass this volume.

Conversely, the minimum volume of carbon stored per hectare of River Red Gums can occur when an area of forest has decayed to a dead tree equilibrium at which point carbon uptake is zero. Absent a monitoring history of River Red Gum condition of sufficient length required to specify the period of time until tree death occurs, a general rule-of-thumb is applied. This rule of thumb is that ten years without flooding and below average rainfall will likely cause River Red Gum mortality of any tree age (Doody, 2014, pers. coms.). The minimum possible volume of carbon stored per hectare of dead above ground River Red Gums is ($\mu \pm 1 s.d$):

$$MinCarbonStorage_{(i,t,c)} = 4.7 \pm 5.4 tCha^{-1}$$ **(Equation 6.11)**

Where $MinCarbonStorage_{(i,t,c)}$ is the modelled minimum volume of above ground carbon stored per hectare of dead River Red Gums in each year, t. The minimum volume stored by dead River Red Gums is somewhat greater than zero because carbon remains trapped in woody biomass and coarse litter of dead trees which are subject to very slow rates of decomposition (Chambers et al., 2000). Note that the variable expressed in Equation 6.11 relates to the stock of carbon in the River Red Gum trees, expressed as tC/ha. This variable does not relate to the annual uptake of carbon, expressed as tC/ha/year, which would reach a value of zero when the tree is dead. In future work, the negative value of carbon uptake (that is, a positive release of carbon into the atmosphere) when a tree is dead due to the decomposition of woody biomass by natural processes would be a worthwhile exploration.

6.5.5.2 Modelling Carbon Storage Decay in Response to Water Deficit

When flooding of a desired volume and timing required to inundate an area of the floodplain is not met, but tree death has not yet occurred, modelled River Red Gum health and carbon stocks exist in a state of decay. River Red Gum carbon stock decay in response water deficit is governed by:

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[37] These assumptions and their necessity were discussed in Section 6.4.2.

$$CarbonDecay_{(i,t,c)} = \exp(vEC_{(i,t,c)})$$ **(Equation 6.12)**

Where $CarbonDecay_{(i,t,c,q)}$ is the annual volume of carbon decay per hectare of River Red Gum in each year without sufficient flooding, v is the decay coefficient, and $EC_{(i,t,c)}$ is the number of sequential periods past the desired return interval for floods in flowband, i. The variable $EC_{(i,t,c)}$ is iteratively updated in each period, t, and can be altered by the choice of reallocation proportion, c. The decay calibration co-efficient is a variable that controls how quickly carbon volume is lost. For River Red Gums it is largely unknown, and so it is must be calibrated to match other known biophysical parameters. In this case, the decay coefficient is calibrated to allow for the decay of carbon stocks from the mean value (~104.4tC/ha) to the value of carbon stocks at tree death (4.7tC/ha). A general rule of thumb of 10 years to cause tree death is applied (Doody, 2014, pers comms.)

As time since the desired return interval $EC_{(i,t,c)}$ increases, the incremental decay in carbon storage capacity increases at an exponential rate; such that the increment of decay in decay period ten is larger than in decay period one. The decay coefficient, v, is calibrated to reflect that after ten years past the desired return interval (i.e. $EC_{(i,t,c)} = 10$) the tree has likely decayed to a minimum rate of annual carbon uptake and tree death has likely occurred, as discussed in the previous section. This occurs for $v = 0.31$. Figure 6.9 demonstrates a conceptual overview of the carbon decay time series from maximum carbon storage to minimum carbon stored in dead trees.

Figure 6.9 Conceptual River Red Gum Carbon Time Series

Source: Own Figure.

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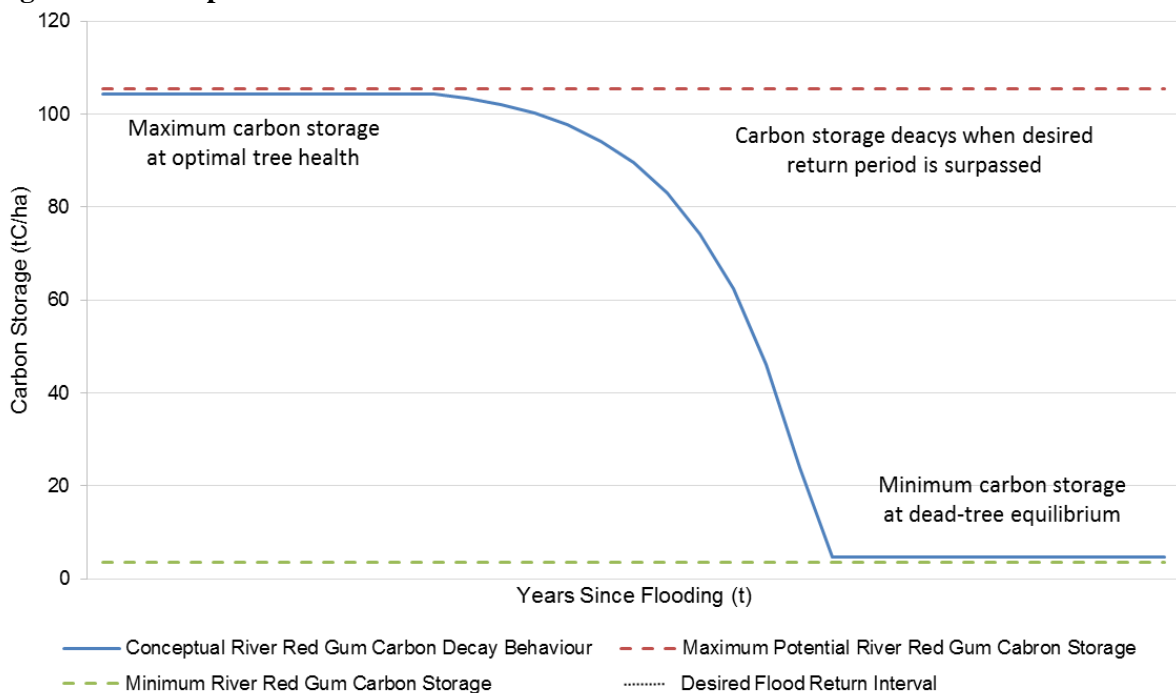
6.5.5.3 *Modelling Carbon Storage Growth in Response to Overbank Flooding*

River Red Gums have considerable capacity for recovery after drought (Doody et al., 2015). To estimate (re-)growth of tree biomass after flooding, the von Bertalanffy-Chapman-Richards (vBCR) forestry function is used (Richards, 1959; Zhao-gang and Feng-ri, 2003). The vBCR has precedence in application to carbon sequestration of native Australian Eucalypt species (e.g. Paterson and Bryan, 2012; Bao and Zhu, 2013). In this study, the vBCR is applied instead of alternative models of tree growth due to its applicability in estimating the change in growth in response to disturbance. Further detail regarding the performance of the vBCR model compared to other forestry growth models, as well as the theoretical derivation of the eight possible cases of the vBCR function, is provided in Zhao-gang and Feng-ri (2003). The generalised integral form of the vBCR function of tree growth is:

$$y_{(t)} = \left[\frac{g}{k} - \left(\left(\frac{g}{k} \right) - yo^{(1-m)} \right) e^{-k(1-m)(t-to)} \right]^{\frac{1}{1-m}}$$ **(Equation 6.13)**

Where $y_{(t)}$ is the increment of annual tree growth, used as a proxy for annual carbon uptake, $t_{(0)}$ and $y_{(0)}$ are initial age and size, respectively, t is the time since planting or disturbance and k, m and n are tree parameters (Zhao-gang and Feng-ri, 2003). A special case of the classic vBCR is the Gompertz equation, which occurs when $m \to 1$, $k > 0$, $n > 0$ and is given by:

$$y_{(t)} = D exp\left(-Fe^{-kt} \right)$$ **(Equation 6.14)**

Where $y_{(t)}$ is the annual increment of tree growth. Based assumption that carbon sequestration is proportional to the tree growth, as discussed in Section 6.4.2, the increment of carbon storage growth is given by:

$$CarbonGrowth_{(i,t,c)} = D \exp(-Fe^{-k.EB_{(i,t,c)}})$$ **(Equation 6.15)**

Where D, F and $k > 0$. F is related to the initial size of the tree and can therefore be thought of as the intercept on the y-axis for the case of $t = 0$ (Zhao-gang and Feng-ri, 2003). k reflects the growth rate of the tree. $EB_{(i,t,c)}$ is the flood counter iteratively summing the number of years that the desired flooding volume and frequencies have been met (i.e. $EB_{(i,t,c)} > 0$ when requirements are met). When $EB_{(i,t,c)} > 0$, tree condition remains in the growth state and the critical threshold into decay has not been passed. In various applications identified in the literature D has been applied as the asymptotic size of the tree reached at $y_{(\infty)}$ or the asymptotic tree growth occurring at $y_{(\infty)}$. In either interpretation of D, the remaining parameters can be calibrated to obtain a growth curve representative of expected forestry dynamics. In the case at hand, D is interpreted to be the asymptotic increment of growth (and

hence carbon uptake) at $y_{(\infty)}$. In the absence of detailed tree growth data for River Red Gums, the growth parameter for a similarly slow-growing Mallee species (*Eucalyptus kochii)* was used, such that *k*=0.06674 (CSIRO, *unpublished data*). Assuming a minimum 15 year period for River Red Gums to reach the maximum volume of carbon able to be stored in the vegetation, the equation was iterated to find $F = 1.39642$. Figure 6.10 provides a conceptual representation of the River Red Gum cumulative sequestration time series.

Figure 6.10 Conceptual River Red Gum Carbon Sequestration Time Series
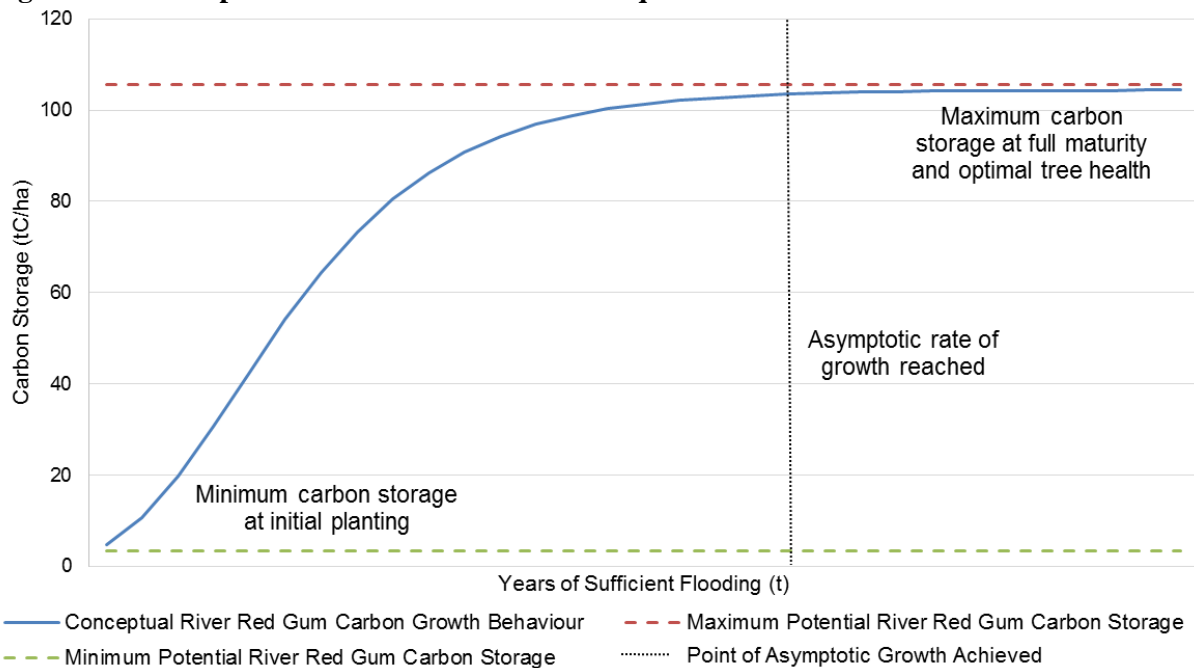
Source: Own Figure.

6.5.5.4 Net Carbon Storage Dynamics

The net carbon stored per hectare of River Red Gums in any period, t, is given as the carbon stored in the previous year, $CarbonStorage_{(i,t-1,c)}$, plus or minus any incremental changes in River Red Gum carbon storage due to overbank flooding or water deficits. The annual net carbon storage is therefore given by:

$$NetCarbonStorage_{(i,t,c)} = CarbonStorage_{(i,t-1,c)} + CarbonGrowth_{(i,t,c)} - CarbonDecay_{(i,t,c)}$$

(Equation 6.16)

Where $NetCarbonStorage_{(i,t,c)}$ is the tonnes of carbon stored per hectare of River Red Gum forest inundated by floods in flowband, i, for each period, t and all possible reallocation proportions, c. $CarbonStorage_{(i,t,c)}$ is a temporary carbon storage accounting variable iteratively updated each period. All other variables have been previously defined. The total volume of carbon stored across the whole floodplain in each period is obtained by summing each hectare of floodplain River Red Gum

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forest, where $w_{(i)}$ is a weight proportional to the area between each floodplain zonal area, $A_{(i)}$, such that total volume of floodplain River Red Gum carbon storage is given as:

$$NetFloodplainCarbonStorage_{(t,c)} = \sum_{i=1}^{8} w_{(i)} NetCarbonStorage_{(i,t,c)} \text{ (Equation 6.17)}$$

6.5.6 Carbon Value

As discussed in Section 6.2.3, the volume of carbon stored in the floodplain River Red Gums is valued using the local market price of carbon and the global social cost of carbon. The monetary value is therefore given by the value per tonne of carbon multiplied by the total volume of carbon stored:

$$CarbonValue_{(t,c)} = CP \sum_{i=1}^{8} w_{(i)} NetCarbonStorage_{(t,c)} \qquad \textbf{(Equation 6.18)}$$

Where CP is the exogenously determined price of carbon in units of Australian dollars per tonne of CO_2 equivalent, assumed to be constant throughout the simulation.

In each period, the marginal benefit of reallocating water to inundate the floodplain is given by the difference in the value of carbon generated by incrementally reallocating water to the environment in a proportion, c, relative to $c-1$. The monetary value of carbon generated by the actions of an EWH annually reallocating water to inundate the floodplain is therefore given by:

$$\Delta CarbonValue_{(t,c)} = CarbonValue_{(t,c)} - CarbonValue_{(t,c-1)} \qquad \textbf{(Equation 6.19)}$$

Where $\Delta CarbonValue_{(t,c)}$ is therefore representative of the environmental opportunity costs of the possible annual reallocation volumes.

6.5.7 Water Allocation Price Model

The water price model is based on Connor et al. (2013) and is obtained by regressing past average annual water allocation prices on allocation levels in the Murrumbidgee catchment from 1996-1997 to 2008-09. The regression model is:

$$Water\,Price_{(t,c)} = \beta_o + \beta_1 log(iw_{(t,c)}) \qquad \textbf{(Equation 6.20)}$$

Where $Water\,Price_{(t,c)}$ is the annual water allocation price per mega litre, $iw_{(t,c)}$ is the past volume of water available for irrigation each year, and β_0 and β_1 are 2,716 and -798, respectively. Although the water price model is simple, consideration of additional rainfall, runoff, commodity prices, or irrigator water trade variables did not further the goal of this study and are thus omitted.

Figure 6.11 shows the observed average annual water allocation and the observed and modelled average annual water price for baseline conditions. As evident in Figure 6.11, there is an outlier in 2006-2007

in the observed water price which is not accounted for in the water price model. This is due to the price spikes observed during the height of the Millennium Drought resulting from high levels of irrigator water trading and national water policy uncertainty which was reflected in allocation water prices.

Figure 6.11 Observed Water Allocations and Observed and Modelled Water Allocation Prices in the Murrumbidgee
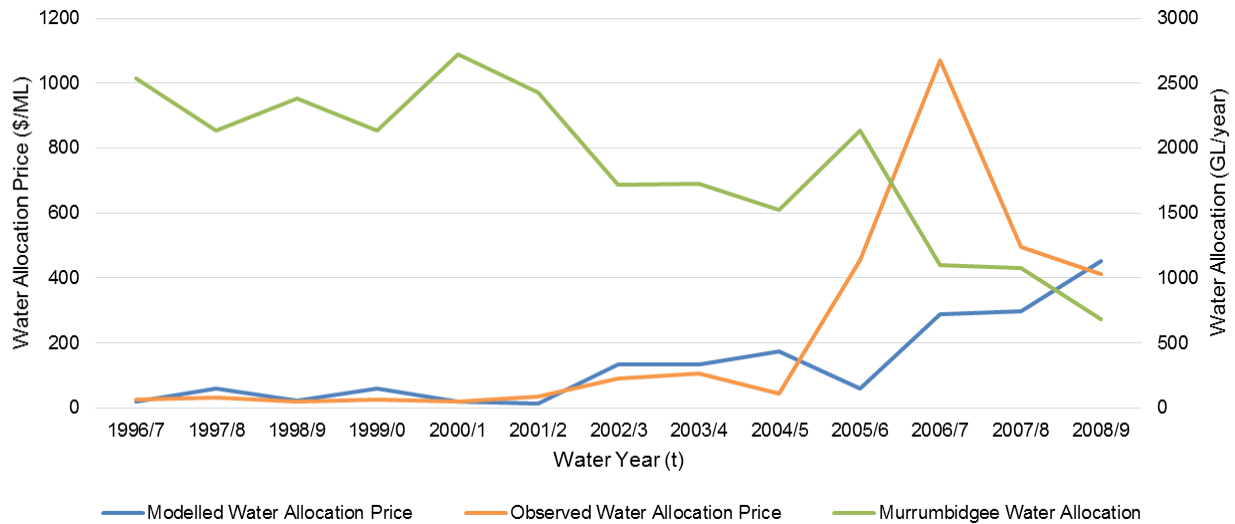
The water price model begins with a baseline irrigation allocation, $iw_{(t,c=1)}$ and iteratively simulates in increments of 5% annual reallocation to the environment. As the annual reallocation increment increases the availability of irrigation water decreases and the price of allocations therefore increase to reflect the increasing scarcity. Including the transfer of water allocations from irrigation to the environment, the water price model is therefore:

$$WP_{(t,c)} = \beta_0 + \beta_1 \log(iw_{(t,c)} - et_{(t,c)})$$ **(Equation 6.21)**

The water price model is used to estimate the cost of irrigation water forgone when water is reallocated to the environment. The volume of reallocated water, $et_{(t,c)}$, has an irrigation opportunity cost $IrrigOC_{(t,c)}$ which is given as the value of the irrigation water foregone that could have otherwise been sold on the water allocation market. As described above, the modelled water price, $Water\,Price_{(t,c)}$, increases incrementally as more water is reallocated to the environment and $IrrigOC_{(t,c)}$ therefore increases as the reallocation volume increases:

$$IrrigOC_{(t,c)} = et_{(t,c)} Water\,Price_{(t,c)}$$ **(Equation 6.22)**

Where $IrrigOC_{(t,c)}$ provides an indication of the cost of irrigation water forgone as water is incrementally reallocated to the environment.

6.5.8 Treatment of Foresight and Water Reallocation Algorithm

6.5.8.1 Treatment of Foresight

Up until this point, the modelled EWH evaluates the costs and benefits of reallocation within a single period only. However, accounting for inter-annual dependencies of environmental health is an important consideration in environmental water decision-making because of the path-dependencies and tipping-points of some ecosystems. Treatment of foresight is also important because of the inter-temporal linkages in irrigated agriculture. On-farm investments, such as infrastructure, property rights and perennial crop plantings, are commonly irreversible (Carey and Zilberman, 2002) or very expensive to remove/undo (Barerenklau and Knapp, 2007). As a result, farm investment decisions are made the basis that the expenditure can be recovered in future time periods (Rowan et al., 2011). Future conditions, such as rainfall and commodity prices, therefore affects the profitability of the farm investment, which subsequently affects further decision making. To account for the dynamics and temporal inter-dependencies in irrigated agricultural decision making, future conditions must be accounted for when modelling decision making in the present time period. Reallocation decisions should therefore be made with the understanding of how reallocating water in the present year will influence ecosystem dynamics and carbon-water trade-offs in the following year. However, the future is not certain and may therefore be treated probabilistically.

A forward-looking EWH concerned with the impact of reallocation decisions in the current period, t, on future simulation periods $t+1$ is modelled. In each period, the modelled EWH is assumed to understand the probabilities of all future state of nature across the decision horizon of $t+n$ years. The decision to reallocate water in each period is made based on the known water inflows, water allocation prices and floodplain carbon stocks in the present period, t, and the expected values of these variable in future time periods.

Accounting for future foresight, the expected value of the changes in carbon storage is therefore given by the sum of the discounted flow of carbon storage benefits (i.e. improvements in carbon stocks) caused by reallocation decisions in the current and future time-periods within the decision horizon. As discussed in Section 6.2.4, expected carbon benefits derived in future periods are discounted by $\delta = 3\%$. The simulation continues to run in iteratively updated decision horizons until the end of the simulation, which occurs at $T = 113$.[38] The present value of the flow of possible carbon benefits generated by annual reallocation is given by:

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[38] The carbon storage benefits generated by reallocation in years near the end of the simulation persist past the final period of the simulation (T=113). To account for this, a long-run analysis was undertaken and the results reported in Section 7.3.1.

$$CarbonValue\,Pr_{(t,c)} = \sum_{t=1}^{T}\sum_{c=1}^{11}\left[\sum_{t=0}^{n}\Delta CarbonValue_{(t,c)}\,\frac{1}{(1+\partial)^n}\right] \quad \textbf{(Equation 6.23)}$$

Where T is the length of the whole simulation, c is the proportion of water to be reallocated from consumptive use to the environment, t is each simulation period, n is the number of years in the decision horizon, δ is the discount rate applied, and $\Delta CarbonValue_{(t,c)}$ is the monetary value of changes in carbon storage, as defined in Equation 6.19.

Similar to carbon, the cost of irrigation water forgone over the decision horizon is given as the sum of the discounted value of water reallocated to the environment for all possible relocation proportions in each year. Accounting for future periods, the cost of irrigation water forgone is:

$$IrricOC\,Pr_{(t,c)} = \sum_{t=1}^{T}\sum_{c=1}^{11}\left[\sum_{t=0}^{n}IrrigOC_{(t,c)}\,\frac{1}{(1+\partial)^n}\right] \quad \textbf{(Equation 6.24)}$$

Where $IrrigOC_{(t,c)}$ is expressed in Equation 6.22 and all other variables are previously defined. As discussed, $CarbonValue\,Pr_{(t,c)}$ and $IrrigOC\,Pr_{(t,c)}$ are simulated for all discrete reallocation proportions, across the hydrological series (T=113) in iteratively updated decision horizons. The decision horizon is chosen to be three years (i.e. $t+n=3$), such that the modelled EWH considers the present year and two years ahead in the future. Sensitivity regarding the years in foresight is investigated and discussed at length in Section 7.7.

6.5.8.2 Reallocation Model

The water reallocation model is a rules-based simulation algorithm which that simulates the decisions of an EWH who has the option to annually reallocate water to maintain and enhance floodplain River Red Gum carbon storage. The modelled EWH is faced with the expected costs and benefits of reallocation over the iteratively updated decision horizon, given by $CarbonValue\,Pr_{(t,c)}$ and $IrrigOC\,Pr_{(t,c)}$ as described in the above section.

Having developed models of both the expected costs and benefits of annual reallocation accounting for periods now and in the future (i.e. $\Delta CarbonValue$ and $IrrigOC\,Pr_{(t,c)}$), the problem therefore becomes one of identifying a discrete annual reallocation volume corresponding to reallocation proportion, c, to generate a flow of $et_{(t,c)}$, for which the present value of the flow of carbon benefits generated by overbank flooding equals or exceeds the present value of the costs of reallocation. Modelled reallocation occurs at the point of marginal equilibrium, h, within an acceptable level of tolerance, j, which is specified as the observed long-term average water allocation price of 5GL.

6.6 Treatment of Uncertainty[39]

Any model is an abstraction of reality and incompletely describes the natural system and uncertainty is therefore inherent in any model. This thesis adopts the definition of uncertainty proposed by Walker et al. (2003) and is depicted in Figure 6.12.

Figure 6.12 Defining Uncertainty in Modelling.
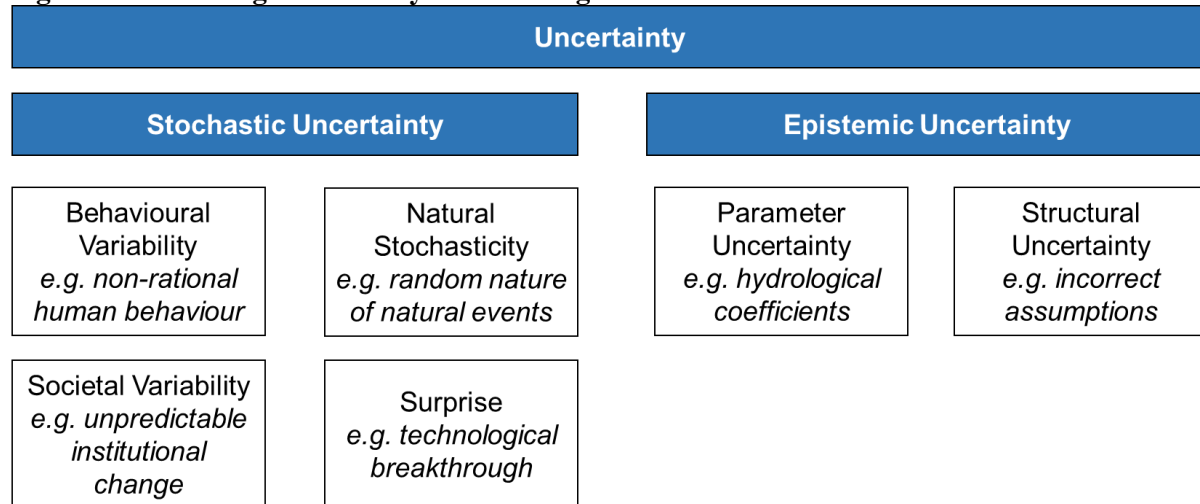
Source: Recreated from Settre et al. (2017), adapted after Walker et al., (2003), pp. 13-14.

Epistemic (i.e. knowledge) uncertainty arises from incomplete or erroneous knowledge or data leading to uncertain parameter values and model structure. Stochastic uncertainty arises from the inherent spatial and temporal variability of the natural world, including the climate, society and human behaviour. Stochastic uncertainty can be defined either measurable (e.g. defined by a statistical distribution) or unmeasurable where there is unpredictability of future events. In the context of HEM, epistemic uncertainty manifests as uncertainty inherent in the model, such as parameter uncertainty resulting from limited data available for calibration or errors in model structure. Model structure uncertainty describes the uncertainty of causal relationships between variables, between inputs and variables, functional form of the model itself, model boundaries and assumptions (Walker et al., 2003). Epistemic uncertainty and stochastic uncertainty are reducible and irreducible, respectively (Matott et al., 2009).

In cases with high epistemic uncertainty, neglecting the compounding effect of parameter and structure uncertainty can lead to the recommendation of different management programs with follow-on implications for various costs and benefits of alternative scenarios (Pena-Harro et al., 2011). The treatment of reducible epistemic uncertainty is therefore motivated by the need to minimise the risk of a sub-optimal decision informed by model results (Matott et al., 2009). Additionally, addressing

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stochastic uncertainty aids in the design of a policy robust to future variability (Hession and Storm, 2000).

6.6.1 Sensitivity Testing Methodology

To identify and address uncertainty in the HEM, a two-stage methodology was used. First, an uncertainty matrix was developed following the method presented by Walker et al. (2003), which provides a heuristic tool to identify the type, location and impact of various sources of uncertainty within the model. An uncertainty matrix framework is shown in Table 6.11 to demonstrate this method.[40]

Table 6.12 Uncertainty Matrix Framework.

Location	Level				Nature	
Context	Statistical uncertainty	Scenario uncertainty	Recognised ignorance	Total ignorance	Epistemic	Stochastic
Inputs						
Parameter						
SUM						

<u>Source:</u> **Walker et al. (2003)**

Based on the results of the populated uncertainty matrix, the variables identified as having the highest likelihood and impact on the modelling results were prioritised to be sensitivity tested. The sensitivity analysis (SA) was carried out for the identified parameters in a simple manner using high and low values to address the model robustness to uncertainty in input values. The SA procedure involved altering the values of input parameters previously identified in the uncertainty matrix, holding all other variables constant and evaluating the change in outputs. Note that due to the highly integrated nature of the HEM and the large number of variables and parameters, it was not feasible to undertake a sensitivity test on all parameters. In the current treatment of uncertainty, the stochastic uncertainty of climatic variables is not addressed. Instead, to account for the influence of possible climate impacts on inflow volume (which is the driving hydrological variable in the model presented), a scenario analysis was undertaken for moderate and extreme climate change scenarios informed by CSIRO Sustainable Yields results for the Murrumbidgee (CSIRO, 2008c). These results are reported alongside sensitivity analysis results in Section 7.7. Future work regarding the inclusion of stochastic inflows and ecological parameters is discussed in Section 9.4.

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[40] In Table 6.11 statistical uncertainty refers to uncertainty that can be described in statistical terms (e.g. a probability distribution of an uncertain parameter); scenario uncertainty refers to the uncertainty of possible eventualities in the external environment which influences the model (e.g. median, dry, extreme dry climate change scenarios); recognised ignorance refers to the case where there is a fundamental lack of understanding about a process or structure to the point that there is insufficient knowledge to construct possible scenarios (e.g. government policy in 200 years' time); total ignorance refers to the things we do not know we do not know and therefore collectively fail to consider or anticipate (e.g. black swan events). Each level of uncertainty increases in ascending order from determinism (e.g. fully understood and mathematically describable system) to indeterminism (e.g. completely unknown and unable to be described mathematically).

6.7 Conclusion

This chapter has presented the development of a HEM constructed to investigate the possibility of reallocating water to the environment to inundate floodplains and generate tradable carbon credits through improved carbon uptake in River Red Gum floodplain forests. This chapter has detailed each aspect of the model contains representation of hydrology and overbank flooding, carbon dynamics, carbon valuation, water prices, probabilistic foresight and water reallocation decisions. An overview of the uncertainty methodology has also been presented. The data sets and assumptions used have also been detailed and discussed. The chapter situates the model development within a description of the case study to which it is applied, focusing on the existing water market and the potential carbon market which the model simulates. This chapter has also provided an overview of the economic foundations of the hydro-economic modelling and ecosystem service modelling approach, including an introduction to the ecosystem service classification, ES cascade, valuation of ecosystem services and flow-dependent ecosystem service generation. Overall, this chapter has provided a theoretical foundation of ecosystem service and hydro-economic modelling, as well as a detailed description of the model used to generate the results presented in the following chapter.

CHAPTER 7: Carbon and Water Market Opportunities in Environmental Water Reallocation: Results of the Hydro-economic Model

This chapter is a combination of two expanded working papers: '*Can the generation and sale of carbon credits in natively vegetated floodplains offset the costs of annual environmental water purchases?*' and *'Strategic water and carbon market opportunities for a self-financed environment water holder to improve floodplain forest carbon storage'*. The working papers focus on the potential opportunity for overbank flooding to generate biomass growth and carbon storage of sufficient value to offset the cost of water required to create the flooding. Based on the model results a carbon-water trading strategy is proposed and the optimal temporal opportunities to utilise the proposed strategy to finance environmental water purchases is analysed, as well as a discussion of the possible policy implications and challenges. The versions of the working papers included in this thesis present the results only and model theory and methodology were discussed previously in Chapter 6.

Overall, it is suggested that a carbon-water trading strategy may present one avenue to offset, or at least reduce, the cost of an EWH purchasing supplementary environmental water to achieve positive ecological benefits from overbank flooding.

Publication Details

Publication Title	Can the generation and sale of carbon credits in natively vegetated floodplains offset the cost of annual environmental water purchases?
Publication Status	☐Published ☐Accepted for publication ☐Submitted for publication ☒Unpublished and unsubmitted work written in manuscript style
Publication Details	N/A

Principal Author

Name of Principal Author (Candidate)	Claire Settre		
Contribution to the Paper	Ecological and carbon model development, calibration and validation, re-allocation model development, economic valuation, sensitivity and uncertainty testing, model verification, model coding in GAMS, paper writing and editing.		
Overall percentage (%)	80%		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	28/8/2017

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

- the candidate's stated contribution to the publication is accurate (as detailed above);

- permission is granted for the candidate in include the publication in the thesis; and

- the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Jeff Connor		
Contribution to the Paper	Guidance in ecological and carbon modelling, water price model, assistance in paper structure, writing and editing.		
Signature		Date	21/8/2017
Name of Co-Author	Sarah Wheeler		
Contribution to the Paper	Assistance in paper structure, data provision, discussion, editing and revisions for publication.		
Signature		Date	21/8/17

Publication Details

Publication Title	Strategic annual carbon-water trading opportunities for a self-financing environmental water holder to improve floodplain forest carbon storage.
Publication Status	☐Published ☐Accepted for publication ☐Submitted for publication ☒Unpublished and unsubmitted work written in manuscript style
Publication Details	N/A

Principal Author

Name of Principal Author (Candidate)	Claire Settre		
Contribution to the Paper	Carbon model development, calibration, re-allocation model development, economic valuation of carbon storage, sensitivity and uncertainty testing, model verification, model coding in GAMS, paper writing and editing.		
Overall percentage (%)	80%		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	28/8/2017

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

- the candidate's stated contribution to the publication is accurate (as detailed above);
- permission is granted for the candidate in include the publication in the thesis; and
- the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Jeff Connor		
Contribution to the Paper	Guidance in ecological and carbon modelling, water price model, assistance in paper structure, writing and editing.		
Signature		Date	21/8/2017
Name of Co-Author	Sarah Wheeler		
Contribution to the Paper	Assistance in paper structure, data provision, discussion editing and revisions.		
Signature		Date	21/8/17

7.1 Introduction

Balancing consumptive and environmental water-use trade-offs is an ongoing challenge in water scarce regions. In some settings, this challenge has been managed using strategic water market transactions to buy water from irrigation farmers and reallocate it to the environment. In the MDB there are possibilities to buy supplemental environmental flow in an active low transaction cost water allocation market, as discussed at length in Chapter 3. In addition, there may be possibilities to sell carbon sequestration credits generated by improved floodplain forest conditions, caused by increased volumes and frequency of overbank flooding.

This chapter applies a HEM (as detailed fully in Chapter 6) to investigate the prospect of whether reallocating water for floodplain River Red Gum forest recovery and maintenance in an MDB catchment can generate carbon credits of sufficient value to offset the market cost of water. The HEM includes a representation of river flows, floodplain inundation, forest carbon dynamics, carbon credit value and consumptive water-use opportunity costs. The model is applied to the Murrumbidgee catchment in the MDB, where there is a functioning water market and a potential carbon market.

The results are presented in the following format: firstly, the baseline carbon storage capacity of the floodplain River Red Gums is analysed and valued. The following section details the possibility of offsetting supplementary annual environmental water purchases through the generation of carbon credits via floodplain tree growth caused by overbank inundation. Assuming a forward-looking EWH, it is shown that opportunities to implement the proposed carbon-water trading strategy occur when periods of ecological change (such as early decay, drought recovery or natural flood response) coincide with moderate to low water allocation prices. Within these periods, it may be possible to offset, or at least reduce, the cost of strategic environmental water purchases under certain economic and hydrological conditions, subject to a range of sensitivities and uncertainties. The final results section compares the modelling results to previous models, to determine the benefit of including a more representative model of environmental dynamics. Lastly, the results of the sensitivity analysis are presented.

The chapter concludes with a detailed discussion of the potential business case, which can be employed by EWHs to finance environmental water purchases. It is shown that implementation of the proposed carbon-water trading strategy is subject to a number of uncertainties and policy challenges which would need to be overcome to realise this opportunity, including methods for integrating carbon storage benefits from existing national parks into national carbon accounting schemes.

7.2 Baseline River Red Gum Carbon Dynamics and Economic Value

The model is run under baseline conditions for the purpose of: a) establishing a basis for comparison of reallocation results; and b) comparing modelled forestry dynamics to observed floodplain behaviour.

Under baseline conditions, modelled environmental water availability does not meet the desired environmental watering requirements across all floodplain zonal areas. The proportion of time environmental watering requirements are not met varies spatially across the eight floodplain zonal areas used in the modelling. These floodplain zonal areas represent floodplain heterogeneity with each requiring inundation on different frequencies, and differing flow levels to induce inundation, as shown in Figure 7.1. Environmental watering requirements in floodplain areas seven and eight (inundated by large floods of >1,700GL and >2,700GL, respectively) are rarely met under baseline conditions, due to very high volume floods required to inundate these large areas and higher elevation zones, as depicted by the area graph in Figure 7.1. This model result is consistent with existing literature about the Murrumbidgee catchment, which found river regulation and high levels of water abstraction have reduced the volumes available to inundate broad floodplain areas at higher elevations, ultimately resulting in the shrinking of the Lowbidgee floodplain area over time (Kingsford, 2003).

Flowband one is an exception to the general finding that environmental flow requirements are less frequently met in higher elevation zones. Results show that the frequency of not meeting desired environmental watering for flowband one exceeds that for flowbands two to five (Figure 7.1). This is because flowband one (50GL) has a very short desired return period of less than one year (1:0.95), meaning that all years with less than 50GL flow to the floodplain are counted as a year in which the desired return period is surpassed. This occurs in 14 years throughout the 113 year simulation, primarily in the Federation Drought, the 1980-1982 Drought and the Millennium Drought. Whilst flowbands two-five have larger flood volume requirements, they are required less frequently and these requirements are met more frequently under baseline conditions.

In periods where the environmental watering requirements are not met, modelled carbon decay dynamics occur. Figure 7.2 depicts the modelled trend in River Red Gum carbon storage and environmental water availability for the simulation period under baseline (no reallocation) conditions. It is evident that carbon decay dynamics are particularly pronounced in the Federation Drought (1895-1903) and Millennium Drought (1997-2010) periods, where the annual rate of River Red Gum carbon sequestration declines towards around half of initial levels, due to reduced water availability and a subsequent increase in tree water stress. Modelled carbon dynamics for baseline conditions are consistent with field observations of significant floodplain damage during drought (MDBC, 2003, Kingsford, 2003; Cunningham et al., 2010) and international evidence of reduced carbon assimilation rates during prolonged water scarcity periods (Gatti et al., 2014).

Figure 7.1 Proportion of Time Desired Return Period for Flooding per Floodplain Area is Exceeded for Baseline Conditions
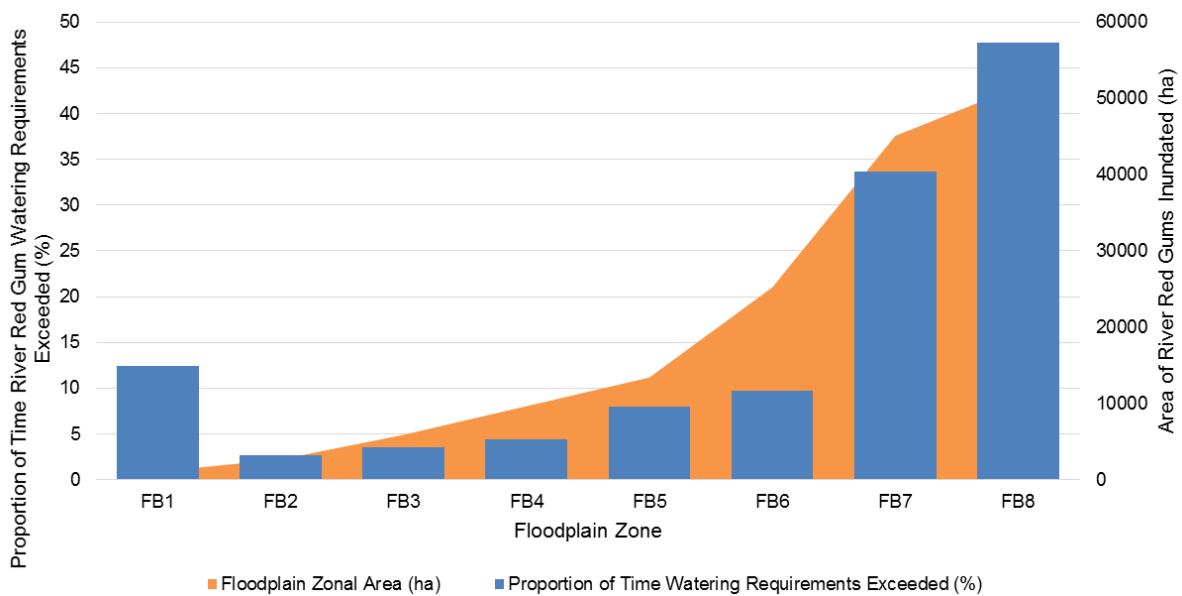
Source: Own Figure.

Figure 7.2 Modelled Environmental Water Availability and River Red Gum Carbon Storage Trend for Baseline Conditions in the Lowbidgee
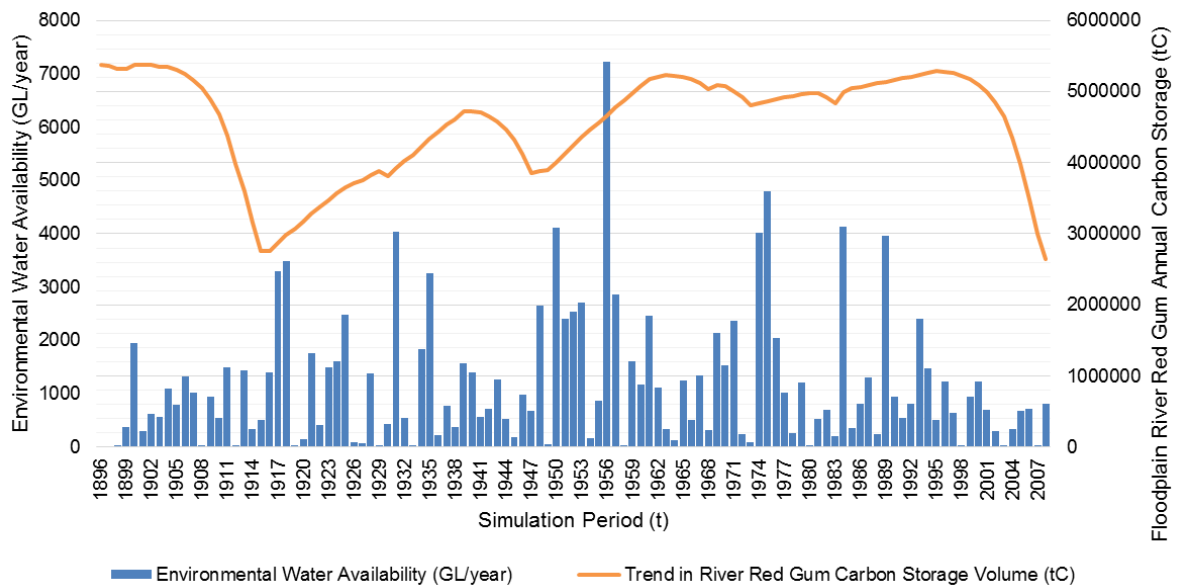
Source: Own Figure.

Under baseline conditions, including periods of decay during drought, the floodplain River Red Gum forest has an annual mean carbon storage capacity of approximately 4.5 million tC. This corresponds to a baseline economic value of between $45 and $218 million in carbon credits for a carbon price of $10 to $48/tCO$_{2e}$, respectively.[41] This value is substantially less than would occur if floodplain watering

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[41] $10/tCO$_2$ is the lowest price from the 2015-2017 Australian carbon credit actions and $48/tCO$_2$ is the middle ground estimate of social cost of carbon in Australian dollars in 2015.

requirements were fully satisfied. For example, when environmental watering conditions are fully satisfied, the maximum value of carbon storage on the floodplain each year, assuming a mean carbon storage capacity of $104.4tCha^{-1}$ of carbon per hectare River Red Gums (Smith and Reid, 2013), is between $53 and $258 million dollars for carbon prices between $10-$48/tCO_{2e}, respectively. This value is generated only when floodplain environmental water requirements are fully met (i.e. the proportion of time the desired return period exceeded tends to zero for all floodplain zonal areas).

7.3 The Impact of Annual Reallocation to the Environment

River Red Gum condition and carbon storage capacity can be improved by strategically reallocating water when it is economically justifiable to do so. It is found that for carbon prices greater than approximately $30/$tCO_{2e}$ it may be economically justifiable to purchase annual water allocations and reallocate these to the environment, in order to generate River Red Gum carbon storage benefits of value sufficient to justify the costs. This opportunity exists for the range of modelled water prices spanning approximately $20-73/ML[42], which occur primarily in periods of low water scarcity (i.e. water abundance). A more detailed exploration of the conditions for which it is economically justifiable to reallocate water to the environment to generate carbon credits of sufficient volume and offset the costs of water allocation purchases is discussed in Section 7.4. The following section details the impacts of reallocation on water availability and subsequent changes in carbon storage capacity.

Improvements in carbon storage occur due to a reduction in the proportion of time the floodplain forest experiences decay dynamics. The influence of annual trading on reducing the proportion of time the desired flooding intervals are exceeded is shown in Figure 7.3, for a reallocation strategy informed by a carbon price of $30/$tCO_{2e}$ and a decision horizon of three years[43].

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[42] Note that in the model, the water allocation price is restricted by a minimum of $20/ML. In reality, it is possible that water allocation prices drop below this price, at which it will still be economically justifiable (in fact, it will be more so) to purchase water allocations for the environment.

[43] The choice of carbon prices and the number of years in foresight are described in depth in Chapter 6.

Figure 7.3 Proportion of Time the Desired Return Period for Flooding per Floodplain Area is exceeded for Baseline and Optimal Reallocation Conditions
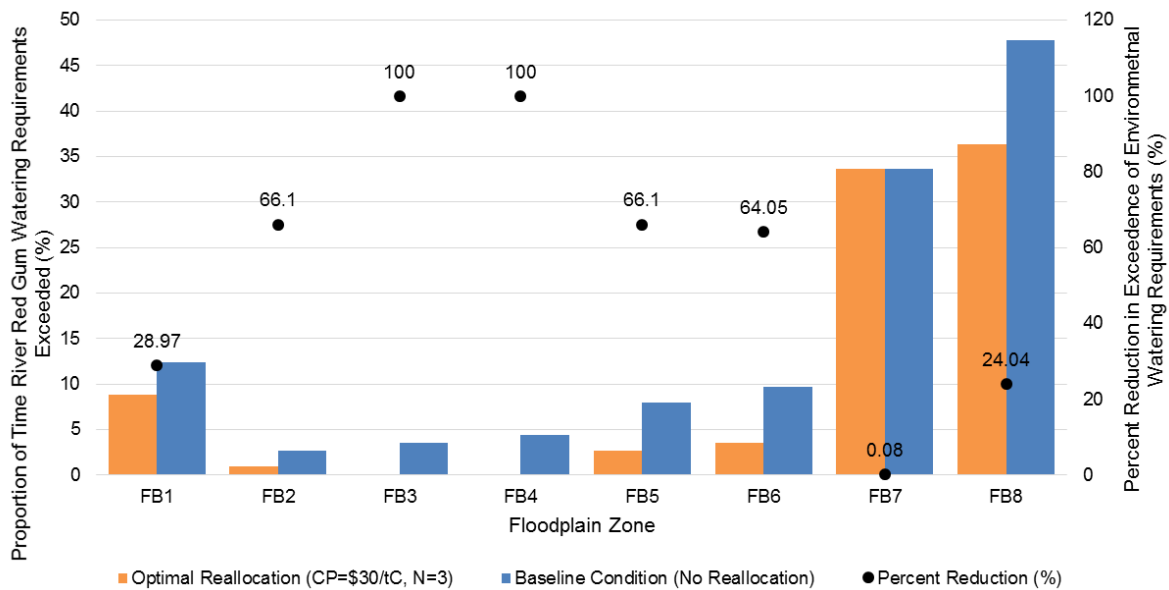
As can be seen, annual reallocation has a particularly pronounced impact on floodplain zonal areas one to six, which are inundated by flows of 50GL/year to 800GL/year. The deficiency of flooding in flowband one is rectified by only small volumes of permanent reallocations greater than 50GL each year (MDBA, 2012b).

Annual reallocation, even when incentivised by a higher carbon price as explored in Section 7.6.1, has comparatively limited ability to improve the flooding regime for the large, higher elevation floodplain zonal areas inundated by floods of 1,700GL and 2,700GL. This result is due to the prohibitive cost of acquiring the required volume of water to create these flows. The exception to this result is isolated opportunities to extend floodplain inundation during naturally occurring very high flood events, such as that which occurred in the 1950s. During such periods, the price of allocation water is low due to water surpluses, and there may be advantages to reallocating water to extend inundation if River Red Gum carbon stocks are sub-optimal leading up to the flooding event. This increase in maximum flood volume to inundate zonal area eight (which is the largest and requires the highest volume of reallocation for inundation) is shown in the descriptive statistics and in the hydrograph in Figure 7.4. This figure shows the resulting environmental water availability over the simulation for the optimal reallocation relative to baseline conditions. As shown in Figure 7.2, the mean environmental water availability for baseline conditions is 1,175.6GL/year with an annual peak flow of 7,230GL and a minimum of zero.

Figure 7.4 illustrates the ability for the annual reallocation strategy to subtly alter the environmental water hydrograph by reallocating water in a pattern, which serves to prevent drought conditions in isolated low flow years and extend some moderate floods peaks. This is discussed at length in Section 7.5 of this chapter. From the perspective of consumptive water availability, there is a moderate decrease

in high availability years and a small isolated decrease during the final drought period of the simulation, as shown in Figure 7.5. The resultant effect on water price is discussed in Section 7.3.2.

Figure 7.4 Modelled Environmental Water Availability for Baseline and Optimal Annual Reallocation Scenario
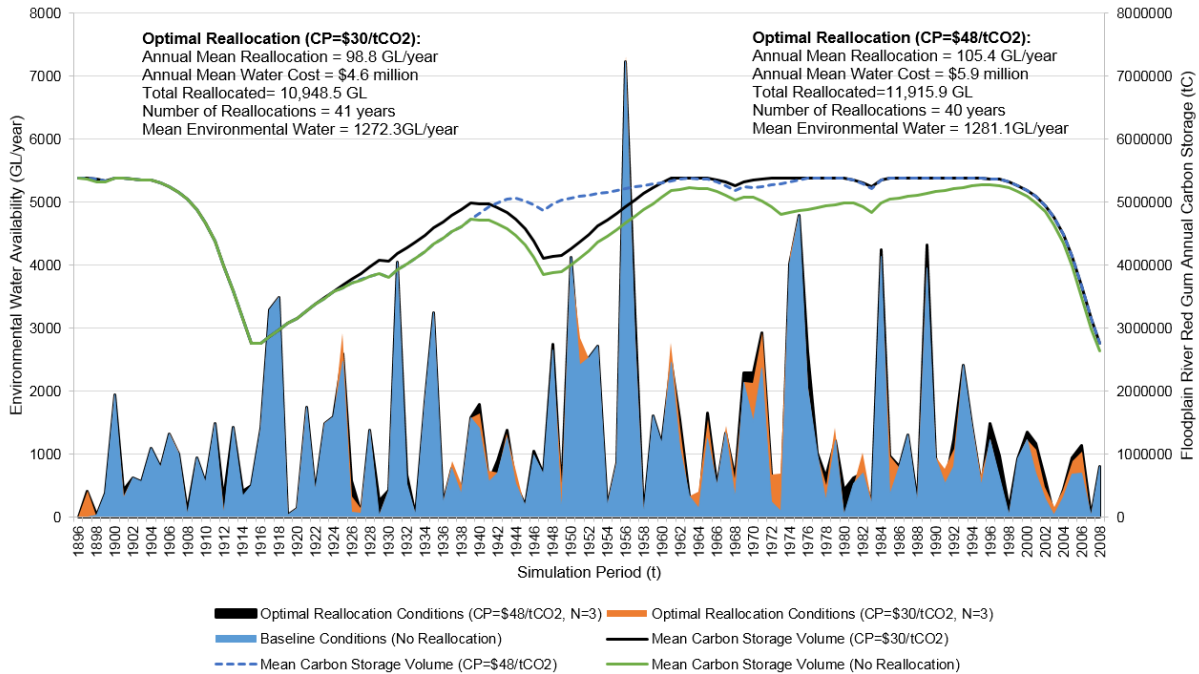
Source: Own Figure.

Figure 7.5 Modelled Consumptive Water Availability for Baseline and Optimal Annual Reallocation Conditions
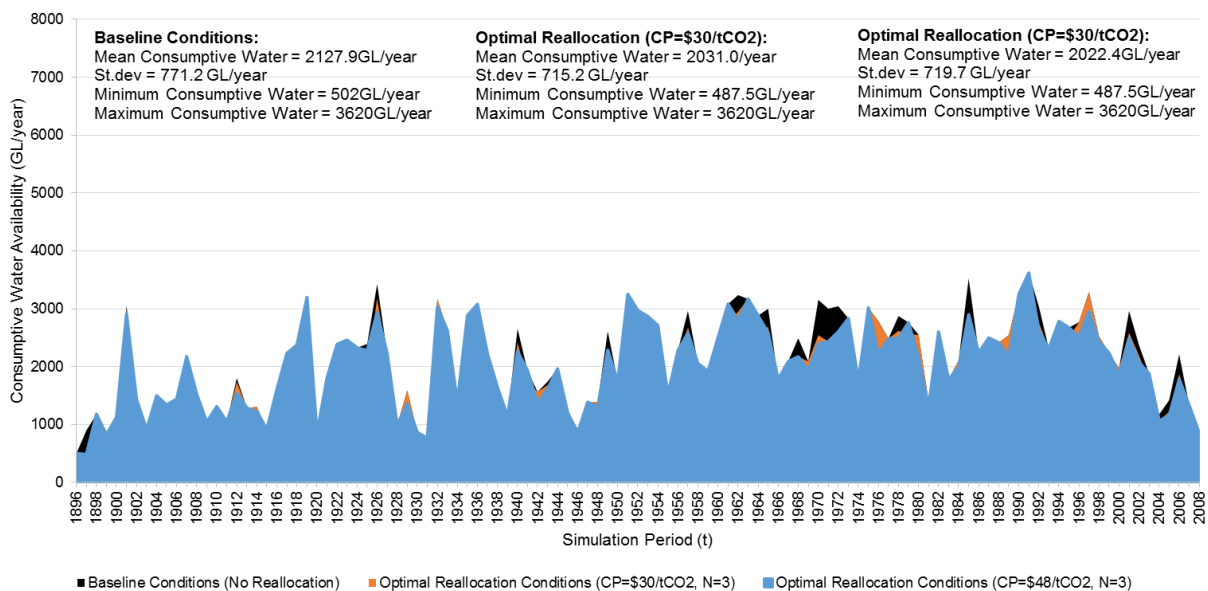
Source: Own Figure.

As hypothesised, improvements in environmental water availability resulting from annual EWH trading activities have the ability to affect the modelled River Red Gum carbon storage dynamics. Carbon storage behaviour, for baseline and optimal reallocation scenario, is shown in Figure 7.6. As shown in

Figure 7.6, annually reallocating water to the environment when it is economically justifiable to do so has no impact on the maximum volume of carbon stored across the floodplain, but does increase the mean and minimum carbon volumes. Over the length of the simulation, the percentage change in carbon stocks relative to the baseline averages 4% and is highest in the simulation periods corresponding to the 1970s, during which time tree health is maintained as a result of EWH trading relative to the baseline, which moderately decays leading into the 1980-1982 drought period.

Figure 7.6 Modelled River Red Gum Carbon Storage Dynamics for Baseline and Optimal Reallocation Scenario
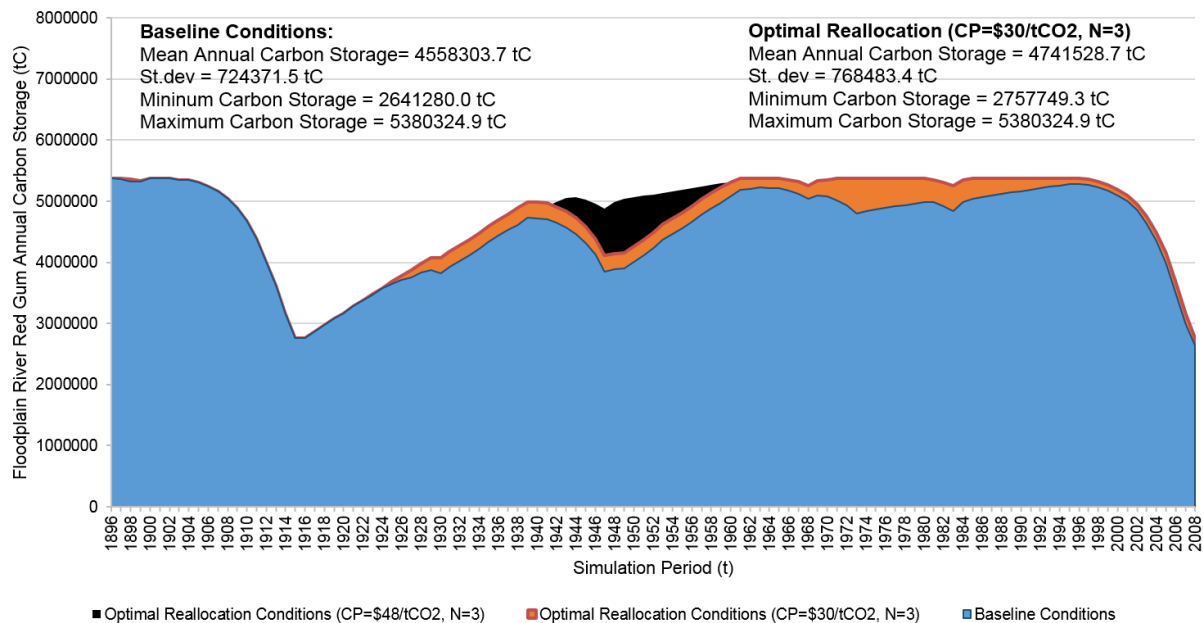
Source: Own Figure.

7.3.1   Long-run Impacts of Annual Reallocation

Carbon sequestration is cumulative in growing forests. Changes in carbon storage behaviour caused by reallocation therefore affects the ecosystem trajectory for future years, and the benefit of improved carbon storage persists beyond the simulation horizon. To account for the long-run benefits, the model was re-run over the historical time-series, using the final carbon storage capacity as the input for the initial storage capacity, for both baseline and annual reallocation conditions. The long-run analysis was run for a total of 226 years (double the initial simulation).

Figure 7.7 demonstrates the long-run impacts of reallocation on the carbon storage capacity, for a reallocation strategy informed by a carbon price of $30/tCO$_{2e}$ and a decision horizon of three years. The long-run (T=226) improvement in carbon storage capacity due to reallocation is marginally larger (~2.2% greater) than the carbon benefit observed during original time-series (T=113) due to persistence of benefits past the original simulation horizon. However, over an extended period of time the River Red Gum carbon sequestration behaviour tended towards the trend observable under baseline conditions, and no difference in annual carbon storage capacity is evident past the 133rd simulation

period. The relative lack of significant difference in the short-term and long-term trends in modelled carbon storage behaviour is due to the presence of significant drought conditions at both the beginning and end of the historical hydrological time-series, which is repeated in the long-run analysis. This result may differ if the model was re-run over a randomly generated sequence of hydrological conditions.

Figure 7.7 The Long-run Impacts of Annual Reallocation on Mean River Red Gum Carbon Storage Behaviour for Baseline and Reallocation Scenario (CP=$30/tCO$_{2e}$, N=3)
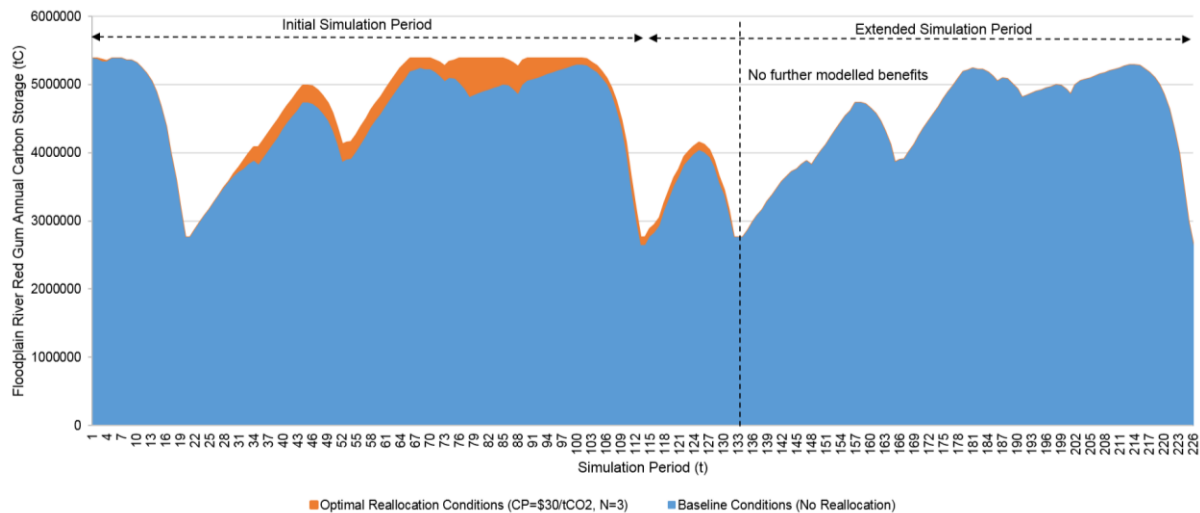
Source: Own Figure.

7.3.2   Annual Reallocation Impact on Water Price

As previously discussed, an EWH actively trading in annual water allocations may affect the water market by bidding up or bidding down the price (Loch et al., 2012). The model results indicate that reallocation occurs in relatively low or moderate irrigator opportunity costs, corresponding to modelled low water price periods in the range of $20 to $73/ML[44], as discussed above. A result is that annually reallocating water to the environment has only a marginal impact on the long-term average modelled water allocation price. However, there is a moderate effect on the maximum water price and an increase in water price variability over the simulation. The impacts on water price are relatively insensitive to the changes in reallocation strategy caused by altering the carbon price, because the frequency of reallocation does not change significantly when carbon prices change, as shown in the results in Section 7.7.1. The water price for the baseline and optimal reallocation scenarios are shown in Figure 7.8. Overall it is found that water price is not significantly affected by the reallocation strategy informed by the HEM. This confirms existing literature suggesting actively trading EWHs are likely to have relatively small impact on water allocation prices (Kirby et al., 2006, Connor et al., 2013).

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[44] Note that as previously discussed, the proposed strategy would also be viable for water allocation prices lower than $20/ML. However the model restricts water allocation price minimum at $20/ML.

Figure 7.8 Modelled Impact on Water Allocation Price

Baseline Conditions (No Reallocation):
Mean Water Allocation Price= $114.3/ML
St.dev = $122
Min = $560.8/ML
Max = $20/ML

Optimal Reallocation (CP=$30/tCO2):
Mean Water Allocation Price= $121.5/ML
St. dev = $126.5/ML
Min = $20/ML
Max =$570.9

Optimal Reallocation (CP=$48/tCO2):
Mean Water Allocation Price = $122.8/ML
St. dev = $128.2/ML
Min = $20/ML
Max = $570/ML
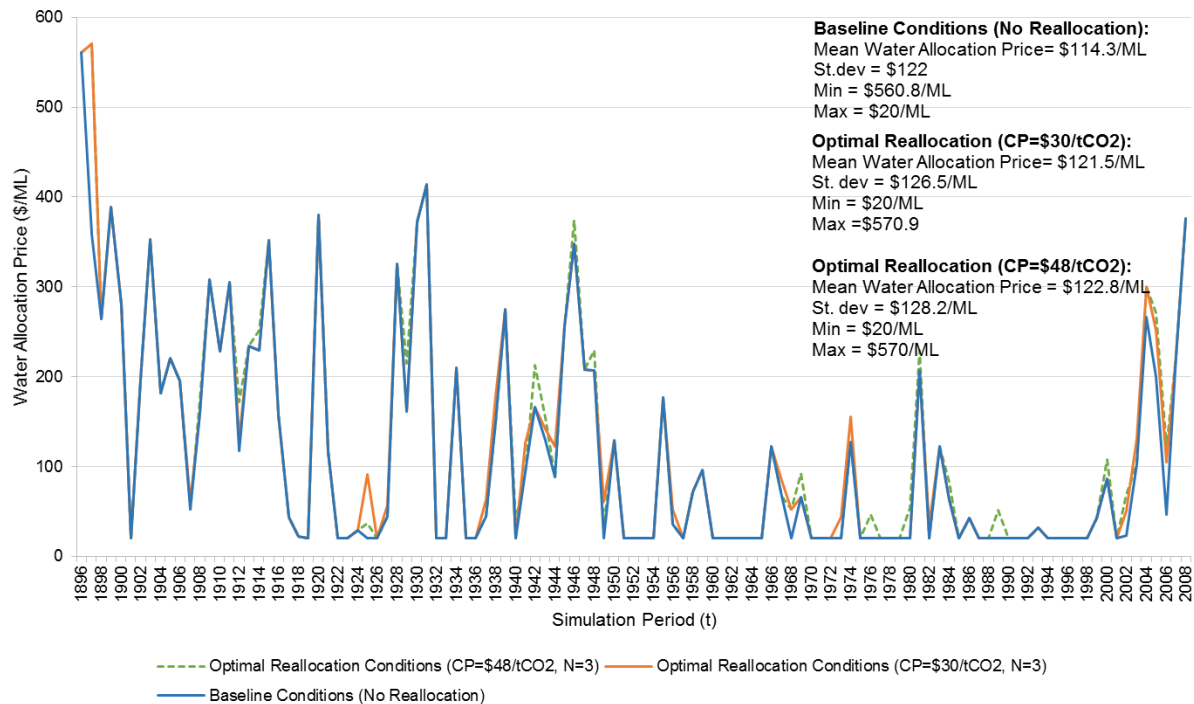
Source: Own Figure.

7.4 Potential to Offset Water Allocation Purchases with Carbon Credit Generation and Sale

The annual improvements in River Red Gum carbon storage caused by annual reallocations to the environment are shown in Figure 7.6, which represents carbon that has either been sequestered or prevented from decay that would not otherwise occur under baseline conditions. The increase in carbon therefore has the potential to be sold if a market for carbon credits was active. As shown above, results indicate that under certain conditions it can be economically justifiable to reallocate water to the environment to generate carbon credits. This implies there may be opportunities where the value of improved River Red Gum carbon storage, caused by increases in the volume and frequency of the overbank flooding regime, to be able to offset the cost of water allocation purchases required to generate the flooding.

It is found that such an opportunity may exist for carbon prices greater than approximately $30/tCO_{2e}$. Carbon storage improvements valued at lower carbon prices may not fully offset the cost of water purchases, but may be used to subsidise the overall cost of water allocation purchases. For the economically justified annual reallocation strategy in response to a carbon price of $30/tCO_{2e}$, the annual mean modelled carbon storage increases River Red Gum carbon stock by an average of 183,225tC (~4% increase from a mean of 4,558,304tC to 4,741,529tC). This results in the generation of an additional $5.4 million of annual carbon storage benefits, which could be achieved with an average annual water purchase expenditure of $4.6 million. If the modelled EWH was to consider alternative carbon values, such as the global social cost when weighing reallocation decisions, an increase of an average of 218,962tC (~5% increase from a mean of 4,558,304tC to 4,777,266tC) would be generated

by strategic water reallocation, resulting in an annual increased flow of carbon storage benefits of $10.5 million achieved at an average annual cost of $5.9 million. A summary of change in carbon storage volume, the overall (total) carbon storage benefits, and total expenditure over the simulation period are shown in Table 7.1 for two carbon prices: a) the minimum market carbon price for which reallocation costs can be offset; and b) the social cost of carbon. These results are presented graphically in Figure 7.9.

Table 7.1 Modelled Water Allocation Purchase Costs and Carbon Credit Benefits

		Total Volume of Allocations Purchased over Simulation (GL)	Total Water Allocation Purchase Cost ($ million) *[cost]*	Value of Carbon Stored in Floodplain River Red Gums ($ million)[45]	Carbon Value Improvement from Baseline ($ million) *[benefit]*	Net Economic Benefit ($ million) *[benefit –cost]*
Carbon Price ($/tCO₂e)						
	30	10,948.4	526.9	16,073.7	621.1	94.2
	48	11,915.9	667.1	25,911.8	1,187	520.7

Source: Own Table.

The increased value of carbon storage, evident in Figure 7.6 and Figure 7.9, relative to the baseline is predominately derived by improved River Red Gum condition in the lower elevation floodplain areas (flowband zones one to six), which require smaller reallocation volumes to achieve flooding. Marginal improvements in watering conditions are evident in higher elevation floodplain areas (flowband zones seven and eight) due to opportunities to extend naturally high flooding events. Note that due to modelling assumptions, limiting the maximum River Red Gum carbon storage capacity to the observed mean carbon capacity per hectare, as well as omitting other floodplain carbon storage components, the modelled increase in carbon storage and value are likely a significant underestimate of the overall carbon benefits generated by the reallocation strategy.

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[45] Note that under baseline conditions (no reallocation to the environment) the forest has an economic value given by the market value of the total volume of carbon stored in the forest, absent any supplementary environmental water availability.

Figure 7.9 Mean Carbon Storage Volumes, Water Costs, and Carbon Storage Benefits for Baseline Conditions and Optimal Reallocation Scenarios
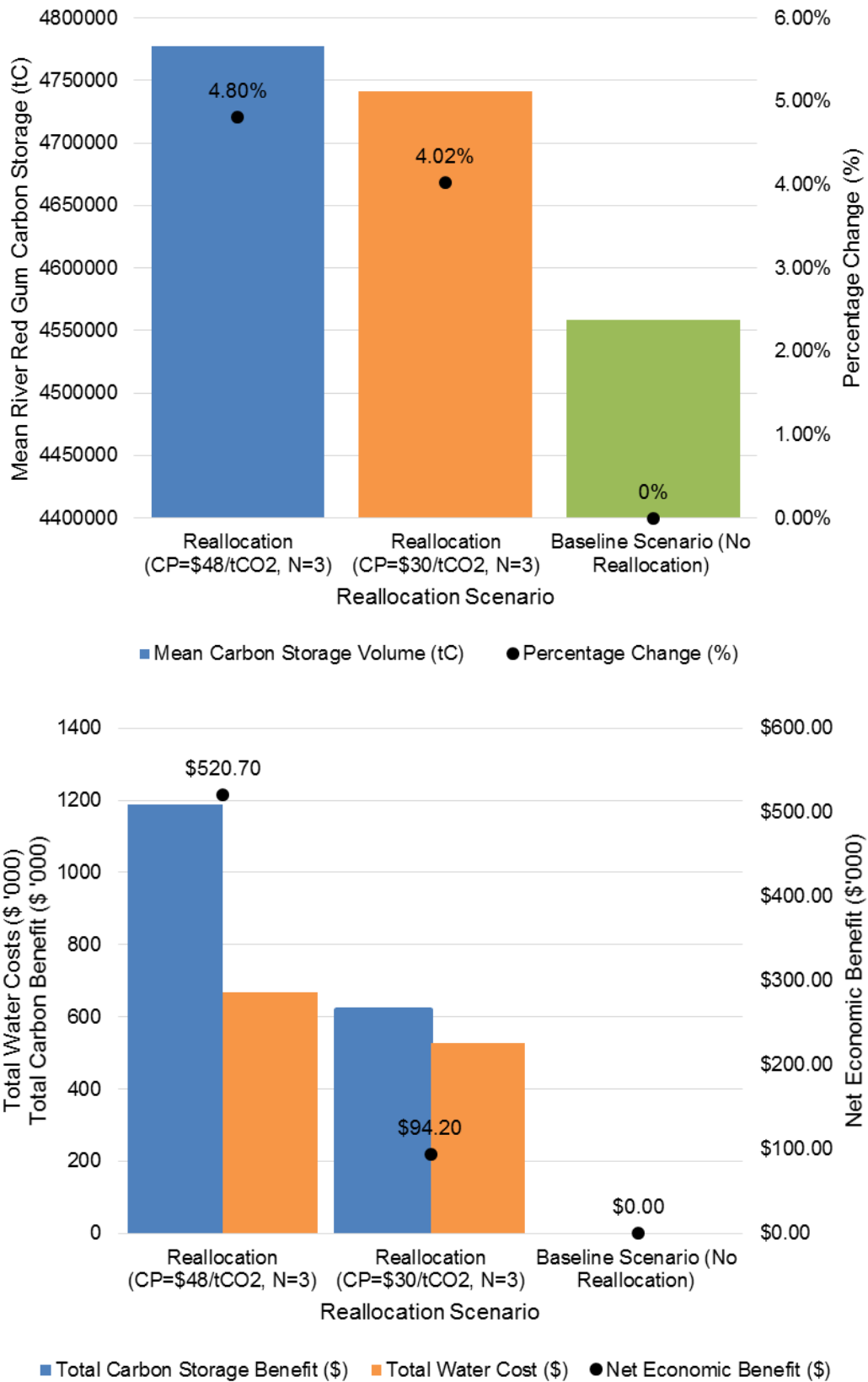
Source: Own Figures.

As can be seen in Figure 7.9, consideration of a higher carbon price, such as the social cost of carbon at $48/tCO_{2e}$, increases the volume of annual water reallocated to the environment. This occurs because the higher carbon price increases the marginal equilibrium point, resulting in a higher volume EWH trade vicinity. The impact of carbon prices on the reallocation strategy and subsequent net economic benefit is presented in the sensitivity analysis in Section 7.6.1.

Overall, it appears there is scope for the carbon benefits generated by increased frequency and volume of overbank flooding, caused by an annually trading self-financed EWH to offset (e.g. equal or exceed) the cost of annual reallocation purchases. The possible business case resulting from this finding is presented in the following section.

7.4.1 Proposed Carbon-Water Trading Strategy

Based on the above results, a self-financing carbon-water trading strategy was developed and is depicted in Figure 7.2. Results above indicate that the proposed strategy is potentially viable for carbon prices greater than $30/tCO_{2e}$ (sensitive to a range of conditions, discussed in Section 7.6.1). However, lower carbon prices may also play a role in reducing the expenditure of environmental water purchases by partially offsetting purchase costs. Further, if carbon was priced at a value reflecting the social cost of carbon, which is currently AU$48/tCO_{2e}$ and proposed to almost double to US$69/tCO_{2e}$ by 2050 (IWGSC-CO_2, 2015), this strategy may become increasingly more viable in the future.

Figure 7.10 Proposed Carbon-Water Trading Strategy
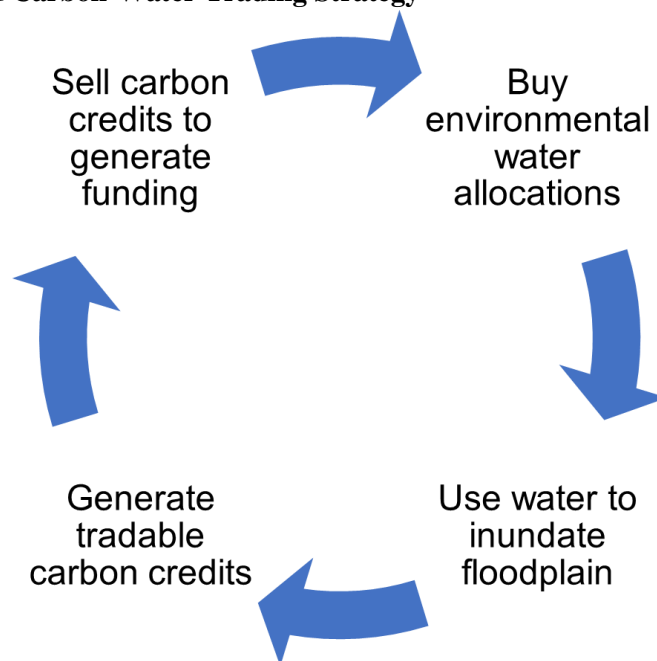
Source: Own Figure.

One possible hurdle to implementing this approach is the demand for carbon credits, if they were able to be sold freely on the market.[46] However, given the current emphasis of carbon neutrality in Australian policy, which allows concerned businesses and cities to become carbon neutral by purchasing offsets generated by carbon farming initiatives (DEWNR, 2017), there may exist enough demand for carbon credits for this approach to be a viable means of financing environmental water purchases. Further, as outlined by Wheeler et al. (2013) and reinforced with survey results in Chapter 8, MDB irrigators have a noted preference for allocation sales to the environment over entitlement sales, indicating there would likely be sufficient volumes of water made available by irrigators willing to engage in environmental water allocation trade.

The strategy depicted in Figure 7.10 is similar to counter-cyclical water trading, which involves the sale of water when it is least needed by the environment in order to buy additional water at crucial times (Kirby et al., 2006; Connor et al., 2013). A version of counter-cyclical trading has been institutionalised in the CEWH environmental water trading strategy, described in Section 3.11.3, and has been successfully implemented in the MDB (CEWO, 2016). The success of this comparable approach to financing water allocation purchases suggests there may be potential for effectual use of the proposed carbon-water trading strategy, although a number of challenges would first need to be overcome, as discussed further in Section 7.7.

7.5 Opportunities to Implement the Proposed Carbon–Water Trading Strategy

Opportunities to implement the modelled carbon-water trading strategy occur in specific hydrological, ecological and economic conditions. Offsetting the cost of reallocation is viable in periods when carbon dynamics are transitioning between states, such that there is an inter-annual difference between carbon storage and potential for additional environmental water to prevent carbon stock decay or improve growth. This supports existing ecological literature, which identifies significant socio-ecological gains by targeting action in periods of ecological transition (Folke et al., 2004). Reallocation also predominately occurs in periods of relatively low irrigator opportunity cost which corresponds to low water allocation prices, as discussed previously. Based on the modelling results and assuming a forward-looking EWH, three particular scenarios were identified where key carbon-water trading opportunities occur: a) very early drought conditions; b) naturally high floods; and c) drought recovery periods. To best understand how this approach may be used in the MDB, it is useful to discuss reallocation opportunities in the context of historical hydrological conditions, as shown in Figure 7.11. As can be seen, there are limited opportunities for reallocation during the first few decades over the historical time-series, due to extreme drought conditions resulting from the Federation Drought. The following section details the hydrological and economic opportunities for reallocation.

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[46] Additional policy challenges regarding the establishment of carbon credit markets in Australia are discussed in Section 7.7.

Figure 7.11 Opportunities to Implement the Proposed Carbon-Water Trading Strategy in the Historical Hydrology Time-series
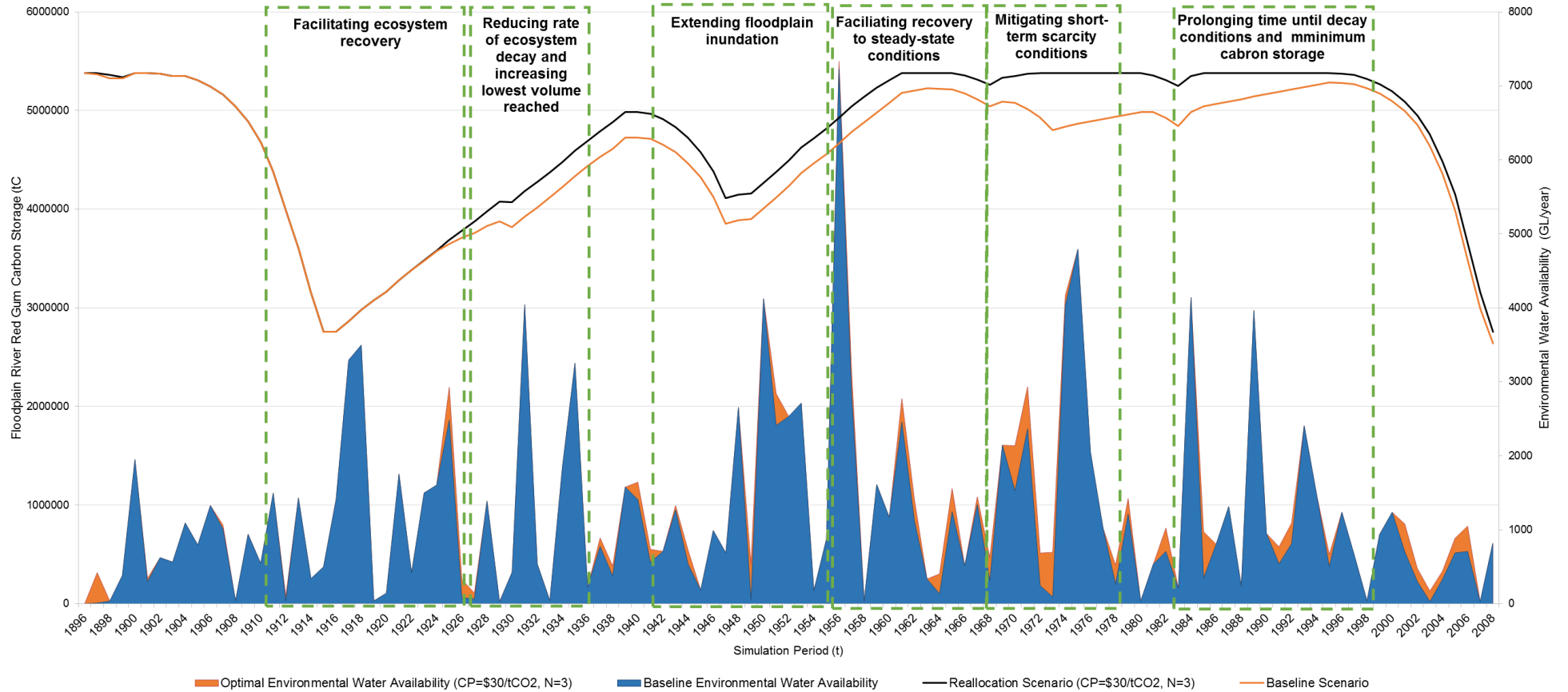
Source: Own Figure.

7.5.1   Initial Drought Conditions

Assuming a forward-looking EWH, a key opportunity to implement the proposed carbon-water trading strategy occurs during the very early years of what becomes protracted drought conditions. These opportunities occur when the potential inter-annual carbon opportunity costs are moderately large, but are not yet exceeded by irrigator opportunity costs, which increase during drought due to high water allocation prices. This strategy can partially mitigate the impact of short-term scarcity (e.g. Eastern Australia Drought 1980-1982) on carbon dynamics by reallocating water to low elevation areas, which would otherwise decay in relatively short periods of scarcity due to the requirement for small and frequent inundation. It was found that providing moderate reallocation volumes (<300GL) leading into drought conditions improved modelled carbon stock losses on low elevation floodplain areas in dry years that followed.

Once drought conditions have fully manifested, there is limited opportunity for economically justified reallocation due to high irrigator opportunity costs. The high cost of water during drought also limits EWH capacity to purchase water of sufficient volume to inundate mid- and high-elevation floodplain areas, which are predominately responsible for the aggregate decay of floodplain carbon. However, during protracted drought conditions (e.g. the Millennium Drought), there are some feasible opportunities to reallocate small to moderate water volumes to aid the persistence of River Red Gums in floodplain areas inundated by floods of 50-270GL. This is feasible due to the extreme decay of River Red Gums during this period (MDBC, 2003; Cunningham et al., 2010), which is expensive when the value of carbon storage is monetised, thus providing sufficient incentive for small to moderate volumes of reallocation.

7.5.2   Naturally High Flood Events

Another circumstance where annual reallocation can generate carbon credit value in excess of water costs is during moderately large flow events when annual reallocation is able to increase flood volume and floodplain inundation extent. These opportunities are driven by the considerable carbon value that can be derived by altering the carbon dynamics in higher elevation floodplain areas, which can typically only be reached during naturally high flooding events (>800GL). In particular, these opportunities occur when irrigator opportunity costs are low due to high consumptive water availability and the watering requirements of low elevation floodplain areas are typically met. In some such cases, the carbon improvement in higher elevation floodplains during flood events can exceed irrigator opportunity costs, resulting in opportunistic cases to purchase allocations at a low price in order to extend the naturally occurring flood event. The historical flooding events in the 1950s and 1970s, as shown in Figure 7.4, provide an example of when this strategy could be implemented. In the MDB, a similar strategy is currently employed called "piggy-back flows" which involves timing environmental water releases to coincide with natural floods, in order to achieve a wider range of environmental benefits through higher volume flooding.

7.5.3 Drought Recovery

A third opportunity to facilitate reallocation by generating carbon credits is in cases where the rate of ecosystem recovery post-drought can be increased by improved watering conditions. River Red Gum recovery, and hence carbon uptake from drought, occurs in periods of persistent moderate flows. In such cases, irrigator opportunity costs are low due to increased flow conditions, establishing opportunities for reallocation. In periods of drought recovery (e.g. the decade following the Mid-century Drought 1937-1947), modelled reallocation increased the annual mean environmental water available and removed the occurrence of low environmental water years (<200GL). This aided in both facilitating the continued modelled recovery in low elevation floodplain areas and preventing a return to decay dynamics, thereby generating increased carbon storage value sufficient to justify the reallocation.

Model results indicate that the purchase and delivery of allocations to facilitate recovery quickens the return of ecosystem dynamics to steady-state conditions, improving ecological resilience to future dry periods. In the absence of drought recovery reallocation, isolated dry periods would induce a shift to decay dynamics and total carbon storage would reach a lower point (which would also decay from a lower point of ecosystem health). This would occur within a shorter time period than under an inundation intervention. Results indicate that reallocation is not only economically justified during periods of drought recovery, but also generates lasting value by increasing the resilience of carbon stocks against future dry periods and increasing the lowest volume of carbon reached.

7.5.4 Comparison of Results with Existing Models

As highlighted in under the literature review in Chapter 5 when modelling environmental water reallocation, the representation of environmental functions has been largely conceptual in nature and has not fully considered environmental benefit valuation governing reallocation. To investigate the impact of including a more detailed representation of environmental behaviour and monetary environmental value on results, the model presented here was run using: a) a conceptual environmental damage function without representation of environmental benefit economics value; and b) the carbon dynamics model and valuation detailed in Chapter 6 which involved a carbon environmental damage function and representation of environmental benefit economics value. In comparing the model all aspects were kept constant, except how environmental dynamics were represented. The subsequent optimal reallocation strategy for both functions was then compared. The following section briefly describes the conceptual model used and the results of the comparison.

7.5.5 A Conceptual Environmental Damage Function

The conceptual damage function adapted from Connor et al. (2013) is used as a comparison to the carbon dynamics model used in the HEM presented within this thesis. The conceptual damage function represents environmental damage as a positive, non-ecological numerical indicator that increases exponentially as the time interval between environmental watering events exceeds the desired return

period for each floodplain zonal area (Connor et al., 2013). Importantly, this approach assumes a full and immediate recovery of ecosystem health after drought and does not account for time lags or the possibility for steady-state ecosystem conditions. Based on a forward-looking EWH, the conceptual environmental damage function is given by:

$$ED\Pr_{(i,t,c)} = \sum_{t=1}^{T}\sum_{c=1}^{11}\left[\sum_{t=0}^{n} w_{(i)}\exp\frac{eh_{(i,t,c)} - EF_{(i)}}{EF_{(i)}}\right] \qquad \textbf{Equation 7.1}$$

Where $ED\Pr_{(i,t,c)}$ is the environmental damage occurring over the modelled EWH's decision horizon, enumerated for all possible unique reallocation volumes in each year, and iterated over the whole simulation time-series (T=113). The variable $eh_{(i,t,c)}$ is an iteratively updated summation variable counting the number of years since flooding required to inundate each floodplain area occurred. $EF_{(i)}$ is the desired return period for each flowband i and $w_{(i)}$ is a unit-less weight corresponding to the area between each floodplain zonal area $A_{(i)}$. Figure 7.12 shows the behaviour of the conceptual damage function for modelled baseline hydrological conditions. As demonstrated, the conceptual damage function captures the expected behaviour of ecosystems in water scarce periods, particularly at the beginning and end of the simulation corresponding to the Federation and Millennium Droughts.

Figure 7.12 Conceptual Environmental Damage Function in the Lower Murrumbidgee Catchment for Baseline (No Reallocation) Conditions
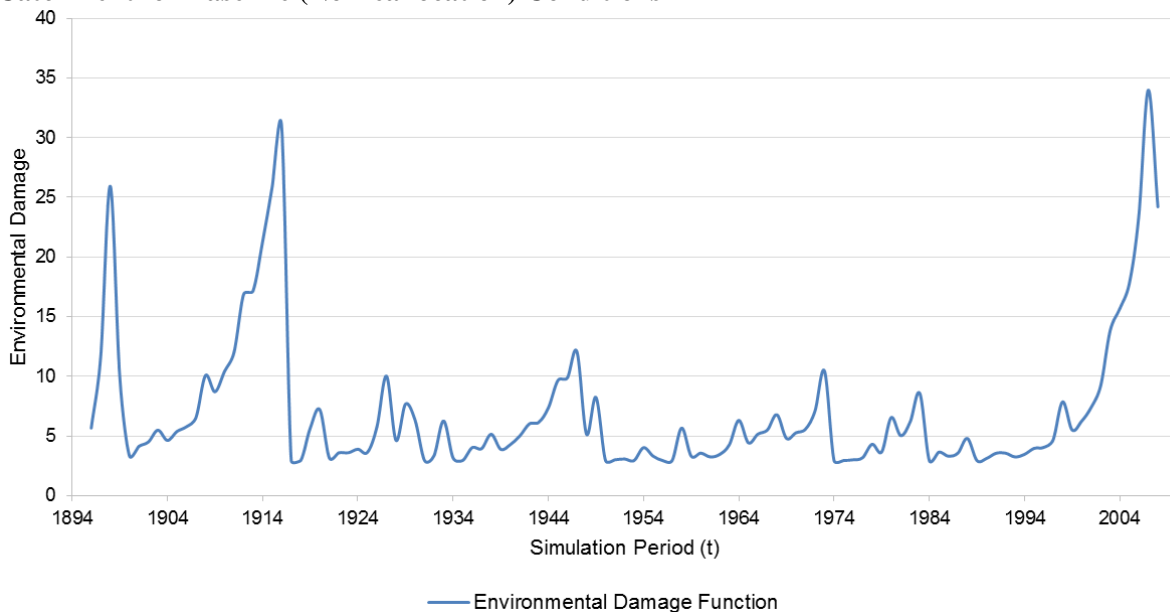
Source: Own Figure.

7.5.6 Comparison of Reallocation Results

Comparing the reallocation results informed by the HEM presented within this thesis and those informed by the damage function presented above in Section 7.5.5, it is found that the model within this thesis results in a less frequent and more targeted reallocation strategy. This ultimately creates a more variable environmental watering regime, which is better suited to the environmental watering requirements of floodplain vegetation (Overton et al., 2009). Table 7.2 shows a range of summary statistics for environmental water availability resulting from the two modelling approaches relative to baseline conditions, including the $S80$ score for the respective flow regimes. A $S80$ score is a hydrological measure of inter-annual variability given by the dispersion about the mean flow, represented by: $S80 = \left[\left(90^{th} \: percentile \: - \: 10^{th} \: percentile \right) / 50^{th} \: percentile \right]$ (Sanford et al., 2007).

Table 7.2 Descriptive Statistics of Modelled Environmental Water Availability

Environmental Water Availability	Environmental Representation Used to Inform Reallocation		
	Baseline	Simplified environmental damage function [Section 7.5.5]	Carbon sequestration and valuation model [Chapter 6] (CP=$30/tCO$_{2e}$, N=3)
Minimum (GL/year)	0	25.1	0
Maximum (GL/year)	7230.6	7230.6	7333.4
Mean [SD] (GL/year)	1186.1[1234.2]	1424.2 [1178.0]	1272.5 [1232.6]
S80	3.4	2.1	3.1
Number of leases [% of years reallocation occurs]	N/A	65 [58%]	41 [36%]

Source: Own Table.

Noticeably in Table 7.2, the environmental watering regime output from both models reduces the inter-annual variability relative to baseline, by reducing the severity of isolated low water availability years (e.g. the 1980-1982 drought). However, due to the identification of opportunities to extend natural flooding (i.e. marginally increasing the peak of high volume floods), the HEM presented in this thesis results in a smaller reduction in flow variability relative to the flow regime informed using the conceptual damage function. Observable differences between the reallocation strategies informed by the two modelling approaches are as follows. First, using the conceptual damage function it is found the best opportunities for reallocation occur during drought (Connor et al., 2013). This result is evident in Figure 7.13, which presents the environmental water availability for baseline conditions and the two modelling approaches, and shows higher levels of environmental water availability during the simulated Millennium Drought period for the flow regime informed using the conceptual damage function. Conversely, the HEM presented in this thesis refines this result and it is shown that the best opportunities for efficient reallocation occur specifically in years leading into drought and during drought recovery periods, as described in Section 7.5. Second, using the HEM from this thesis, it is found that for carbon prices of $30/tCO$_{2e}$ or above there are some marginal opportunities to improve the health of high elevation floodplain areas through the extension of naturally occurring flood events.

These opportunities were previously unidentified using the conceptual environmental damage function. This result demonstrates that the improved representation of carbon dynamics and valuation has enabled the identification of opportunities for cost-effective annual reallocation, which would not have otherwise been evident in previous modelling approaches.

Figure 7.13 Modelled Environmental Water Availability for Baseline Conditions and the Two Modelling Approaches
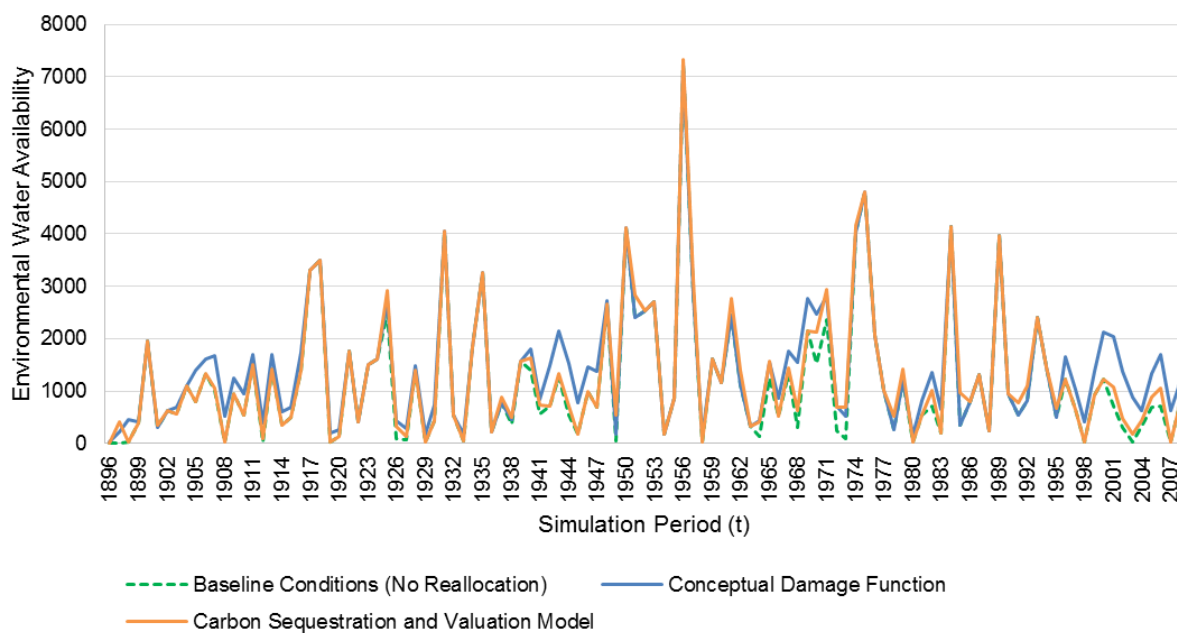
Overall, it is suggested that under-representation of biophysical behaviour and monetary value of ecological change (e.g. the generation of carbon sequestration credits) can lead to an imprecise picture of environmental opportunity costs which may result in missed reallocation.

7.6 Uncertainty Analysis and Treatment

As described in the methodology in Chapter 6, an uncertainty matrix (UM) is developed for the hydro-economic model to identify the source and extent of uncertainty within the model. Using the UM approach identified 24 locations (sources) of uncertainty affecting the model, though this is unlikely an exhaustive list. Of these, five sources were identified to be investigated in the sensitivity analysis, based on the level of uncertainty they contribute and ability to analyse the effects through sensitivity testing. These variables are: a) carbon price; b) number of years in foresight; c) surface water availability; d) initial carbon storage conditions; and e) maximum carbon storage conditions. Note that some sources of uncertainty have not been included in Table 3 because they are driven predominately by other factors. For example, future water allocation prices are uncertain, but are controlled by allocation volume, which is determined by water availability. As such, water availability is investigated as the driving uncertainty.

The UM shows that recognised ignorance and scenario uncertainty are the largest contributor to model output uncertainty. Scenario uncertainty arises largely from the uncertainty of future conditions, such as possible climate futures and changes in carbon pricing. Recognised ignorance is identified for ecological parameters and structural processes where there is insufficient data to fully describe system behaviour[47], such as the maximum carbon storage capacity per hectare of River Red Gums or the time until tree death under baseline conditions. In these cases, recognised ignorance does not mean there is no knowledge regarding a particular aspect of the model, but rather there are some aspects which remain unclear or not fully known. Model structure uncertainty is also identified regarding the assumed relationships between variables within the model. For example, the HEM represents a direct relationship between the frequency and volume of overbank flooding and floodplain tree health, but there may be a more complex relationship or other driving factors that are not accounted for (e.g. access to groundwater, soil moisture content, species competition, salinity etc.).

Typically speaking, model structure uncertainty is rarely addressed in integrated environmental modelling, due to constraining of resources and rigid single-discipline approaches to often pluralistic problems (Jakeman et al., 2006). In addition, model structural error is difficult to detect when time-series of adequate lengths are not available to conduct validation and verification (Jakeman and Letcher, 2003), as is the case for a large number of integrated water resource models constructed in the MDB (Settre et al., 2017). In the absence of alternative modelling structures available for testing at the present time, model structure uncertainty is not further analysed in this thesis.

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[47] In reality, ecological data is rarely completely observed and all ecological data is therefore a snapshot of system dynamics. This is also true for the structural relationships between environmental processes, which evolve as scientific understanding increases. This is evident by comparing the relationship between flow volumes, inundation extent and floodplain health presented in MDBA (2012) and Doody et al. (2016) which present data for the same tree species and the same location, but are marginally different due to improved primary data availability used in Doody et al. (2016).

Table 7.3 Uncertainty Matrix

Location (source)		Level				Nature[#]	
		Statistical uncertainty	Scenario uncertainty	Recognised ignorance	Total ignorance	Epistemic	Stochastic
Context	Future water availability*		✓			✓	✓
	Water reallocation policy		✓			✓	
	Carbon pricing policy		✓			✓	
	EWH decision-making horizon*		✓			✓	
	Floodplain inundation relationship to floodplain health (model structure)		✓			✓	✓
	Floodplain health relationship to carbon sequestration (model structure)			✓		✓	✓
	Carbon sequestration growth dynamics (model structure)			✓		✓	✓
	Carbon sequestration decay dynamics (model structure)			✓		✓	✓
	Technical error	✓				✓	
Inputs	Catchment properties	✓					✓
	River flows	✓					✓
	Desired environmental watering requirements		✓				✓
	Area of River Red Gums inundated		✓				✓
	Maximum carbon storage capacity*			✓		✓	✓
	Minimum carbon storage capacity (dead trees)					✓	✓
Parameter	Carbon prices*		✓				✓
	Water allocation price		✓				✓
	Biophysical calibration parameters			✓		✓	✓
	Rate of tree decay (time until tree death)			✓		✓	✓
	Rate of tree growth (time until full tree recovery)			✓		✓	✓
	Initial carbon storage at t=0*		✓			✓	
SUM		3	10	7	0	15	16

Source: Own Table.

Note: * **Identifies sources of uncertainty tested in the sensitivity analysis.**

Some uncertainties are both stochastic and epistemic. For example, realized climate change impacts on flows are unknown (i.e. epistemic) and flow is a naturally stochastic variable (e.g. stochastic).

7.6.1 Sensitivity Testing Results

7.6.1.1 Overview

The viability of the carbon-water trading strategy, as well as the frequency, volume and timing of annual reallocation choices are sensitive to a number of variables. Selection of variables to be tested are informed by the results of the UM. A summary of results is presented in Table 7.4. This table shows the total volume of water allocated to the environment is given as the sum of annual volumes across the simulation period, as is the total cost of the water purchases. The River Red Gum carbon value is given by summing the monetary value of annual carbon storage volumes across the whole simulation. The improvement in carbon value relative to baseline is given as the difference between baseline and reallocation scenarios. The monetary *net* benefit of reallocation is thus computed as the sum of benefits (improved carbon value) minus the sum of costs (water purchase costs).

It was found that HEM results are sensitive to carbon prices, the number of years in the decision-making horizon, and the volume of surface water availability that drives water allocation prices. In particular, it was found that changes in carbon prices specifically affect the volume of water to be reallocated to the environment, while changes in the ability for the EWH to probabilistically account for future conditions affected the frequency and timing of modelled reallocation. A result is that the number of reallocation years are relatively insensitive to changes in carbon price, but the volume of the reallocation is sensitive to carbon price changes. Changes in surface water availability, which drives the water allocation price, has impacts for both the volume and frequency of reallocation[48]. The volume of carbon stored at the beginning of the simulation and the maximum possible volume of carbon stored both have little impact on the reallocation choices over the length of the simulation. The relative insensitivity to the volume of carbon stored at the start of the simulation is an important result, as it indicates that the proposed carbon-water trading strategy could be feasibly implemented to facilitate carbon storage growth during any period where the ecosystem is storing at least 25% of the baseline long-term annual mean carbon storage.

Interestingly, the changes in reallocation strategy obtained by varying parameter values or inflow scenarios do not cause proportional changes in carbon storage. This results from non-linear responses in River Red Gums carbon storage dynamics to hydrological change, and because the solution algorithm of the HEM selects reallocation to occur in ecologically strategic periods that have optimal benefit to River Red Gum condition. This ensures that reallocation is chosen strategically, in periods where

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[48] In addition to changes in water availability, a valuable future extension would be to examine the impact of changes in flow sequence as well as flow magnitude. This could be explored by introducing stochastic flow variables to construct random flow sequences that better represent the variability and unpredictability of river flow, which is expected to increase with a changing climate. The reason this was not undertaken in the thesis was because stochastic simulation required considerable computing power which was not available for use.

greatest environmental benefit for the volume of water per unit cost are chosen first and at lowest prices. This essentially results because the model dynamically maximises for ecological bang for buck.

Further, as discussed above, changes to flow scenarios alter both the water allocation price (and hence water costs) and the baseline carbon volumes (and hence environmental opportunity costs). As such, it was found that it is difficult to determine if the magnitude of change in the reallocation regime relative to the baseline is a result of changes in water prices or carbon stocks. This relationship is depicted in Figure 7.14. The subsequent difficulty is a result of the highly integrated nature of the HEM.

The complexity of the model output sensitivity, specifically pertaining to the multitude of changes driven by altering inflow scenarios, demonstrates the difficulty in quantifying the magnitude of uncertainty drivers and post-model evaluation (e.g. validation and verification) as discussed previously with regard to highly integrated environmental models (Van Asselt and Rotmans, 2002; Letcher and Jakeman, 2003).

Figure 7.14 Water Allocation Price and Floodplain Carbon Storage Influence Reallocation Decisions and are Sensitive to Changes in Water Availability
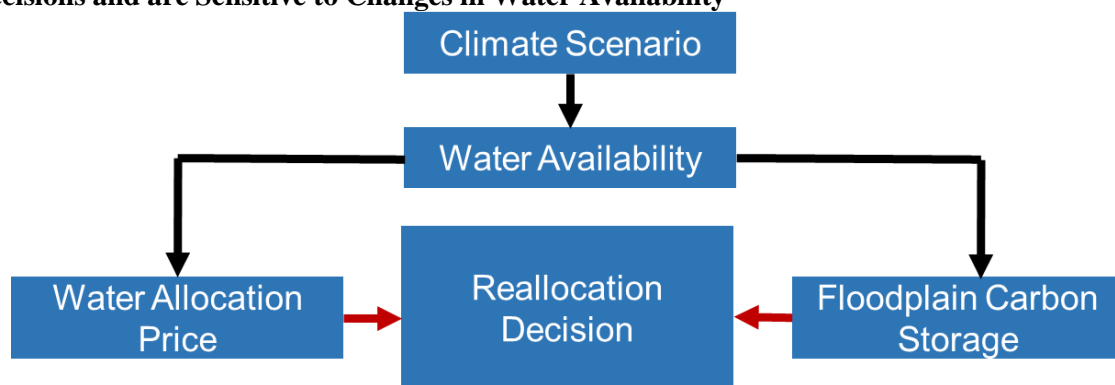
Source: Own Figure.

Overall, based on the sensitivity analysis, it is apparent that the proposed carbon-water trading strategy may be viable for all carbon prices greater than $30/tCO_2$ for certain climate change scenarios and under *Basin Plan* conditions, as well as for a range of initial volumes of carbon stored. Table 7.4 provides an overview of sensitivity results and the following section describes the sensitivity analysis results. Dashed entries in Table 7.4 indicate proposed carbon-water trading strategy is not viable under those conditions.

Table 7.4 Summary of Sensitivity Testing Results for Optimal Reallocations

Baseline Conditions	Total Volume of Allocations Purchased (GL)	Total Water Allocation Purchase Cost ($ million)	Value of Carbon Stored in Floodplain River Red Gums ($ million)	Carbon Value Improvement from Baseline ($ million)	Net Economic Benefit ($ million)
Baseline Conditions					
• CP=$30/tCO_{2e}	N/A	N/A	15,452.6	N/A	N/A
• CP=$48/tCO_{2e}	N/A	N/A	24,724.2	N/A	N/A
Carbon Pricing					
• CP=$30/tCO_{2e}	10,948	526.9	16,073.7	621.1	94.2
• CP=$50/tCO_{2e}	11,915	667.1	25,911.8	1,237.1	520.7
• CP=$70/tCO_{2e}	13,506	929.1	31,745.4	980.2	51.0
• CP=$90/tCO_{2e}	15,267	991.0	47,556.8	1,198.9	207.9
• CP=$110/tCO_{2e}	14,245	1,111.2	59,393.7	2,773.9	1,622.6
Water Availability (CP=$30/tCO_{2e})[b]					
• Median Dry (-9%)	2,383	147.1	11,064.7	278.8	131.7
• *Basin Plan* Scenario (+20% enviro. water)	10,087	499.5	11,301.1	515.2	15.7
• Median Wet (+13%)	15,214	502.1	11,180.9	533.6	31.5
Decision Horizon (CP=$30/tCO_{2e})					
• N=2	10,537	312.5	15,780.3	169.4	15.0
• N=3	10,948	526.9	16,073.7	621.1	94.2
Starting Carbon Storage (CP=$30/tCO_{2e})					
• μ-0.25 μ	-	-	-	-	-
• μ -0.50 μ	11,570	588.4	16,073.8	621.1	32.7
• μ -0.75 μ	11,066	522.8	16,073.8	621.1	98.2

<u>**Source:**</u> **Own Table.**

<u>**Notes:**</u> **a) The value of carbon under baseline conditions is given as the market value of the carbon stored in the existing River Red Gum forests before the delivery of supplementary environmental water allocations. For example, under baseline conditions a hectare of forest storing 100tC is valued at $3,000 for a carbon price of $30/tCO_{2e}.**

b) Based on an analysis of 75 year modelled time-series.

7.6.1.2 Carbon Pricing

Changing the price per tonne of carbon alters the value assigned to the annual increment of growth or decay in carbon stock, as well as the overall value assigned to the River Red Gum floodplain forest. Changes in the carbon price therefore changes the volume and frequency of the optimal reallocation strategy, as shown in Table 7.4. As the carbon price increases there is an approximately linear increase in the total volume of water reallocated to the environment over the simulation period ($R^2=0.81$ for the fitted linear equation). Likewise, for higher carbon prices the total value of the carbon stocks relative to the baseline increases due to the increase of carbon prices, ultimately driving higher reallocation volumes. From a biophysical perspective, higher volumes of strategic reallocation increase overall floodplain health and carbon storage capacity. Conversely, lower carbon prices reduce incentives for reallocation; and if society places no value on carbon (i.e. CP=$0/tCO_{2e}$) there are naturally no opportunities to implement the proposed strategy.[49] Because the changes in carbon pricing alters the timing and volume of the optimal reallocation strategy, the cost of water also changes in response to changes in carbon prices. Water price is driven by exogenously determined water supply and hence is annually variable. Water costs generally increase as carbon prices increase (as does carbon benefit), but not necessarily in a linear fashion. This non-linearity is in part responsible for the non-linear trend in the profit gained from reallocation for various carbon prices.

In terms of reallocation strategy, as carbon prices increase beyond the minimum requirement of $30/tCO_{2e}$ there are additional opportunities available to reallocate during dry periods, where high carbon prices allow the potential gains from reallocation to exceed the irrigation opportunity costs, which are otherwise typically in excess of carbon benefit due to water prices reflecting scarcity during drought. As carbon prices rise, there is also additional incentive to reallocate water to extend naturally occurring flooding events in order to reach higher elevation forest areas that contribute significantly to the volume of carbon stored on the floodplain. There is also an increase in the amount of modelled consecutive lease years as carbon prices increase.

7.6.1.3 Foresight Years

The choice of decision horizon is critical to the choice of frequency and timing of modelled annual reallocations[50]. In particular, as the decision horizon increases there is an increase in the frequency of reallocation, predominately in dry years. This is because longer decision horizons better capture the magnitude of change in carbon stocks during ecological transitions. The instances of consecutive reallocation years also marginally increases. However, the difference in mean volumes reallocated for a decision horizon of two to three years is not statistically different. It is therefore apparent that the

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[49] Note that this does not consider the economic benefit of other ecosystem services that are derived from increased environmental water availability, the change in which may also economically justify reallocation.

[50] Note that the consideration of additional foresight years may alter the reallocation regime, it is considered sufficient to limit this analysis at N=3 years in line with established understanding in the literature of decision maker's diminished ability to account for and act on the conditions in future years beyond this amount.

change in the number of years in foresight has more of an impact on the timing and frequency of reallocation years than on the volume of water reallocated. The impact on the annual reallocation strategy for changes in foresight years of one, two and three years are shown in Table 7.5. Note that due to computer resource limitations, it was not possible to test for decision horizons greater than three years. However, given the very small incremental change in mean and total volume reallocated with the change from a two to three year foresight, little incremental gain from modelling a longer expected value horizon would be anticipated.

Table 7.5 Mean and total volume of water reallocated to the environment for decision horizons of one, two and three years for CP=$30/tCO$_{2e}$

Planning horizon (years)	Mean Reallocation Volume [SD] (GL)	Total volume reallocated over simulation (GL)	Number of reallocation years
1	25.8 [91.7]	2,918.4	11
2	93.2 [175.4]	10,537.3	34
3	96.8 [160.3]	10,948.4	41

Source: Own Table.

7.6.1.4 Surface Water Availability

Changes in surface water availability affect the volume and frequency of economically justifiable reallocations to the environment. When the period corresponding to the Federation Drought is included in the estimates of overall cost and benefits of reallocation, there is a reduced viability for the proposed carbon-water trading strategy, due to very high water allocation costs and the marked trend towards minimum ecosystem health conditions. This effect is heightened for the dry climate change scenario (9% reduction), in which case annual reallocation is not a viable means to revive ecosystems once in the middle of protracted drought conditions. Removing the Federation Drought period from the simulation increases the viability and benefits generated by the proposed trading strategy. These results are presented in Table 7.4. By excluding the Federation Drought period, for a 9% reduction in surface water flows (median dry climate CSIRO (2008)) there remains a number of opportunities to implement the proposed trading strategy. However, overall reallocation opportunities are reduced compared to baseline climate scenario (historic climate) due to increased water allocation prices driven by scarcity, which in turn increases irrigation opportunity costs and reduces the occurrences of equimarginality. This effect occurs due to the removal of many of the periods of low water allocation prices (i.e. less than approximately $50/ML) that were relied upon to purchase water allocations when they are inexpensive but valuable to the environment. In very dry years under the current carbon pricing conditions, there may be opportunities to reduce the cost of allocation purchases but not fully offset the cost. Higher carbon prices are likely to occur in the future (Inter-agency Working Group on the Social Cost of Carbon, 2015) accompanied by a more variable and dry climate, suggesting that the proposed carbon-water trading strategy may have applicability in the future.

Conversely, a possible wetter climate in the future results in higher volumes of water available for the environment. Under a wetter climate, higher availability drives decreases in water allocations, therefore increasing the amount of water available to be purchased for the environment at a lower cost. However, there is also a higher baseline level of ecosystem health (and carbon value) under wetter climates due to increased frequency and volume of overbank flooding. This is a similar case for the *Basin Plan* scenario (plus approximately 20% environmental water availability), which increases the level of ecosystem health and floodplain carbon value for baseline and reallocation scenarios. Interestingly, as higher volumes of water become available to the environment permanently (e.g. *Basin Plan* conditions), there is a reduced need for annual reallocations during moderate isolated low flow years.

Overall, the HEM is high sensitivity to changes in surface water availability is not a surprising result, as it has been previously shown that inflows and diversions (hence, costs and benefits) of reallocation are more sensitive to changes in future climactic scenarios than they are to the volumes reallocated to the environment (Kirby et al., 2014a).

7.6.1.5 *Initial Carbon Storage at t=0*

The assumed initial volume of carbon stored on the floodplain at the start of the simulation plays a role in the trajectory of an ecosystem. The initial volume of carbon stored in the River Red Gum forest impacts the behaviour of carbon sequestration for the initial 20 years of the simulation. For all test levels of initial carbon, the carbon storage capacity recovers after 20 years to baseline level, as is evident in Figure 7.15 which demonstrates this effect for baseline (no reallocation) conditions. This result is supported by the long-run analysis presented in Section 7.3.1, which demonstrates long-term ecosystem dynamics are not influenced by starting conditions.

Figure 7.15 Floodplain River Red Gum Carbon Dynamics Sensitivity to Initial Carbon Storage Volumes at t=1896
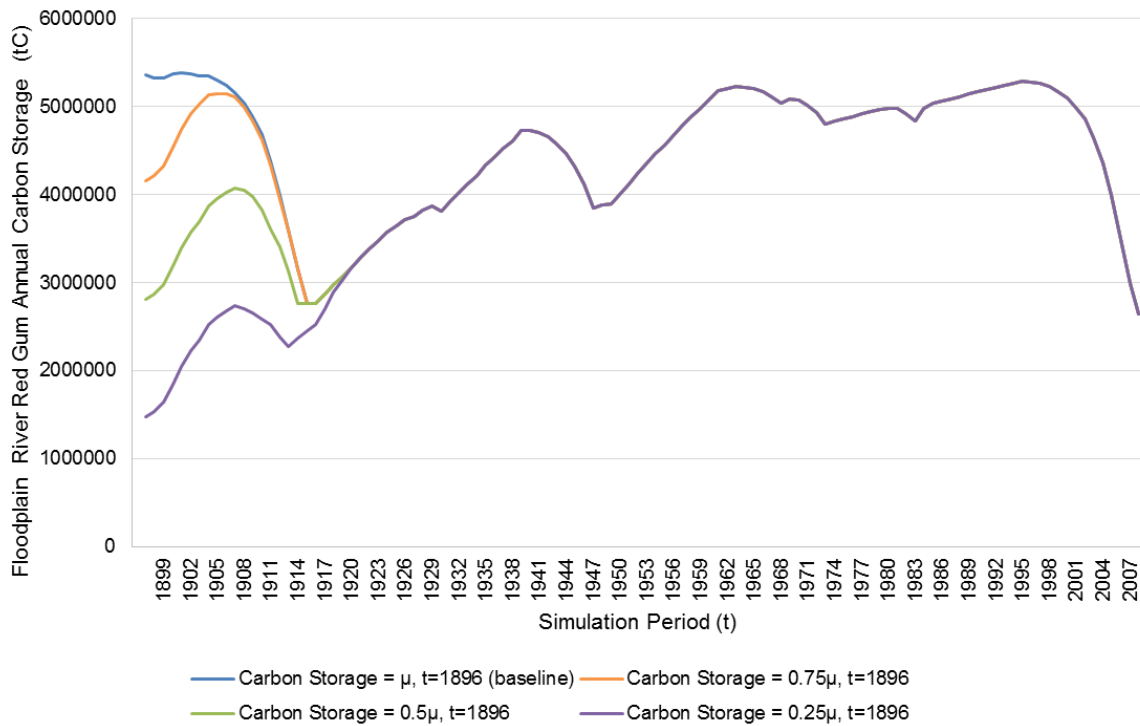
The sensitivity results indicate there is opportunity to implement the proposed carbon-water trading strategy, for conditions in which the initial carbon stored is up to 75% less than the mean carbon storage capacity per hectare. When carbon storage is less than 25% of the assumed mean (i.e. 26.1tCha^{-1}), large volumes of water would be required to improve ecosystem health, which would incur considerable cost and would not be economically viable. However, changes in the initial carbon storage volumes of less than 75% have limited impact on the viability of the carbon-water trading strategy or the chosen reallocation volumes. This result indicates that although the initial ecosystem condition and carbon storage capacity is unknown for the Lowbidgee, the model results are robust to this uncertainty for up to a 75% reduction in the mean volume during the initial simulation period.

7.6.1.6 Maximum Carbon Storage Capacity

Maximum boundary conditions for the capacity of carbon storage per hectare of River Red Gum forest alters the trajectory of carbon assimilation over the simulation, as shown in Figure 7.16. If no boundary condition is imposed, the amount of carbon increases over the length of the simulation. This is not a reasonable biophysical assumption, as forests would naturally tend to a biologically defined maximum, but it is included for completeness. The difference in total carbon stored at the end of the simulation when imposing no boundary conditions (blue line) and the mean boundary condition (purple line), as shown in Figure 7.16, demonstrates the considerable uncertainty that is incurred due to the absence of maximum storage data, which is cumulative over the course of the simulation.

Figure 7.16 River Red Gum Carbon Storage Dynamics Sensitivity to Maximum Carbon Storage Boundary Conditions
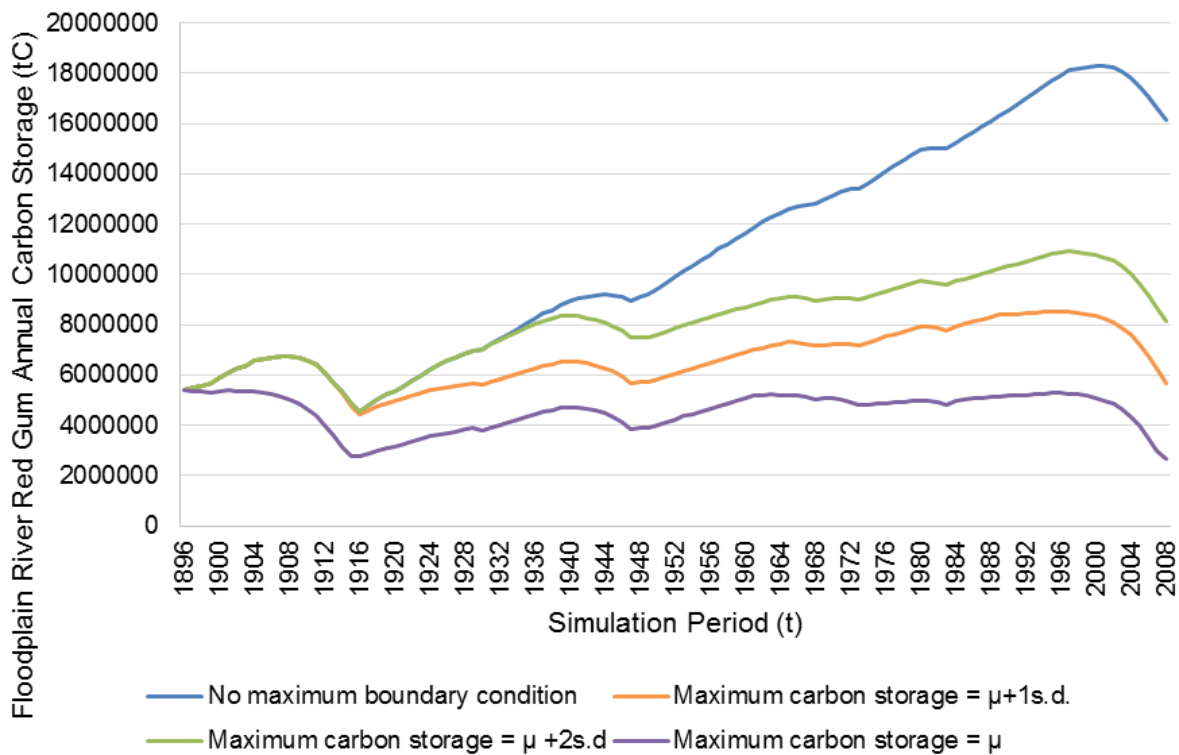
Removing upper boundary conditions in the sensitivity analysis results in no limit being imposed on the modelled economic value that can be generated by improvements in carbon storage, but there is limited change to the cost of reallocation. Therefore when the net economic benefit is calculated (e.g. value of improved carbon stores – cost of reallocation), an unfeasibly large and unrealistic value is obtained, and hence is not included in Table 7.4. This result demonstrates, qualitatively, that economically justified reallocation is possible for a range of maximum carbon storage capacities above the mean (which is modelled in the baseline case), and would potentially generate higher economic returns. Greater carbon storage than is modelled with baseline parameters may be possible if additional vegetation groups in the floodplain are considered and if possible further sequestration impacts downstream are considered.

7.7 Discussion

Managing forests to generate carbon credits is an active process involving actions taken to promote carbon uptake and prevent carbon loss (Law, 2013). In a system consisting of a hydrologically connected river and floodplain, this chapter has shown that augmenting the overbank flooding regime through annual water reallocation in strategically targeted hydrological conditions may constitute one such action. Further, it was shown that it may be cost-neutral to do so, if the added carbon storage (or prevented carbon loss) can be traded at a market value greater than $30/tCO$_{2e}$ to offset, or at least reduce,

the cost of the additional environmental water. The following sections discuss the potential challenges of implementing this result.

7.7.1 Potential for a Self-Financing EWH

As described in Chapters 2, 3 and 4, the process of reallocating water in the MDB has been extremely costly. This has partially been due to the emphasis placed on irrigation infrastructure upgrades since 2015 (MDBA, 2016a), but also to the fact that water entitlements are a scarce and expensive commodity. This is a particular challenge for allocation acquisitions because allocation prices are driven up by scarcity, which corresponds to periods when environmental assets would benefit most from the delivery of supplementary allocations. For EWHs without access to government funds, or in the case of cash-strapped public agencies, the cost of allocation purchases may pose a considerable burden. To incentivise EWHs to buy water for the environment, there is therefore a need for it to be cost-effective.[51] The results of the HEM indicate that for regions with active water markets and carbon markets, offsetting annual water purchases through the generation and sale of carbon credits may present one means of addressing this need.

As discussed in Chapters 3 and 8, there are few but increasing numbers of NGOs engaging in the market for environmental water in the MDB and internationally, thus increasing the relevance of self-financing schemes independent of government funding. Parallel to this, there is a growing social and government emphasis on the offsetting of CO_2 emissions through the purchase of carbon credits. The carbon-water strategy proposed here may be useful for EWHs in the MDB as a means to finance environmental restoration in a financially viable way, as well as being beneficial to concerned cities or companies interested in offsetting their CO_2 emissions. The potential is for a novel revenue stream to assist a self-financed EWH alleviate the financial burden of acquiring water for the environment and a convincing business case for increased investment in environmental water holdings.

There are two self-financing methods currently employed by government and non-government EWHs engaged in buying water for the environment. The first is counter-cyclical trading, which is employed by the CEWH to sell water allocations when it is not needed by the environment, generating funding to buy water when it is more needed (Kirby et al., 2006; Connor et al., 2013). The second method was developed by MDB Balanced Water Fund, originally introduced in Chapter 3, which involves the purchase of entitlements and partial sale of the allocations. The acquired entitlements are managed such that 80% of the annual allocations are sold on the market to generate value for investors, and 20% of allocations are dedicated to environmental flows (e.g. an 80/20 approach) (B. Richter, pers. comm. 2016). Comparing the carbon-water trading strategy to the two existing financing methods, it is clear that counter-cyclical trading and the 80/20 approach require considerable upfront capital from either private investors or government funds to purchase entitlements from which the allocations are derived,

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[51] The role of funding in EWH decision-making is examined further in Section 8.6.

which can be costly.[52] Conversely, the proposed carbon-water trading strategy does not require the initial ownership of entitlements to generate allocations which can be sold, but rather a single financial injection to initially buy the water allocations which can then be used to generate the carbon credits, which in turn can be sold to fund further allocation purchases, and so forth. Further, the proposed carbon-water trading strategy has an advantage over existing self-financing mechanisms because unlike annual water allocations, carbon credits do not expire after one year and may be owned long-term if desired. Alternatively, generated carbon credits could be held until favourable market conditions arise for sale. This attribute negates the idea held by some EWHs[53] that the purchase of annual allocations is not a secure financial investment due to their time-limited nature, with carry-over subject to availability only in certain regions. Further, as the strategy proposed employs acquisition of allocations rather than entitlements for the environment, it is possible that the proposed strategy could increase willingness for irrigators to engage in water reallocation programs and aid in the structural adjustment of farming communities (Wheeler et al., 2013a).

Based on the sensitivity testing of the model it is apparent that the proposed strategy is reasonably robust to changes in water availability, but the frequency and volume of the reallocation decisions would necessarily change. This indicates there is scope for the carbon-water trading strategy to be implemented once sustainable diversion limits (SDL) planned for the MDB have come into effect. However, under the fully implemented SDL, it is likely that environmental watering requirements will be more often satisfied due to the increase in baseline environmental water availability (thus satisfying floodplain zonal areas of lower elevations) and therefore annual water reallocation may be required to occur less frequently. Overall, it is suggested that the proposed carbon-water trading strategy could provide a novel revenue stream for self-financing EWHs to engage in the market and allow them to generate considerable environmental benefit from improved floodplain health and carbon storage. However, there are a number of policy challenges that must be overcome in order to implement this strategy, as discussed below.

7.7.2 Policy Challenges

Carbon credit trading in Australia is in its relative infancy and significant policy progress would be required to fully implement the proposed carbon-water trading strategy. In particular, the recognition of additional carbon credits generated by naturally existing and mature forests would be required. At present, Australian carbon offset policy and emissions reduction required under the Kyoto Protocol places emphasis on carbon credits accrued by new plantations. Existing non-production natural forests (e.g. not timber plantations) are typically excluded from carbon accounting systems. Further,

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[52] The long-term average price for a 1GL water entitlement is roughly $200,000, depending on its security and location.

[53] More details on the challenges of investments in time-limited water products and an exploration of EWH perceptions are presented in Chapter 8.

sequestration programs have relied on the plantations of non-native species planted for timber production, which also has had negative implications for Australian biodiversity (Conte and Kotchen, 2010). The reason for exclusion of existing native forests is a result of a lack of ease in the ability to quantify the carbon stored in highly heterogeneous existing native forests (Jonson and Freudenberger, 2011) and on the assumption of limited additionality (Law, 2013). Additionality refers to the ability of a carbon offset project to demonstrate distinct carbon benefits relative to that which would naturally occur under a business-as-usual scenario (Gillenwater, 2012). The analysis presented here suggests that strategically targeted overbank flooding may well provide additional benefits that were not otherwise realised under the baseline scenario, particularly during specific hydrological sequences discussed in Section 7.5. This suggests that strategies to improve the overbank flooding regime may be considered as an active management action, which could allow native forests to enter into carbon accounting schemes. This finding supports existing literature, which suggests that the key determinant of carbon stock change in native forests are parameters affecting accumulation of carbon in forestry biomass (Keith et al., 2015), which in the case presented here refers to the biomass accumulation generated by improved overbank flooding.

There may be a number of benefits in considering the expansion of policy to allow for active management of existing native forests to accrue and sell carbon credits, given the capacity for floodplain native tree species to sequester large volumes of carbon (Smith and Reid, 2013). Namely: a) providing a clear economic incentive for the conservation of native forests; b) a novel forestry-based income alternative to deforestation for natural resource managers (e.g. the owners of forests, such as private plantation owners of public government lands); and c) incentive for the purchase of environmental water to ensure improved forest condition.

Overall, the monetary value of native floodplain forests, as presented here, makes a significant case for the preservation and management of native forests for improved carbon storage and the inclusion of native forests in national carbon accounting schemes. This is particularly pertinent in Australia, where despite the importance of native forests in climate mitigation, project funding for native forest protection has been comparatively limited compared to funding to alternative carbon storage schemes such as new plantation carbon sequestration (Jonson and Freudenberger, 2011).

An additional challenge to implementing the proposed strategy is assigning property rights for the additional carbon generated. It would be logical to assume the carbon credits belong to the entity that owns the forest, such that carbon property rights would be aligned with land rights. However, without the actions of the EWH annually reallocating water to inundate the floodplain, the carbon may not have been generated. In such instances, the ownership of the additional carbon could potentially be attributed to the entity that generated the carbon, such that carbon property rights align with the water rights used to generate them. In the case at hand, the majority of the Lowbidgee forest is within the bounds of

Yanga National Park, which is federally owned. This means that if the CEWH was to implement the proposed carbon-water trading strategy there would be no inconsistencies in the ownership of property rights, because the land, water and carbon rights would all be federally owned. However, in cases where a non-government EWH may act to inundate floodplain forests on public lands, or a government EWH inundates privately owned forests, the ownership of the carbon and the subsequent profit from sale of carbon credits is less well defined. The considerable challenges in defining the ownership of additional carbon suggests that this strategy would be best employed by a state or federal EWH on publically owned lands, or by private EWHs on privately held lands, but not a combination of the two without considerable additional institutional agreements. The question of the institutional arrangements supporting carbon property rights is not the focus of this thesis and so further discussion is not presented, although it is clearly warranted in future research.

Lastly, for the carbon-water trading strategy to be implemented there would need to be a significant increase in the price per tonne of carbon. Over the last few years, the ERF auctions have kept the average carbon price just under $12/tCO_{2e}$ (CER, 2015a; 2015b; 2015c; 2016a; 2016b; 2017). As discussed in Section 7.3, under certain hydrological and economic conditions the cost of allocation purchases could be fully offset for carbon prices greater than $30/tCO_{2e}$. Under Australia's current pricing scenarios, the proposed carbon-water trading strategy may be used to partially offset the cost of annual allocation purchases, which would still be a useful tool for EWHs engaged in the market, but would not be able to fully cover water purchase costs or generate a profit. A higher market price of carbon in Australia, such as $35/tCO_{2e}$ set by the US-EPA and used by the IMF, is a prerequisite for the full application of the proposed trading strategy. The importance of appropriately pricing carbon to enable economic and environmental actors to achieve positive ecological outcomes has been most recently highlighted in the *Paris Agreement* on Climate Change. Notably, and in support of the findings in this chapter:

> "*If we really want to send market signals to enable enterprises to make their decisions under optimal economic conditions, which may be optimal ecological conditions, then the issue of carbon prices inevitably arises as it is the most tangible signal that can be sent to all economic actors*" [Hollande, 2015].

Source: IMF, 2015. Available: https://www.imf.org/en/News/Articles/2015/09/14/01/49/pr15473, [Accessed, 3rd July, 2017]

7.8.3 Expansion to Additional Marketable Ecosystem Services

Although this research considers only the carbon sequestration and storage ES, and only within the Lowbidgee, it is known that improvements in environmental water availability generates other ecosystem services of considerable value (CSIRO, 2012; Bark et al., 2016). Based on this knowledge, it is hypothesised that similar strategies involving the trade of marketable ecosystem services might also provide a viable means of offsetting water allocation purchases. One such marketable ES could be

biodiversity, for which existing credit schemes and markets already exist. In existing schemes within NSW, biodiversity credits are created by management actions undertaken by landowners to enhance and protect biodiversity, which can be sold to developers (e.g. urban sprawling, mining interests) who have a negative impact on biodiversity through their commercial actions (NSW Department of Environment, Climate Change and Water 2009). Biodiversity credits can be generated based on the protection of threatened species and their habitat. As shown by CSIRO (2012), enhanced habitat condition constitutes one of the largest ES improvements when environmental water availability is increased, therefore indicating a possible avenue for the generation and sale of biodiversity credits to offset water allocation purchases.

Future work would benefit from a wider consideration of ecosystem services generated by water reallocation to fully simulate the ES and irrigation water trade-offs, as well as exploring additional offset and trade strategy possibilities. Further methods to potentially assess these benefits are presented in Appendix C.

7.8   Study Caveats and Limitations

To achieve the level of integration presented in this HEM, a number of assumptions were required which can introduce some study limitations. The following section describes the key caveats and limitations of the results.

Firstly, floodplain wetlands have enormous spatial and temporal complexity. The HEM presented in this thesis considers a simplified representation of one catchment and tree species only, which may not be representative of other river catchments elsewhere. The degree to which carbon credits generated by overbank flooding can offset environmental water purchases is a function of the catchment geomorphology and eco-hydrology of the river basin in question. Modelling results are therefore valid only for the Lowbidgee River Red Gum community, though there is wider explanatory and conceptual value.

Due to the absence of existing data, the value of carbon stored in other prominent Lowbidgee tree species (e.g. Black Box and Lignum) is not considered, nor is the carbon stored in grassland, soil or woody debris. The results in this chapter are therefore a substantial underestimate of the total volume (and economic value) of carbon stored on the whole Lowbidgee floodplain and the overall incentive for reallocation. In addition, improvement in River Red Gum health is measured solely as the market value of carbon and does not account for other perceptions of value, such as habitat value, existence value, sense of place and cultural significance, or additional value gained from improved environmental water availability, which previous studies have shown to be considerable (CSIRO, 2012; Bark et al., 2016). Additionally, flows allocated for environmental purposes in the study area can have positive environmental impacts downstream, specifically with regard to water quality. However, these have not

been modelled in this thesis and further work would benefit from considering the downstream impacts of increased upstream environmental water availability.

Further, throughout the simulation it has been assumed that the carbon price is exogenous and constant. This is a reasonable assumption given the relatively little movement in carbon prices derived from the Clean Energy Regulator auctions during 2015-2017, but may not be representative of other conditions or future pricing policies. Future research would therefore benefit from integrating a variable carbon price into the HEM, to assess the impact of temporally variable carbon pricing on EWH trading strategy.

Importantly, in the absence of primary data, it is assumed that the maximum volume of carbon stored per hectare is limited by the mean capacity observed in field surveys (Smith and Reid, 2013) and the model is calibrated based on this assumption. This has the effect of truncating potential carbon growth to the mean volume, thereby providing a significant underestimate of the total carbon (and hence economic) benefit generated by reallocation. Future research could further improve the representation of the floodplain carbon dynamics as ecological data improves in length and quality in the future.

Another limitation to the model is the absence of consideration of general equilibrium effects on, for example, agricultural commodities. An analysis of impact on water allocation price is undertaken and presented in Section 7.3.2, but further work to examine wider impacts would be beneficial in future iterations of this model.

An additional and important caveat to the model is that this model assumes the EWH has perfect foresight of future economic and biophysical conditions, such as water prices and carbon stocks. In reality, hydro-ecological systems are complex and market participant rarely have full understanding of expected values and expected states of nature. As such, the results presented in this chapter represent the results derived from an optimal water reallocation strategy based on perfect information, rather than a realistic reallocation strategy based on imperfect information, human error and the stochastic events. While this means that the specific values may not be realised if implemented in the reality (due to missed opportunities as a result of imperfect foresight, for example), the results are valuable in providing a well-informed estimate at the magnitude and direction of expected costs and benefits.

The final caveat of this model is that, as with all models, the results presented in this chapter are derived from a HEM that is a mathematical abstraction of reality. The results are therefore indicative of possible realised impacts and opportunities, but do not represent observed effects. Model results must therefore be interpreted and implemented with judicious consideration of the limitations and caveats discussed above.

7.9 Conclusion

The results and discussion of a highly integrated HEM, incorporating a novel representation of floodplain carbon storage behaviour used to evaluate the strategic water reallocation, have been presented in this chapter. The HEM builds on the tradition of environmental and consumptive water-use value trade-off analysis, and extends the literature by more detailed consideration of ecosystem response. The results of the HEM demonstrate that the inclusion of a biophysical model of carbon growth and decay dynamics enabled the identification of opportunities for reallocation during periods of inter-annual differences in annual carbon sequestration, which would not have otherwise been evident.

Using the improved means of representing carbon floodplain dynamics, it was determined that there may be scope for the market value of carbon credits generated by improved floodplain condition to finance the purchase of water allocations for the environment. This may provide a novel source of income for a self-financing EWH interested in generating improved environmental flows and floodplain health, and help to reduce the financial costs of buying water for the environment through the market.

The results demonstrate that the opportunities to implement this strategy occur in specific hydrological and economic conditions. These conditions occur primarily in cases where low to moderate water allocation prices coincide with periods of ecological change, such as periods leading into drought condition, during ecosystem recovery after drought, and some marginal opportunities to extend naturally occurring high flooding events. The strategy is viable for carbon prices greater than approximately \$30/tCO$_{2e}$, although lower prices may be used to partially offset water allocation purchase costs under some scenarios. The strategy is viable in low to moderate water price scenarios, between \$20/ML to approximately \$73/ML. The scope of the opportunities to offset the cost of water purchases and the ecosystem impact of reallocation were found to be sensitive to a number of assumptions. In particular the volume of reallocation is sensitive to changes in carbon pricing, while the frequency and timing of reallocation is affected by the ability for the modelled EWH to probabilistically account for future conditions. Both the reallocation volume and frequency are sensitive to changes in water availability.

Overall, the HEM results indicate that there may be a convincing business case for a self-financing EWH to consider purchasing water allocations to generate floodplain carbon credits, in order to finance future environmental water allocation purchases. However, before such a strategy could be implemented, a number of policy challenges must be overcome, including a commitment to a higher carbon price and the inclusion of existing native forests in carbon credit accounting schemes.

To further explore the role of funding arrangements and the strategies of self-financed EWHs, the following chapter presents a qualitative analysis of how water trusts engage with the market to acquire environmental water in the western US.

CHAPTER 8: Localism, Community Engagement and 40 cups of Coffee: The Role of Non-government Environmental Stewards in Water Management in the US and Applications to the MDB

This chapter presents an expanded version of a working paper of the same title. The majority of work presented in this chapter was undertaken as part of an Endeavour Research Fellowship undertaken at the University of California, Riverside, and departs from the MDB context to consider the use of water markets in the western US. A detailed discussion of the complexities of western water law are omitted for brevity and relevance but can be found in cited texts and briefly overviewed in Section 3.9. This chapter combines both qualitative and quantitative methodologies to assess the role of non-governmental environmental water holders (NGO EWH) in facilitating market-based environmental water transactions[54] to the environment in the western US through the lens of localism and social trust. Drawing on additional MDB survey result, the original working paper was expanded to include a broader discussion of non-government organisation (NGO) involvement in the water market in Australia, as well as a discussion on the opportunities and limitations of subsidiary-style management of environmental water.

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[54] This chapter draws on literature and data from the western US context. For consistency with existing literature the American water lexicon is adopted. For example, environmental water transfers (environmental water trade in the Australian vernacular); water transaction (water trade); water rights (water entitlements); one year lease (water allocation); transfer tools (water products).

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8.1 Introduction

Water scarcity affects a number of major water basins around the world and poses considerable challenges for the governance of fresh water resources. In particular, shifts towards drier climates in semi-arid and fully appropriated watersheds in south-eastern Australia and western US presents risks to some irrigated agriculture and flow-dependent ecosystems. In the western US, market-based environmental water transaction (EWT) programs have been unevenly implemented to increase instream flow availability for habitat and species conservation (Neuman, 2004; King, 2004). EWT programs are set within a polycentric water governance approach incorporating federal leadership, catchment scale decision-making and non-governmental involvement (Garrick et al., 2011). Within this context, water-use is governed by the prior appropriation doctrine, which prescribes the reasonable and beneficial use of water in an abstraction pattern consistent with the seniority of the water right (Tarlock, 2001). The polycentric approach to governance includes a diverse range of water actors and reflects a political transition away from state-led command-and-control management of resources that is also evident elsewhere in the world (Settre and Wheeler, 2016). Considerable literature has been developed to describe this process of reform, particularly in Australia, however a number of issues that remain less addressed in the literature. Namely, what is the best way to implement water policy reform (Wheeler et al., 2017) and whom, or which institutions, have the right to acquire, own and manage water rights for the environment? Furthermore, what are the best arrangements to maximise irrigator engagement with EWHs to ensure socially and environmentally beneficial outcomes?

Within the polycentric landscape of current water policy in Australia and the western US, freshwater focused non-government organisations (NGOs), also known as water trusts (King, 2004), are playing an increasingly important role in facilitating the acquisition of instream flow rights, particularly in the western US. However, their approach to water acquisition and the vertical and horizontal linkages between NGOs and the wider water community, including government agencies, remains largely under-examined. This chapter therefore seeks to explore the role of NGOs in facilitating and managing EWTs and identify insights, if any, which may be applicable to international cases where non-government actors play a comparatively smaller role in the environmental water market. One such place that may benefit from these insights is the MDB in Australia, which is seeing an observed increase in private and NGO involvement in the water market, such as the *MDB Balanced Water Fund* administered by Kilter Rural and the Nature Conservancy, previously discussed in Section 3.8.

Further, while NGO and private sector participation are well established in the US, the non-government ownership of water as a historically public resource is a relatively novel concept internationally. In a world where market-style transactions are being more commonly implemented as solutions to environmental problems (e.g. water markets, emissions trading schemes and biodiversity credit schemes), understanding the role of NGOs and private enterprise in instream flow transfers may contribute to the wider discussion of non-government engagement in market-based environmental

policy. The aim of this study is therefore to generate an understanding of the role and challenges of non-government environmental water holders (NGO EWHs) in the market for environmental water in the US, as well as identifying applicable insights to the Australian water market context.

8.2 Environmental Water Transfers Background[55]

An environmental water transfer (EWT) or instream flow transaction is the voluntary reallocation of water from consumptive use to the environment. EWTs occur within a market framework, where water rights holders forgoing water-use are compensated at a market or negotiated rate. The degree to which market-based transactions can be used as an environmental water management tool is subject to the maturity of the supporting water market institutions and regional instream flow legislations which protect the (re-)abstraction of instream flows (Garrick et al., 2011). In states where institutional arrangements allow for it, EWTs have emerged as a considerable segment of the water market. For example, from 1990 to 2003, EWTs accounted for 17% of volume leased (4,785GL) and 16% of volume sold (234GL) in the western states[56] of the US (Brown, 2006). Commercial water market literature suggests that EWTs account for approximately 40% of total volume of trade in the western US (WestWater Research Inc., 2014), although this result is difficult to validate.

Environmental water products are typically traded at lower prices per acre-foot compared to agricultural water trades, owing to the short-term nature of the transfers (Doehety and Smith, 2012) and the derivation of compensation price through negotiation, rather than mature markets or price discovery mechanism such as auctions. EWTs can differ by volume, duration, timing and individual contractual agreements, as shown in Table 8.3. Agricultural water rights holders are the primary sellers of water to the environment, with state and federal agencies the largest buyers by volume (Hanak and Stryjewski, 2012). However, mirroring the diversification and decentralisation of water resource management, additional private entities, NGOs and watershed conservancies have also become buyers of environmental water.

8.3 Literature Review

8.3.1 Localism and Social Trust in Resource Management

There are many different modern definitions of social capital, which diverge on the finer points of this term (e.g. see Sissiainen, 2000). However, in essence social capital is the set of networks, norms of reciprocity and co-operation that enable social and civil engagement and facilitate good governance (Putnam, 1993). Social capital within communities is fed by a shared identify, shared set of norms and

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[55] Considerable background on environmental water transactions and discussion of the US context has been provided in Section 3.9 and will not be repeated here.
[56] Western states of the US include: Washington, Oregon, California, Montana, Utah, New Mexico, Wyoming, Arizona, Idaho, Nevada and Colorado.

a common set of values that enable co-operation among individuals or groups (Brain, 2007; Hoogesteger, 2013a).

Ostrom and Ahn (2003) argue there are three pillars of social capital that apply to a community's ability to solve collective action problems, such as water resource management issues. These are: i) social trust; ii) networks; and iii) formal or informal rules or institutions (Ostrom and Ahn, 2003). A particular focus of the qualitative data collection for this chapter (described in Section 8.4.1) is the role of social trust in the success of market-based water reallocation programs.

The success of a water reallocation program requires stakeholder acceptance and participation (Lane-Miller et al., 2013) both from; a) irrigators who own the rights, and b) the EWHs who are to acquire and manage them. Wheeler et al. (2017) argue that an important influence on full stakeholder participation is social trust, which is the belief in other's honesty, integrity and reliability (Taylor et al., 2007). In particular, trust in the agencies and institutions implementing water policy reform is a vital key in making sure socially beneficial outcomes can be achieved (Wheeler et al., 2017).

There is evidence to suggest that rural American communities or residents of small towns are more altruistic, honest and trusting than other community members (Putnam, 2000). This result has also been replicated in Australia by Onyx and Bullen (2000) who show that residents of rural NSW have overall higher levels of social capital compared to residents of Sydney and score higher in terms of feelings of trust, safety, community participation and neighbourhood connections. Putnam (2000) suggests that one reason for this is that being involved in community life is more appealing when the scale of community is smaller, such as in rural areas. In addition, Schneider et al. (2003) demonstrate that it is through repeated and regular interaction with others that trust and collaboration are built. Both of these explanations have an underlying theme of proximity to others (e.g. small community) and localism (e.g. commitment to the local community). For example, Sligo and Massey (2007) show that family and local social networks remain the greatest source of trusted information in farming communities (despite, for example, access to external information on the internet). Hoogesteger (2013b) suggests that this occurs because social capital operates on the norms of trust and reciprocity which commonly exist *within* regional farming communities, but which may not necessarily exist at an inter-community or inter-sectional level. In the MDB, this is demonstrated by Wheeler et al. (2017) in an analysis of an irrigator survey which revealed that irrigators responded more favourably to statements such as: *'I trust my community water group',* than to *'most people can be trusted',* indicating that irrigators were more likely to trust those within their local community than people in general. This result confirms findings by Lubell (2004) who shows that rural American agricultural communities trust other community members and local government agencies equally, but trust higher order governments less so. One reason for the distrust of higher order government agencies is a lack of frequent interaction (e.g. low levels of repeated social interaction) between government agents and rural community members (Palmer et al.,

2009). Additional reasons include a perception that governments do not represent rural interests (Sligo and Massey, 2007), and a long-term perception of injustice perpetuated by the government against rural communities (Gross, 2011).

For water reallocation programs predominately facilitated by federal agencies, such as the case in Australia and in some programs in the western US (see Chapters 2 and 3 for background), the lack of social trust between rural agricultural communities (who own the majority of water entitlements) and government agents poses an interesting hurdle for the effectiveness of water reallocation. One additional avenue that has been explored in environmental water management is the involvement of non-government organizations (NGO) as stewards of environmental water. The following sections review the current literature on NGO involvement in water reallocation programs and their role in subsidiary-style management of environmental water.

8.3.2   Non-Government Environmental Water Holders

Freshwater focused NGOs engaged in acquiring water for the environment are commonly referred to as *water trusts* (Neuman and Chapman, 1999; King, 2004; Garrick et al., 2011). Water trusts are typically private, non-profit, market-orientated organisations concerned with the protection of ecologically significant and flow-dependent ecosystems or species (Harder, 2006).

Water trusts evolved out of the land trust model of conservation (Neuman and Chapman, 1999; King 2004) and originated predominately in the Pacific North West of the US following the passage of progressive state legislation adopted to restore instream flows, such as the Oregon's *1987 Instream Water Rights Act*, which declared instream uses to be beneficial uses of water (Neuman, 2004). Importantly, the statute allowed for the creation of instream rights through three mechanisms, one of which being the *"purchase or lease…of an existing right or portion thereof for conversion to an in-stream water right"* (Neuman and Chapman 1999, p. 138). Prior to the passage of this statute and those similar, the water trust model was largely irrelevant due to the rigid appropriation laws requiring water to be extracted and put to productive beneficial (e.g. consumptive) water-use (Harder, 2006). In response to the *1987 Instream Water Rights Act*, the Oregon Water Trust (OWT) was established in 1993 as the first water trust in the western US (Neuman, 2004), and was founded on the principles of free-market environmentalism (Neuman and Chapman, 1999) popularised by Anderson and Leal (e.g. Anderson and Lean, 1991; Anderson and Leal, 2001). The water trust concept has since spread to other parts of the western US, where it has been influential in lobbying and reacting to significant progress in western water law pertaining to instream flow protections (King, 2004).

However, from 1993 to the present, literature detailing the role of water trusts and how they use the market to achieve environmental outcomes, or the wider discussion of private management of water rights for the public benefit, is relatively scarce. For example, only Neuman and Chapman (1999); Neuman (2004); King (2004) and Harder (2006) have explored this concept directly. Neuman and

Chapman (1999) examine the first five years of the OWT and detail the early challenges faced in acquiring water, namely; a lack of reliable market data and water valuation methods, scientific uncertainty regarding the volume and impact of acquired instream flows, ongoing legislative and political hurdles, as well as resistance from irrigation communities and agricultural lobby groups. This research was updated in Neuman (2004), which detailed the accomplishments of the OWT and highlighted the applicability and usefulness of the market-based water trust model, specifically with regards to meeting environmental water demands through flexible and voluntary reallocation programs without the need for protracted litigation or lobbying for new regulations. Significant challenges were also identified, specifically with regards to overcoming the perception of removing water from land and interrupting historic irrigation patterns. Although Neuman and Chapman (1999) and Neuman (2004) offer considerable and valuable insights, both are written from the perspective of an EWT practitioner within the OWT and hence read as a narrative rather than a critical evaluation. Further, they also draw on a single case study, which has explanatory value but limited generalizability outside of the Oregon context.

King (2004) provides a more generalised analysis of the function and actions of water trusts. It is shown that water trusts have played a significant role in the transition from top-down governance approaches towards a voluntary and market-based method of conservation in areas of the western US, and have contributed to the more pluralistic water governance arrangements evident today (King, 2004; Garrick et al., 2011). In particular, King (2004) demonstrates the utility for private water trusts to aid state agencies lacking the necessary human and financial resources to acquire water rights for the environment, by purchasing the water privately and transferring the rights into state ownership to be used for public benefit.[57] Importantly, the study also demonstrates that while water trusts may have emerged from the land trust model, a number of modifications have been made. Namely, water trusts use a wider range of market-based acquisition methods than are available to land trusts, and have also accepted a financially and socially necessary shift from permanent to short-term conservation approaches.

Outside of the context of the western US, little literature exists regarding NGO ownership and management of environmental water. One exception is Hoogesteger (2013b) who examines the role of NGOs in establishing and supporting water user associations in the Ecuadorian Andes.[58] Hoogesteger (2013b) argues that external NGOs play an important role in establishing water user groups through four avenues, namely: a) developing mutual trust between community members; b) facilitating the adoption of a common normative framework (e.g. water rights definitions and water-use rules); c)

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[57] This practice currently occurs in states in which private entities are permitted to acquire instream water rights, but are not permitted to own the newly acquired right. An example of this is Colorado, where only the state has the right to hold acquired instream water rights (Scarborough, 2012).

[58] Note, this paper does not directly deal with NGO environmental water management, but does provide some useful insights into the role of NGOs in localised water management.

connecting community members with external agents; and d) developing local organizational and technical capacities. In essence, it is argued that NGOs play an important role in developing social capital and trust to increase the efficiency of water management. In particular, Hooesteger (2013b) highlights the role that NGOs play in facilitating water management co-ordination at the inter-community level between water user associations (which have in some cases been implemented to manage formally state-owned and operated irrigation infrastructure) on issues that affect multiple users.

Another instance where NGOs have begun to play a supplementary role in water management is the MDB, Australia. A prominent example of EWH NGOs in the MDB is the *Water Trust Alliance* which is a group of local and regional NGOs brought together by the MDB Wetland Working Group in 2010[59] (MDB Wetland Working Group, 2017). Water can be donated to members of the *Water Trust Alliance* and donations over a certain amount are tax deductible (Wheeler et al., 2013d). Using a survey of irrigators, Wheeler et al. (2013b) found that there was a non-trivial portion of irrigators who are willing to donate seasonal water allocations to the environment, although the recognition of environmental need and intention to donate allocations was affected by uncertainty around seasonal water allocations (Wheeler et al., 2013d). More recently, a private NGO called the MDB Balanced Water Fund, administered by the Nature Conservancy and Kilter Rural (a rural investment fund), has become engaged in the environmental water market in Australia (further details in Section 3.8.1). In addition, there has been development of vertical linkages between the federal CEWH and local NGO water holders, such as Nature Foundation SA and the Ngarrindjeri Regional Authority (CEWH & NRA, 2015), both in the Lower Murray region of South Australia. Further details of these linkages were explored in Chapter 4. These developments indicate an increasing scope for non-federal agencies to engage in environmental water purchases and management. If this is indeed the case, it is beneficial to understand how existing NGO EWHs operate in current contexts, such as stewards of environmental water in the western US.

8.3.3 NGO EWHs as Stewards of Environmental Water

Since their origins in the Pacific North West, *water trusts* have expanded in institutional and geographic scope beyond the definition generally adopted in the literature as described in Section 8.3.2. In general, a trust is a fiduciary agreement between one party who holds the legal title of property on behalf of another and who uses it for the benefit of a third party. Presently, there are a number of groups that operate as water trusts in namesake only, for example; The Colorado Water Trust. Examining the structure and actions of NGOs in the environmental water market, it is clear the term *water trust* is somewhat ambiguous. Oversimplification and ambiguity regarding the actual rights and responsibilities of NGO EWHs can lead to a misunderstanding of their role in the environmental water market. It is

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[59] The Water Trust Alliance consists of the Australian Conservation Foundation, the Environmental Water Trust established by the Nature Conservation Council (NSW), Healthy Rivers Australia, the Murray-Darling Association, the Murray-Darling Wetlands Working Group and the Nature Foundation SA.

therefore suggested that NGO EWHs can be more accurately conceptualised as stewards of environmental water. For the purpose of this thesis, NGO environmental water stewards is used to specifically describe the non-state or non-federal actors involved in acquiring water rights for instream flows.

Environmental stewardship theory originated with Aldo Leopold who proposed the term *'land ethic'* reflecting a person's ecological conscious and personal responsibility for the land (Leopold, 1949).[60] Modern definitions of stewardship vary considerably but broadly pertain to those who take direct action to conserve or enhance the condition of the environment (Palmer, 2006). Traditionally, environmental stewardship has fallen within the realm of government responsibility, for example through the protection of national parks. In the US, government environmental stewardship is evident in the *Public Trust Doctrine* which mandates federal protection and management of natural resources for the public benefit (Cohen, 1970; Ryan, 2015). The Federal Environmental Protection Agency (EPA) also provides very literal manifestation of government-led environmental stewardship in the US. However, partly mirroring the emergence of market environmentalism into mainstream thinking (Andersen and Leal, 1991) and the decentralisation of water management (Saleth and Dinar, 2004), the stewardship of environmental water finds itself no longer solely a government pursuit and is instead a complicated landscape of federal leadership, state ownership and subsidiary management and engagement.

Within the water policy landscape, NGO EWHs can be conceptualised as the subsidiary (Young 2010; Garrick et al., 2011). Subsidiary theory refers to the idea that '*action should be taken at the lowest level of government at which particular objectives can adequately be achieved*' (Bermann 1994, p.338). There is considerable overlap between subsidiary theory and the principles of localism, and in particular *new localism*, which is characterised by directing decision-making power to front-line managers, local consumers or resource managers, and immediately effected communities (Stoker, 2004). New localism as a political idea refers to the devolution of power to the local level in order to empower local communities to work towards common national goals within an agreed framework and minimum standards (Stoker, 2004). Subsidiary-style management and local control of resources is consistent with wider international water trends throughout the 1990s, which has seen the partial transfer of state powers to water user associations (WUA) (Meinzen-Dick, 2007; Hoogesteger, 2013) and the greater use of economic-based instruments to allocate water (Settre and Wheeler, 2016). The emergence of WUA is particularly evident in developing nations where donors from the international water community have promoted participatory-based collective action solutions to water management issues (Mustafa et al., 2016).

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[60] A number of theologians argue that the stewardship theory has origins in the opening verses of the book of Genesis (e.g. Bauckham, 2010). However, the religious underpinnings of stewardship theory are not vital for the modern understanding of environmental stewardship as it pertains to water and is therefore omitted from this thesis.

In the context of environmental water management, proponents of localised subsidiary-style management and the localism paradigm cite the subsidiary's responsiveness to local environmental needs in real-time, the ability to experiment to find optimal solutions and increased community participation in natural resource conservation and management (Young et al., 2010; Garrick et al., 2012; Lane-Miller et al., 2013). However, as with water user associations,[61] there are also a number of challenges and critiques associated with subsidiary style environmental water management that warrant further investigation. To untangle the role and challenges of NGO EWHs as stewards of environmental water at the subsidiary level in the western US, and examine the applicability for this style of environmental water management in the MDB, a two-stage methodology is employed, as discussed in the following section.

8.4 Methodology

As identified in the introduction, the aim of this chapter is to: a) examine the role of NGO EWHs in western US environmental water markets, with specific attention to the importance of socially-orientated factors (e.g. social trust); b) derive applicable insights for the MDB context; and c) understand irrigator receptiveness to NGO EWH engagement in the MDB. A sequential research approach was therefore employed, firstly sampling from representatives from active EWHs, water conservancies, state governments, instream flow advocacy groups, academics and water supply agencies, among others (see Table 8.1). Secondly, sampling of irrigators was undertaken in the MDB during 2015-2016. From the MDB perspective, this question is important because despite a long history of irrigator surveys and discussion of environmental water reallocation in the MDB, irrigators have surprisingly rarely been asked about who they would rather sell their water to (e.g. federal, state, council, NGOs). Figure 8.1 depicts the methodology for this chapter.

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[61] Water user associations have been widely advocated as a method of maintaining formally state-managed irrigation and hydraulic systems, predominately in developing nations (Hoogesteger, 2013b). The effectiveness of WUAs has been varied, and there exists a number of harsh critiques of this style of management (see Mustafa et al., 2016 for a review of the critical literature). A number of criticisms particularly highlight the capacity for localised water management to exacerbate existing inequalities along financial, racial and gender lines (Meinzen-Dick and Zwarteveen, 1998).

Figure 8.1 Qualitative Interviews and Quantitative Survey Research Methodology

Qualitative Interviews	Survey Analysis
Western USA	*MDB Australia*

Explore how NGO EWHs use the water market for environmental outcomes. → Develop applicable insights for the MDB context. → Understand MDB irrigator receptiveness to NGO EWH as buyers of environmental water.
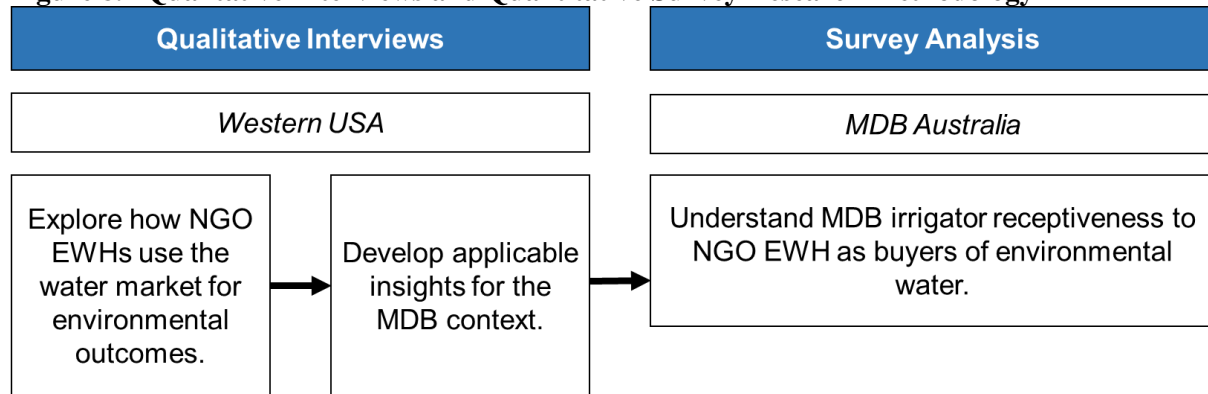
Source: Own Figure.

8.4.1 Qualitative Interviews

To investigate the role of NGO EWHs in EWTs, a series of semi-structured, face-to-face and telephone administered qualitative interviews were conducted between 20th July and 27th October, 2015. Ethics approval was provided by the University of Adelaide Office of Research Ethics, Compliance and Integrity (Ethics Approval No. H-2015-140). Interviews were undertaken predominately in the western US and a small number in north-western Mexico. A total of 49 interviews were conducted, 45 of which were recorded. Interview notes were taken for the four non-recorded interviews.[62] Interview duration was approximately 40 minutes to one hour for face-to-face interviews and 30 minutes for phone interviews. An example of the interview questions is presented in Appendix E.

8.4.1.1 Sampling

Purposive sampling was used to identify and recruit study participants. An initial list of target organisations was developed though a literature review of academic texts, government agency reports, consultancy reports, NGO reports, conference presentations and company brochures. A formal snow-balling method was not employed, but in some cases the sample frame was expanded to include organisations identified by the original participants during the interviews. Interviews were conducted in Arizona (10 participants), California (13), Colorado (10), Oregon (10), and Baja California [Mexico] (2). An additional four interviews were conducted via telephone with participants in Montana (2), Virginia (1) and Baja California (1). However, the location of the interviewee did not necessarily correspond to the geographic scope of their work or knowledge. For example, six participants interviewed within the US reported working predominately on projects in the Colorado River Delta, Mexico. Note that because EWT programs are unevenly distributed between states and basins, this research sought to sample experts from a range of contexts. Because the interviews were semi-structured it was possible to identify saturation levels when no new information was provided to the structured questions asked in every interview. This occurred at approximately 49 interviews at which point no

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[62] Four interview participants declined to be voice recorded during the interview.

more were conducted. Note that saturation may be a function of the organisational sectors sampled and may have been different if an alternative sampling technique (e.g. random sampling) was used. Due to non-responsiveness, the state and federal government sector is not as highly represented as other participant groupings (Table 8.1).

Table 8.1 Participant Sector and Number of Participants

Participant sector	Number of participants (interviews not recorded in brackets)
State government	2
Water conservation board or watershed council	3
Private consulting firm	4 (1)
Academia	5
Policy research centre or think-tank	3 (2)
Advocacy group	3
Non-government organisation	19
Irrigation district or water utility	7
Other[63]	3 (1)
Total Number of Interviews Recorded	45
Total Number of Interviews	49

Source: Own Table.

8.4.1.2 *Data Analysis*

The recorded interviews were fully transcribed. Interview transcripts were thematically analysed using NVivo, a computer-aided qualitative data analysis software program. Non-recorded interviews were used for additional context but were not formally included in the analysis, because the interview notes were not of sufficient length to be comparatively analysed with the interview transcripts in the software package.

8.4.2 Quantitative Survey Analysis

Drawing on results from the qualitative interviews undertaken in the US case study, a series of irrigator survey questions were designed to explore the applicability of the insights gained from the US interviews to the MDB. The questions, detailed in Table 8.2, pertained to MDB irrigator overall willingness to sell entitlements for environmental use; irrigator willingness to sell permanent water to various types of EWHs; and irrigator willingness to sell allocations to the environment. The questions were included in a telephone survey of 1,000 randomly selected irrigators across the southern MDB (sMDB) in southern New South Wales, northern Victoria and South Australia. The survey was piloted and focus grouped first, and subsequently administered by a professional telephone survey company in November-December 2015. The survey contained 83 questions in total, three of which were directly relevant to this study.

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[63] Interviews were conducted with a water rights lawyer, a member of a tribal rights commission and a private irrigation farmer.

Table 8.2 Relevant MDB Irrigator Survey Questions

Q30	Have you considered selling water entitlements to the Federal government for environmental purposes?
	☐ Not interested in selling ☐ I have considered it ☐ I have submitted a tender ☐ Yes, I have sold
Q32	If you did have to sell your permanent water, can you choose your preferred buyer?
	☐ Federal government ☐ State government ☐ Local council ☐ Local irrigator ☐ Interstate irrigator ☐ Non-government environmental organization ☐ Don't know
Q34	Would you consider selling water to the federal government for environmental purposes if you could sell allocations instead of entitlements?
	☐ Not interested in selling ☐ I would slightly consider it ☐ I would seriously consider it ☐ Don't know

Source: Own Table.

8.4.2.1 Sampling

The survey respondents were randomly sampled from irrigator lists in the sMDB irrigation farms and identified as a decision-maker regarding water management on the farm. Survey participants lived in South Australian (209 irrigators), New South Wales (419) and Victorian (372) regions of the sMDB. The sample was representative of the industries across the sMDB and most irrigators reported growing pasture for grazing, pasture for seed hay or silage, cereal including rice, cotton and wheat, and permanent horticultural crops (e.g. viticulture).

8.4.2.2 Data Analysis

Analysis of the survey data was undertaken using various forms of statistical analysis to assess general trends and attitudes. The number of irrigators considering selling water to the government is compared to previous estimates reported in the literature, to identify any trends in the willingness of irrigators to sell water for environmental purposes over time.

8.5 Western US Interview Results: The Role of NGO EWHs

Results of the US interview data analysis suggest that NGO EWHs play a unique role and which has been underrepresented in facilitating environmental water transactions. The following section reports some of the key findings from the interviews relating to the role and challenges of NGO EWHs in the western US.

8.5.1 The 40 Cups of Coffee Principle

With regard to the relationships and interactions between NGO EWHs and local irrigation communities, three key findings are evident from the literature. These are: a) there is a reported preference for engagement with NGOs over higher level government agencies; b) this appears to be caused by an aversion to regulatory repercussions and a loss of local control over water resources; and c) NGOs overcome these aversions through commitment to repeated interaction and negotiation with water rights holders. As shown by Hanemann and Dyckman (2009), a collaborative approach between interest groups changes the outcomes of a negotiation via two mechanisms: a) uncovering new opportunities for action (as discussed in the context of flexible watering arrangements in Section 8.5.2); and b) by fostering an exchange of information and social learning. As the exchange of information and increase in social capital increases, a greater level of mutual understanding is reached potentially leading to an increase in payoffs for both parties (Hanemann and Dyckman, 2009). In the case at hand, the benefits of collaborative action to achieve mutually beneficial pay-offs appears to be well understood by NGO EWHs engaged in buying water. In particular, there appears to be a conscious strategy of building social capital through repeated interaction with rural communities to develop norms of trust and reciprocity, which in turn contributes to confidence that the commitments of a social exchange will be fulfilled (Ostrom 1990; Schneider et al., 2003).

The method of establishing social trust through repeated interaction with local irrigation community networks is referred to by interview participants as the "*40 cups of coffee approach" [Flow restoration director, Oregon]*. This approach describes the idea that the time taken to drink 40 cups of coffee is roughly the time taken for trust and reciprocity to be established between an EWH and a water rights holder, so that a EWT can be agreed. This approach allows the EWH to understand under which conditions (e.g. informal rules) an irrigator would be willing to engage in instream flow transfers. A water reallocation project manager in Oregon describes that:

> *"Before someone becomes willing and open to the idea of relinquishing a property right they need to trust you as a person, so that generally takes a lot of cups of coffee…you're learning about their operation, their family, and they're learning about you as a conservation outfit" [Project manager, Oregon]*

This reported commitment to repeated interaction and the embedment of personnel into the community provides one possible explanation of how NGO EHWs are able to establish trust with water rights holders. This finding supports existing literature regarding the importance of trust in environmental conservation practices (Pannell et al., 2006). In addition, interview data suggests that the commitment to repeated interaction – and hence a presence in the community – enables a greater level of responsiveness to local agri-environmental conditions, thus contributing to an increased ability to achieve positive ecological outcomes. For example, having a physical presence in the community

allows EWHs to become aware when an irrigator has spare water he may be willing to sell or lease for environmental uses. This attribute is particularly important in regional farming areas where social and business networks often overlap. These findings are consistent with stewardship theory regarding resource management, which cites response to local conditions in real time as a key advantage of subsidiary-style management of water resources (Young et al., 2010).

Further, in line with commitment to repeated interaction with the water rights community, the interview data reveals that most EWTs are achieved through negotiation regarding the volume and price of water, rather than through bulk solicitations, auctions[64] or anonymous purchases through the markets. The way in which negotiation on the terms of environmental water transfers may contribute to the construction of shared objectives, which in turn builds social capital, is discussed in Section 8.8. An EWT transaction specialist from an NGO EWH organization summarises the approach to building trust through negotiation:

> *"I hear you you're nervous, I hear you that you don't think this is going to work.*
> *But I also hear that you need tools. So how can we balance that enough for you to*
> *be willing to take a one year risk and try this out?" [EWT consultant, Arizona]*

Within this context, an additional theme emerging from the interview results is increased irrigator willingness to engage with NGO EHWs rather than regulatory agencies. Interview participants from the non-government sectors cited a distrust of federal involvement in private farming enterprises as contributing factor to this attitude. In particular, a "*fear of state or federal intervention and loss of local control" [Stewardship director, Colorado]* exhibited by farming communities was identified in numerous interviews. In exploring this idea throughout the interviews, it is found that a key concern held by irrigators is that by engaging in EWTs with state and federal agencies, it allows regulatory agencies to meter water-use and take subsequent regulatory action such as water right curtailment if a discrepancy is identified. It is found that EWH and government agency interview participates are acutely aware of this aversion held by irrigators, and in one particular case it was found that confidentially agreements have reportedly been signed between NGO EWHs and water rights holders to protect water-use data and engage irrigators in EWTs, as highlighted below:

> *"We're trying to [find] better information about the basin's dynamics and overall*
> *water usage….there's a distinction in working directly with a government agency*
> *verses a NGO because we [NGO] have the ability to sign a confidential agreement*
> *with the rights holder to protect that data" [External affairs director, California]*

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[64] Auctions for water rights are a potential acquisition tool and have been explored in the context of acquiring environmental water (Hartwell and Alwyard, 2012). Interview data shows that *"auctions have been designed in many basins, but implemented in few where you (i.e. the EWH) have the stomach to raise the profile of the operation" [EWT consultant, Oregon].*

8.5.2 Instream Flow Transfer Tools

Methods of reallocating water to the environment are referred to as *environmental transfer tools*. It is found NGO EWHs use a wider range of transfer tools than has been previously reported in academic literature. The interview results suggest that the nature of transfer tools employed by NGO EWHs are a function of: a) irrigator preferences for flexible transfer arrangements, community aversion to *buying and drying;*[65] b) institutional constraints hindering permanent transfers (e.g. high transaction costs); and c) EWH fiscal limitations and the eco-hydrological requirements of stream restoration. In the majority of cases analysed, short-term or informal arrangements are a preferred adaptive management tool for both NGO EWHs and water rights holders, which supports existing literature suggesting the vast majority of instream flow acquisitions have been facilitated by short-term agreements, as previously identified in the literature (e.g. Scarborough and Lund, 2007). For example:

> *"The farmers can participate as an economic tool for their own operation, it becomes a ranch management tool and it also becomes an environmental tool…. it doesn't rely on permanent action but rather a very nimble, flexible and adaptive collaboration"[EWT consultant, Arizona].*

Table 8.3 summarises the transfer tools identified in the interview results. The degree to which these tools are implemented is a function of the enabling legal conditions at a state and regional level and is not applicable in all states. Also note that the definition of 'long-term' leases varies by state. In most cases, long-term transfers refer to leases greater than five years.

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[65] "Buying and drying" is colloquial expression referring to the process of permanently selling water but not the land it is attached to, therefore potentially leaving an irrigator in the tricky situation of having land but no permanent water licence.

Table 8.3 Transfer Tools Employed by Non-government Environmental Water Holders

Transfer Tool	Description
Permanent tools	
Efficiency upgrades	Appropriators conserve water or improving efficiency and permanently dedicate the saved water to the environment.
Permanent acquisition	A water right is purchased from an appropriator and used for environmental watering in perpetuity. The instream right will retain the appropriation date of the original right.
Land and water purchase	Agricultural land with an associated water right is purchased and the water right is retired (left instream).
Water right appropriation	Appropriating water for the environment in under-appropriated streams. The instream right will be awarded a new appropriation date.
Temporary tools	
Annual lease	A water right is leased for environmental use for one year.
Long-term lease	A water right is leased for environmental use for a period ranging between 5 to 10 years. A variation is an opt-out lease, allowing irrigators to choose to opt-out at the end of each year within the lease period.
Split season lease	Water rights are partly used for irrigation and partly leased instream during the same season within one year.
Seasonal storage and banking	Water is banked during wet periods and used to replace instream irrigation abstractions during dry periods.
Fallowing agreement	A forbearance agreement to fallow irrigated land and leaving the volume of historically used water instream. Fallowing arrangements may also be permanent.
Options contracts	
3 in 10 agreements	Water is leased to the environment for 3 out of 10 years to restore streamflow, can be activated by mutual agreement or low-flow triggers.
Low flow trigger agreements	Water is leased to the environment once an agreed low-flow trigger is surpassed.
Groundwater agreements	
Groundwater switch	Part or all of a surface water right is substituted by groundwater pumping.
Payment to cease pumping	A water appropriator is compensated to forego pumping groundwater. This is a form of forbearance agreement.
Urban initiatives	
Urban agreements	Municipal households conserve water to save money, which is donated to purchase instream flows or support habitat conservation.
Land and water-use change	
Point of diversion change	The point of abstraction is changed, typically from a small stream to a main channel to improve tributary condition.
Upstream-downstream change	An upstream water right is exchanged for a downstream water right of equal or higher seniority to maintain river flow for a greater length of the river.
Land use dynamic change	Land use is changed to reduce agricultural water demand and the saved volume is used for instream use.
Water donations	
Charitable donations	Water is donated to a charitable trust for instream use. Water rights donations may be tax deductable.
Estate planning	A water right from a farming estate is donated for instream use upon the death of the water rights holder.

Source: Own Table.

In obtaining water rights for instream purposes an important distinction is the difference between *appropriation* and *acquisition*. Appropriating water involves administratively obtaining a water right in an under-appropriated watershed. This right will be awarded a new appropriation (priority) date as a junior right. Conversely, when water is acquired for the environment it retains the same apriority date as the original irrigation right. As such, interview participants stress the importance of acquiring senior water rights to afford maximum protection for the instream flows. Additional characteristics, such as the location of the water right in the river basin, are described in Section 8.5.4.

The prominent use of intra- and inter-annually flexible transfer tools (e.g. split-season leasing or yearly leasing) reported in the interviews is an example of multi-functional use of the water[66]. Multi-functionality is the idea of using the same property right (in this case a water right) for both environmental and agricultural uses within one year. For example, by timing agricultural water deliveries in such a way as to obtain instream co-benefits (e.g. using conveyance water as environmental base flows) or undertaking a split-season lease to ensure minimum flow volumes during an ecologically crucial period. Multi-functional arrangements therefore may provide irrigators the flexibility to engage in stewardship of the riparian ecosystem without compromising the long-term productivity of the farming operation. Interestingly, the principles of multi-functional transfer tools are consistent with dual-use[67] approaches to land stewardship, which has been used to great effect internationally (Dobbs and Pretty 2004; Dobbs and Pretty 2006; Cocklin et al, 2006). Irrigator preferences for short-term, low-risk transfers was a prominent theme in the interview data. A particular EWT practitioner identified that "*long-term permanent purchasing of water rights is not something that our clients are remotely interested in*" [Program assistant, Arizona].

While short-term flexible transfers are the dominant approach, there are trade-offs compared to more permanent arrangements. For example, permanent acquisitions provide greater ecological security (Garrick et al., 2011), highest long-term flow benefit (Aylward et al., 2016) and long-term planning ability for EWHs. However, from the interviews it is evident that opportunities for NGO EWHs to undertake permanent acquisitions occur relatively infrequently. In cases where permanent transactions have been brokered by NGO EWHs, interview participants identify poor farming land quality, low water-use efficiency, or changing land and water-use dynamics as factors contributing to opportunities for permanent acquisitions:

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[66] Recreational initiatives are identified as avenues to increase environmental water supply. Recreational instream flow transfer tools involve the lease or acquisition of water to support instream recreational activities, such as boating and white water rafting, which can have additional ecological co-benefits.

[67] For example, a farmer may be incentivised to maintain a minimum level of environmental condition on the farm to contribute to aesthetics of the landscape, tourism attractions, hiking trails across farm land, native animal passage. When the farm remains productive there is therefore multi-functional uses of the land.

> *"Where they [EWHs] figured out how to acquire water permanently tends to be*
> *where there is an opportunity to increase efficiency and permanently allocate saved*
> *water instream; or there is changing land use dynamics so the highest and best use*
> *of water suddenly becomes water for instream"* [Program director, Oregon].

In addition to the transfer tools presented in Table 8.3, NGO EWHs also engage in informal forbearance arrangements regarding the non-diversion of water, particularly in cases of high levels of legal and hydrological uncertainty, irrigator hesitancy or high administrative costs. This is because these characteristics, particularly legal and hydrological uncertainty, raise both the risk and transaction costs of engaging in formal acquisition methods.

8.5.3 Disseminating Information

Characteristic of complex environmental policy landscapes (Functowisz, 1999), uncertainty resulting from conflicting, unavailable or poorly distributed information provides a hurdle to adopting new approaches. Poorly disseminated or highly complex water rights information (e.g., tradable volume, trading rules) is identified as a key obstacle for irrigator participation in EWT programs. The interview results are consistent with existing literature, which identifies the rules of forfeiture concerning short-term or informal agreements are often misunderstood by irrigators (Szeptycki et al., 2015).

As previously discussed, local networks remain a key source of knowledge in rural farming communities and action is inspired by knowledge which irrigators perceive as trustworthy (Sligo and Massey, 2007). Building on the results presented in Section 8.5.1, it is found that due to the development of social capital between NGO EWHs and irrigation communities, NGOs can play a key role as conduits of market and water right information. This is consistent with literature identifying the exchange of information as a result of collaborative decision-making and which influences stakeholders' perception of payoffs in a negotiation (Hanemann and Dyckman, 2009), such as an instream water transfer. Market and water right information is distributed primarily through workshop events and local networks. For example:

> *"We [NGO EWHs] go out and put on these workshops…that got us in touch with a*
> *land owner who was interested in trying it [environmental water transfer] out, then*
> *he got in touch with other land owners, his neighbours….people talk"* [Program*
> *assistant, Arizona].*

The role of NGOs in disseminating market information is part of a wider phenomenon identified in the literature. For example, Hoogesteger (2013b) shows that external NGOs can play a role in bringing people into a single normative framework (e.g. understanding and working within the same rules for water rights transfers). Through the lens of social capital, the process of channelling market information from higher order suppliers of information (e.g. government agencies) to irrigators is a process of

bridging, which describes the development of stronger and more selective relationships *between* people, groups or institutions further up or lower down the social or civil ladder (Rydin and Holman, 2004; Brain, 2007; Hoogesteger, 2013a). Viewing the role of NGO EWHs through this lens further reinforces their part in the development of social capital with irrigation communities, and also highlights the importance of social factors in the success of environmental water transactions.

The importance of social factors, such as the provision of trusted market information, in the success of environmental water transactions and increased irrigator participation is exemplified below:

> *"One of the first questions we get is …'you're going to diminish my [irrigator]water right, we'll get zeros in those lease years' and we [NGO EWHs] say that if you are enrolled in a transaction program then the law has protections for you. But until we have that conversation then they probably don't know that….we're educating folks as we go"* [EWT transaction specialist, Colorado].

An additional way water rights information is disseminated to irrigation communities is through the substantial administrative processes and considerable due diligence which are required to be undertaken to facilitate EWTs. Instream flow transactions require substantial hydrologic, financial, administrative and legal research to determine the terms of transfer (Doherty and Smith, 2012), such as the tradable volume that may be acquired without causing injury to downstream water users and the monetary value of the water right to be acquired. When the administrative process is carried out in collaboration with the rights holder, this has the effect of disseminating the market information.

8.5.4 Targeted Action

From the interview results it is found that NGO EWHs predominately target tributary systems for instream flow acquisitions. In the Columbia River Basin, the tributary focus is mandated by the Biological Opinion (US Fish and Wildlife Service, 2000), which ensures the operation of the Federal Columbia River Power System does not jeopardise the continued existence of salmonoid species listed under the *US Endangered Species Act (1973).* In a wider context, targeting EWT in tributary areas is a financially and ecologically strategic approach.[68] This is because NGO EWHs typically have limited capacity to buy bulk volumes of water, and more typically acquire volumes likely to have greater ecological impact in tributary streams than main stem rivers. For example:

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[68] A notable exception to this is the Colorado River Delta Water Trust (CRDWT) who work to restore river flows and riparian habitat along the main stem of the Colorado River which empties into the Colorado River Delta. The CRDWT work in collaboration with both US and Mexican governments to deliver these flows, as detailed in Section 3.9.

> *"We'll get offers sometimes of a small amount of water on a big river, it's a good*
> *water right but we are not interested because we can't actually put enough in there*
> *to make a difference" [EWT transaction specialist, Colorado].*

Targeting tributaries also serves as a proof of concept of EWTs as a method to restore instream flows. Interview participants identified tributary targeted restoration efforts as a means to "*demonstrate environmental return on value*" *[External affairs director, California],* referring to the ability for water rights purchased in tributaries to deliver a higher ecological '*bang for buck'.* Further, tributaries tend to be the source of water for fewer appropriators of water rights and therefore make the process of reaching a consensus in transactions negotiation simpler and more effectively enforced. When water rights are not fully appropriated in tributary systems, there may be scope to appropriate water for instream purposes, but the newly appropriated water right is afforded limited protection as a junior right with a recent appropriation date. Nevertheless, this may be an important strategy to protect unappropriated water from future extractive uses.

8.6 Challenges faced by NGO EWHs

The results from the interviews indicate that NGO EWHs face a considerable number of challenges in facilitating EWTs, namely: a) managing legal and hydrological uncertainty; b) obtaining financial sustainability; c) demonstrating the ecological benefit of acquired water to funders; and d) overcoming the institutional inertia inherent in the agriculturally dominated water resource systems. To a certain degree, the limitations imposed by these challenges are dependent upon the size of the NGO and the external relationships with government and funding agencies. For example, when interviewed NGOs engaged with state or federally-led programs (e.g. Qualified Local Entities (QLE) in *The Columbia Basin Water Transactions Program* (CBWTP)) did not cite financial sustainability as a limiting factor. NGOs partnered with state or federal agencies also did not recognise their democratic legitimacy (i.e. acceptance as an elected authority) as a concern for their operation. This finding is consistent with wider literature, suggesting that legitimacy of institutions is largely derived from their engagement in existing social structures and webs of power (Matthews et al., 2010). The following section discusses the challenges faced by NGO EWHs in greater detail.

8.6.1 Managing Legal and Hydrological Uncertainty

Watersheds are naturally heterogeneous, evolving and uncertain systems. It is found that two primary sources of uncertainty limiting the decision-making of NGO EWHs include the legislative and hydrological conditions under which EWTs occur. In particular, property right definitions, tradable volumes, and the magnitude of return flows were identified as uncertain areas of specific concern.

Western US water laws are complex and laws pertaining to the ownership or beneficial use of instream rights vary considerably between states (Garrick et al., 2009). As discussed in Section 8.5.3 legislative detail regarding transfers to the environment can sometimes be misunderstood in farming communities.

Legislative uncertainty at the state level surrounding environmental water transfers, either perceived or real, was identified as a critical factor inhibiting further adoption of EWT programs. In particular, high levels of uncertainty raise the transaction costs of water acquisitions, which acts as a deterrent for both irrigators and EWHs to attempt transactions. This confirms existing findings citing the level of clarity provided by state regulation as a limiting factor on instream flow transfers (Szeptycki et al., 2015). Arizona provides a particularly good example where legislation allows for the transfer of water rights for environmental use (e.g. *Ariz. Rev. Stat. § 45-172* (2013)) however formal instream flow transactions are rare. In answer to how legislative uncertainty affects environmental water acquisitions, one EWT practitioner reveals: *"we are just leasing water rights until someone tells us we can't"* [Program assistant, Arizona]. Elaborating this point:

> *"Everyone is trying to get a full grasp of what's going on...the legal people and the water rights people don't even necessarily know, we're all testing it out because you have to have the process move forward..."* [Program assistant, Arizona].

The uncertainty surrounding volumetric abstractions and transferability is particularly challenging in non-adjudicated[69] or over-appropriated basins, where the spatial and temporal dynamics, priority and volume of a transferable water right may be unknown (Garrick et al., 2011). In particular, in over-adjudicated basins water users have no incentive to conserve water or engage in EWTs because of difficulties in protecting conserved or environmental flows from re-appropriation downstream.

Hydrological and legislative uncertainty also has an impact on the type of EWTs possible. For example, implementing an options contract with a three in ten year agreement initiated by a low-flow trigger (Table 8.1) requires understanding of historic hydrological conditions to determine a baseline flow, a flow gauge to determine low-flow triggers, and an accurate understanding of the water abstraction pattern in order to delineate between climactic or abstraction induced low-flow scenarios. The setup, monitoring and enforcement transaction costs of such an arrangement is likely to be financially burdensome and risky.

8.6.2 Financial Sustainability

Funding was hypothesised as a limiting factor for NGO EWHs to purchase water for the environment. However, based on the interview data it is evident that there is no clear consensus on the role of funding. Instead, it appears that the limiting role of funding is dependent on the size, ambition and existing partnerships maintained by active NGOs.

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[69] An example of non-adjudicate and highly complex basins are the Gila River and the Little Colorado River in Arizona which involves thousands of water rights claimants and has been in the courts since 1970 (the Judicial Branch of Arizona, 2014).

In general, it was found that the majority of NGO EWHs interviewed accessed funding from state or federal agencies at some stage throughout their activities. Funding was granted in the form of program assistance, grants and low interest loans, as previously documented (Doehety and Smith, 2012). In the Pacific North West region, all NGO EWHs interviewed derived funding from state or federal agencies, predominately from the Bonneville Power Administration[70], which funds ongoing restoration efforts in the Columbia River Basin to preserve endangered salmonoid species populations. In this area in particular, one interview participant suggested that: *"they actually have more funds allocated than they can secure flow transactions for right now."[Streamflow restoration manager, Oregon].*

In addition to funds from state and federal agencies, interview data also reveals an increasing number of cases where environmental water transactions have been funded by private entities through philanthropic donations. Where this has been the case, the short-time horizons associated with the single year cash injections from donors were identified as an obstacle to NGO long-term planning ability. This occurs because in the absence of sequential year funding commitments, there is an incentive to act when funding becomes available, rather than in accordance with long-term ecological goals. The uncertain availability of future philanthropic donations may be a contributing factor to the prevalence of short-term, low-cost transactions, which can be more quickly and easily facilitated as money becomes available. Conversely, interview participants also reported some investors' preference for investment in long-term environmental water acquisitions (e.g. permanent water right purchases), as these can be viewed as more environmentally profitable in the long-run. As mentioned in Section 8.5.2 the specific conditions under which it is possible to acquire permanent rights makes meeting this particular donor attitude challenging. Irrespective of NGO size, location or external linkages, the nature of private funding as an infusion of funds from an outside source was identified by interview participants as a limiting factor. For example:

> *'Once we get to scale that [funding] is going to get harder…I think philanthropy needs to be the short-term solution, but maybe we can create more sustainable funding sources in the long-run" [Program director, Colorado].*

An additional aspect of reliance on external funding is the obligation to design environmental water transfer programs which will continue to attract future investment. For example, salmon in the Pacific North West are critically endangered, socially and culturally important, and provide an icon to which government and private funding is directed. However, salmon are not the only endangered freshwater species requiring conservation in the Pacific North West (US Fish and Wildlife Service, 2015). An WT practitioner surmises the challenges of directing funding to other ecosystem aspects requiring program funding:

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[70] The Bonneville Power Administration is a federal department which markets wholesale electrical power generated by federal hydro-electric projects in north western USA.

"You're not going to get funding to restore stream function but you will get funding to restore fish – salmon resonate with people here" [Streamflow restoration manager, Oregon].

The implications of funding on the design of EWT programs and the target restoration species are not fully evident in the interview data, and future research on the topic would be beneficial to the debate.

8.6.3   Demonstrating Benefit

Difficulty in quantifying the impact of hydrological change in an integrated policy-nature landscape is characteristic of complex socio-ecological systems, owing to the varying, non-linear and lagging timescales of change (Levin et al., 2013). For example, waterbird habitat dynamics change very slowly compared to annual water reallocation decision-making, creating the false impression of ecological constancy with respect to annual hydrological change. This is particularly an issue in the cases observed, where short-term (annual or seasonal) is the dominant method of reallocating water to the environment, thus meaning that the impacts (e.g. ecological change) of annual actions are rarely observed in the year the decision is made. For NGOs dependent on external sources of funding to acquire water rights, demonstrating the benefit of environmental water transaction programs plays a role in developing a track record to secure future grants or investments. The importance of targeting tributary streams for water rights acquisitions, as discussed in Section 8.5.2, is further reinforced by the challenge of demonstrating ecological bang for buck.

Importantly, demonstrating the benefit of environmental water transactions is not only targeted towards funding agencies, but also to the rights holder engaged in the transaction and the wider irrigation community. This is important for two key reasons. First, interview participants identify demonstrating the success of environmental water transactions as crucial to encouraging future irrigator participation in these projects. It is evident there is a belief among NGO EWHs that if irrigators are able to discern benefit from increased instream flow, then they are more likely to participate in restoration programs. For example:

"They [farmers] are closer to the land and the water, they understand the impacts and if they can see it making a difference in a stream that flows right next to them then they are more likely to adopt practices and participate actively" [External affairs director, California].

The second important reason is related to reinforcing the social trust between rural irrigators and EWHs, as discussed in Section 8.5.1. This is because obtaining clear benefits from reallocation (for example, clear flowing stream, reduced salinity, an increase in fish migration survival) reinforces that the obligations of the social exchange have been met, both by the water rights holder who sold the right (who did not illegally abstract water after the right was sold) and the EWH (who used the acquired

water for the stated environmental purpose).[71] It is apparent that the process of reporting, both to funding agencies and irrigators engaged in the transfers, strengthen the bridging bonds[72] and the norms of reciprocity between those engaged in the environmental water transfers.

8.6.4 'Water Buffalos' of the Western US

The difficulty of reallocating water to the environment is, among other aspects, a result of the plurality of legitimate perspectives on use of the resource. One such perspective which is historically prevalent in the western US (and many other countries, such as Australia) has been that beneficial use of water is necessarily productive use. Interview participants in Colorado identify institutions and actors who strongly identify with this perspective as '*water buffalos'*.[73] In general, a *water buffalo* is a water institution which is powerful, slow moving and hard to sway (characteristics of the animal from which the metaphor is derived). The description is applicable in instances where one sector or institution holds the vast majority of power and influences policy to protect these interests. The tendency of *water buffalo* institutions to maintain the status quo manifests as institutional inertia and policy stabilization (Rosenchold et al., 2014). The drive for maintenance of the status quo is evident not only in maintenance of existing policy and institutions, but more widely in the very approach society has adopted in managing water resources through the stabilizing of nature (Doremus and Hanemann, 2008).

In the case of the western US, the agricultural establishment is the dominant owner of water rights and one of the sectors that benefits most from resisting dynamic water management. It is interesting to find that some of the most common *water buffalo* objections to environmental water transactions are common to the Australian case as well. For example; rural lobby groups objecting to permanently removing water from agricultural land (*buying and drying*), disruption to rural economy and loss of farming jobs, the loss of family and generational farming because of senior water rights sales, and the impact of stranded assets on the remaining farming community.[74] For example:

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[71] Note that this is a simplistic representation and does not consider additional complications, such as the actions of third parties which may influence the impact of the environmental water transfer.

[72] In social capital theory, bridging bonds refers to the ties that bind together individuals or groups across institutional, territorial, cultural or other boundaries (Hoogesteger 2013b).

[73] The phrase *water buffalo* was specific to interview participants in Colorado but the idea of agricultural lobby groups reinforcing an institutional inertia was discussed by participants across all states.

[74] See Section Section 3.10 for a discussion of stranded assets in the MDB. While a number of challenges remain, many 'water buffalo' concerns were able to be addressed in the MDB through institutional adjustments, such as the unbundling of delivery rights and water entitlements, which has allowed irrigators to sell water entitlements but retain delivery rights and purchase allocations on the water market to meet their irrigation needs, as well as allowing irrigation districts to levy ongoing irrigation fees against delivery rights (instead of water entitlements) in order to generate revenue from water users to maintain irrigation infrastructure (NWC, 2010).

> *"They [water buffalos] are concerned about too many people within the irrigation district selling out and the district can no longer finance operations and maintenance or the rates go up so much that the remainders are penalised. So there is a dis-incentive and a lot of community pressure not to sell." [Senior researcher, Colorado]*

The historical significance attached to the maintenance of the agricultural system and rejection of permanent water transfers to the environment is described by an interview participant:

> *"The water buffalo have a lot of skin in the game and they have a lot of investment in maintaining the status quo. Not only just the amount of crop yield they get but in all the infrastructure and all the effort that they put in, or their fathers or their grandfathers have put into that [water abstraction] system" [Water resource specialist, Colorado].*

A useful tool to examine the actions of *water buffalos* and the hurdles they present to environmental water transfers is *system-justification*. System-justification is the tendency to support the maintenance of the status quo and resist attempts to change it (Jost et al., 2008; McCright and Dunlap 2011). Jost et al., (2008) find that people of conservative political views typically exhibit higher levels of system-justification. A key example of system-justification is the denial of climate change among self-reported conservative white males in the US (McCright and Dunlap 2011). As farm owners and operators in the US tend to be white males (USDA 2012) in rural regions where populations demonstrate a conservative value bias (Wilkinson, 1991; Phillips, 2007), system-justification may well explain the tendency of *water buffalo* groups and institutions to resist change to the status quo which has historically served their interests and endowed them with social and economic power to construct a system in their favour. In such cases it is likely that the ability for NGO EWHs to acquire water rights or advocate for reform is diminished. Without state and federal governments championing the cause of instream flow restoration and water rights reallocations, it is likely that instream water rights reallocation progress will continue to remain regionally patchwork across the western US, and indeed perhaps internationally.

8.7 Southern MDB Irrigator Survey Results

8.7.1   Irrigator Attitudes towards Types of EWHs

The survey responses regarding MDB irrigators' first preferences for the type of EWH they would most prefer to sell permanent water to is shown in Table 8.4. A further breakdown of results is also shown in Table 8.5.

As can be seen across all states, survey respondents list local irrigators and local council, respectively, as the preferred buyers of permanent (entitlement) water. In particular, the vast majority of irrigators in SA (59%), VIC (73%) and NSW (77%) list other local irrigators as their first preference for buyers of

environmental water. As a second preference, Victorian (21%) and NSW (24%) irrigators rate local councils significantly higher than SA (12%) irrigators. In regional areas of Australia, local councils typically own stock and domestic water rights, which are used to maintain public amenities (e.g. public parks and greenspace) within the council area. The result that local councils rank second as the preferred buyer of permanent water (and higher than interstate irrigators) may indicate that irrigator preferences are not necessarily driven by preferences for water to stay in the farming sector, but rather driven by a preference for permanent water to remain within the local region.

This results demonstrated in Table 8.4 and Table 8.5 are therefore consistent with the principles of localism, which as discussed above is characterised a preference for localised control of resources and the diffusion of centralised power to local actors and citizens (Prattchet, 2004; Stoker, 2004).

Table 8.4 sMDB Irrigator Preferences (n=1,000) for the Sale of Permanent Water Entitlements in 2015-16

	Irrigators' Preferred Buyer of Permanent Environmental Water (Rank)		
	NSW	**VIC**	**SA**
Local irrigator	1	1	1
Local council*	2	2	2
State government	4	4	3
Interstate irrigator	3	3	6
Environmental Organisation	5	5	5
Federal government	6	6	4

Source: Own Table.
Notes: '1' is most preferred, '6' is least preferred.
        Analysis does not include 'no answers' responses.
        ***based on a sample of n=150 only. To reduce survey time some options were omitted after the first 150 participants.**

Notably, after the preferences for local sales of permanent water entitlements, there is a divergence of preferences between Victoria and NSW and SA. Two important features are evident. First, SA survey respondents list interstate irrigators as their least preferred buyer of environmental water, while VIC and NSW irrigators list interstate irrigators as their third preference. This result may be explained by the location of SA at the end of the shared MDB system and the legacy of hostility between upstream (QLD, NSW) and downstream states (VIC, SA) regarding water availability at the end of the system (Sim and Muller, 2004; Dayman, 2017). In addition, NSW and VIC irrigators list the Federal government (who would use the water for environmental purposes) as their last preferred buyer of environmental water, while SA irrigator list the federal government as their fourth preference. SA irrigators also have the highest percentage (10%) for the federal government as their first preference of entitlement sales, compared to NSW (4%) and VIC (5%). The difference between SA, NSW and VIC irrigators in their preferences for selling water to the federal government is consistent with results in

Wheeler et al. (2017), who showed that irrigator location (along with age and climate change beliefs) is one of the key factors influencing irrigators' trust of the federal water agencies.

Table 8.5 sMDB Irrigator Preferences for Buyers of Permanent Water Entitlements

	Least preferred (1)	(2)	Neutral (3)	(4)	Most preferred (5)	Don't know (6)	Total
Federal Government							
NSW	42%	28%	15%	7%	4%	4%	100%
VIC	37%	30%	17%	8%	5%	3%	100%
SA	19%	23%	22%	21%	10%	5%	100%

	(1)	(2)	(3)	(4)	(5)	(6)	Total
State Government							
NSW	9%	43%	29%	13%	3%	4%	100%
VIC	6%	34%	34%	21%	2%	3%	100%
SA	10%	24%	31%	22%	11%	2%	100%

	(1)	(2)	(3)	(4)	(5)	(6)	Total
Local Council[a]							
NSW	0%	8%	21%	39%	24%	8%	100%
VIC	0%	11%	36%	30%	21%	2%	100%
SA	16%	18%	28%	22%	12%	4%	100%

	(1)	(2)	(3)	(4)	(5)	(6)	Total
Local Irrigator							
NSW	2%	1%	6%	4%	77%	11%	100%
VIC	1%	2%	5%	4%	73%	15%	100%
SA	0%	2%	8%	11%	59%	19%	100%

	(1)	(2)	(3)	(4)	(5)	(6)	Total
Interstate Irrigator							
NSW	13%	13%	12%	55%	5%	2%	100%
VIC	14%	19%	14%	43%	8%	2%	100%
SA	32%	27%	12%	24%	4%	1%	100%

	(1)	(2)	(3)	(4)	(5)	(6)	Total
Environmental NGO[b]							
NSW	32%	11%	33%	14%	5%	4%	100%
VIC	40%	11%	22%	17%	7%	3%	100%
SA	34%	18%	18%	14%	11%	4%	100%

Source: Own Table.

Notes: **a) based on a sample of 150.**

b) as in footnote above, using a two sample proportion test between NSW and SA, and between VIC and SA, it is found that SA irrigator preferences for the sale of water to environmental NGOs significantly differ from Victoria irrigators at the 5% significance level, and SA irrigator preferences significantly differ from NSW irrigators at the 1% significance level.

Non-reposes answers were coded as 'I don't know'.

Across all states, NGO EWHs were ranked the fifth preferred buyer of permanent water. SA (11%) irrigators had higher preferences for NGO EWHs as buyers of permanent water compared to NSW (5%) and VIC (7%).[75] In NSW and Victoria, NGO EWHs are moderately more preferred as buyers of permanent water than the federal government, which is listed as the least preferred buyer in these states. The preference of VIC and NSW irrigators to sell water to a NGO EWH over the federal government may have interesting ramifications for how reallocation programs can be best designed to maximise irrigator participation, as discussed further in Section 8.8.

The relatively higher preference for SA irrigators to sell permanent water to an NGO EWH or the federal government for environmental purposes is consistent with ABS attitudinal data, which demonstrates that rates of concern about environmental problems are the highest in the Australian Capital Territory (ACT) (90%) and SA (86%), and among the lowest in NSW (78%) (ABS, 2010a). SA also ranks highest in proportion of adults who felt the condition of the natural environment water bad (35%) or declining (62%) (ABS, 2010a).

8.7.2 Irrigator Preferences for Differing Types of Water Sales to the Environment

As previously mentioned in Chapter 3, the market-based water reallocation to the environment occurring in the MDB has been dominated by entitlement purchases. Table 8.6 shows irrigator willingness to participate in water entitlement sales to the federal government over time.

Between 2010-2011 and 2015-2016, it is evident there is a decrease in irrigators willing to sell to the government for environmental purposes (e.g. an increase in "not interested" responses) in NSW and Victoria. There is no significant change in irrigator willingness in SA during these years. In 2015-2016 the vast majority of surveyed irrigators in all states report they are not interested in selling water to the government. In particular, 81% of Victorian irrigators have no interest in selling entitlements, leaving only 6.1% of irrigators who had considered selling water or 1.3% who had submitted a tender (in addition to the 10.7% who had already sold water). It is important to note the irrigators sampled in the 2015-2016 survey are current irrigators, and so the sample population does not represent the views of irrigators who have already sold their entitlements and exited irrigation. It is known from Wheeler and Cheesman (2013) that about 30% of irrigators who sell permanent water leave the farm. The decline in tenders submitted and the number of irrigators considering selling water entitlements is mirrored the transition to infrastructure-based recovery strategies (Hart, 2015), which are moderately more preferred by irrigators than market-based reallocation strategies (Loch et al., 2014). This decline is also mirrored in the reduced budget and frequency of the *RtB* tenders since 2015. The removal of drought pressure during the sMDB is 2010 is also likely to have played a role in reducing the number of irrigators

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[75] SA irrigators' preference for sale to an environmental NGO is statistically different to Victorian irrigators at the 5% significance level and statistically different to NSW irrigators at the 1% significance level, based on a two-sample proportion test.

interested in selling water entitlements, as well as a reduction in farm debt issues following the end of the drought. However, consideration of selling water will increase when water entitlement prices rise due to scarcity pressure caused by a return to drought and the realised impacts of a changing climate.

Table 8.6 Irrigator Willingness to Participate in Water Entitlement Sales to the Federal Government for Environmental Flows in the sMDB in 2008-09, 2010-11 and 2015-2016.

	Not interested in selling water	Had considered it	Had submitted a tender	Had sold water [d]
2015-16[a]				
NSW	74.9%	10.5%	2.1%	12.4%
VIC	81.7%	6.1%	1.3%	10.7%
SA	61.2%	19.1%	3.8%	15.8%
2010-11[b]				
NSW	57.0%	26.0%	10.0%	7.0%
VIC	59.0%	21.0%	8.0%	12.0%
SA	61.0%	15.0%	10.0%	14.0%
2008-09[c]				
NSW	N/A[e]	N/A	N/A	N/A
GMID (VIC)	62.0%	29.0%	N/A	8.0%
Riverland (SA)	44.0%	51.0%	N/A	6.0%

Source: Own Table.

Notes: a) Data from 2015-2016 irrigator survey; NSW (n=419), VIC (n=372), SA (n=209)
b) Data from 2010-2011 survey results presented in Wheeler et al. (2013); NSW (n= 313), VIC (n=359), SA (n=274). "Had considered it" is a sum of "seriously considered selling water" and "slightly considered selling water".
c) Data from 2008-09 survey results presented in Wheeler et al. (2009). Goulburn-Murray Irrigation District (GMID) is located in Victoria and the Riverland is located in South Australia.
d) Categories are mutually exclusive.
e) N/A indicates no data is available.

As discussed at length in Chapters 3, 4 and 7, an alternative market-based approach to acquiring water for the environment is through the expanded use of water allocation trade for the environment. Table 8.7 shows the percentage of surveyed irrigators in 2010-2011 and 2015-2016 who would consider selling water to the environment if they could sell water allocations (seasonal water rights, comparable to a single year environmental water lease discussed in Section 8.5.2) rather than water entitlements. As can be seen, in 2015-2016 a substantial portion of surveyed irrigators would slightly or seriously consider selling allocations to the environment, particularly in SA (59.2%), followed by NSW (36.9%) and VIC (29%). Comparing irrigator willingness over time, it is found that there has been an increase in the number of irrigators willing to engage in water allocation sales to the environment.[76] This increase

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[76] In 2010-11 the survey asked irrigators to report if they would participate in allocation trade for the environment. In 2015-16, the survey asked irrigators if they would slightly consider it, seriously consider it, or are not interested in selling. A measure of the total number of irrigators interested in participating in allocation trades to the environment is given by the sum of the number of irrigators slightly considering it and those seriously considering it. While it is not a direct comparison to the measure used in 2010-11, it is considered indicative and has some explanatory value.

in willingness is most notable in SA. At present, the CEWH has the ability to buy or sell environmental water allocations provided a number of conditions are met, as detailed in Section 3.11.3 (CEWO, 2016). While it is largely unclear how much water the CEWO has acquired through allocation purchases for the environment,[77] it is likely that the number of irrigators reporting willingness to sell allocations, which is over half in SA (59%), exceeds the current levels of participation. Overall, Table 8.7 indicates that approximately one third (29% in Victoria) and above (36.9% in NSW, 59.2% in SA) would be willing to participate in water allocation sales to the environment. These rates of willingness to participate in water allocation sales are significantly higher when compared to the willingness to sell water entitlements or participation in the *RtB* program (Wheeler et al., 2013).

Table 8.7 sMDB Irrigator Preference for Water Allocation Sales to the Environment in 2010-11 and 2015-16

State	2010-11 [b]	2015-2016[a]			
	Would participate in allocation trade for the environment	Seriously consider it	Slightly consider it	Not interested in selling	*Sum of slightly and seriously consider it*
NSW	29%	16.7%	20.2%	62.6%	36.9%
VIC	27%	9.4%	19.6%	70.7%	29.0%
SA	41%	26.7%	32.5%	40.7%	59.2%

Source: Own Table.
Notes: a) Data from 2015-2016 irrigator survey; NSW (n=419), VIC (n=372), SA (n=209)
b) Data from 2010-2011 survey results presented in Wheeler et al. (2013); NSW (n= 313), VIC (n=359), SA (n=274).

8.8 Discussion

Water reallocation is a socially and politically challenging task with no easy answer of how it should be accomplished or who should carry out the acquisition. This study has attempted to untangle the role played by non-government environmental organisations and their use of the water market to acquire water for the environment.

The interview data suggests that NGO EWHs play a unique and previously under-analysed role in facilitating environmental water transactions, and highlights the importance of social capital and trust in environmental water reallocation programs. Attention to the development of social trust and commitment to repeated interactions between NGO EWHs and water rights holders appears to positively affect irrigator participation in reallocation programs, by placing the EWHs in a trusted position to provide market information, negotiate transaction arrangements and fulfil norms of reciprocity. It is also evident that this subsidiary-style management approach to reallocation suffers from a number of challenges, many of which are similar to those documented by Neuman and Chapman

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[77] The CEWO publically reports how much water has been acquired in progress towards the 2,750GL recovery target, although limited information is publically available regarding the CEWH seasonal trading behaviour.

(1999) and Neuman (2004), including managing uncertainty at a hydrological and institutional level, institutional inertia, financial sustainability and overcoming or adapting to traditional farming values opposing the 'removal' of water from land. The similarity in findings highlights that although significant progress has been made in market-based water reallocation to the environment, a number of central issues are still yet to be resolved and further research and policy development are required. Success in overcoming these challenges and increased capabilities appear to be enhanced through vertical linkages with wider power structures, such as federally led and financed reallocation programs.

The irrigator survey data in the sMDB indicates that preferences for the local management of water resources are also prevalent within irrigation communities in Australia, although there are some differences depending on the location of the irrigator. It is also evident that irrigators would rather sell water allocations than water entitlements to the environment, along with a slight decrease in the number of willing water entitlement sellers in 2015-2016 compared to 2010-2011. As reported in Section 8.7.1, the fact that NSW and Victorian irrigators[78] (who own a very substantial proportion of irrigation rights in the MDB) list the federal government as their least preferred buyer of water entitlements may be a cause of concern for the future viability of the federally-led reallocation program in the MDB. New institutional arrangements or additional mechanisms may therefore be beneficial in increasing irrigation participation in future market-based reallocation programs in the MDB. Based on the similarities in regional community preferences for local control of water resources and the increasing scope for NGO EWH engagement in the market, insights from the western US may well be applicable to addressing this future challenge in the MDB. The following section discusses the key findings from the interview and survey data in more detail.

8.8.1 Localism and Social Capital

The interview results present a strong case for the importance of social capital, and in particular the role of local values and trust, in facilitating environmental water transactions. Based on the results presented above it appears that, where NGO EWHs[79] have been successful in facilitating environmental water transactions, transactions have occurred when EWHs are able to offer flexible transfer tools, commit time to repeated interaction with water rights holders and trades through negotiation, and sustaining local values in water resource decision-making.

Through the lens of social capital, it is evident that NGO EWHs develop the formation of outward-looking social capital, which refers to the social and civil ties with groups or individuals outside of the immediate community across institutional, technocratic, territorial boundaries, or other boundaries

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[78] Note that irrigators in the sample are those who have either: a) not sold their entitlements in the previous entitlement tenders, or b) sold some (or all) of their entitlements and remain irrigating because they owned surplus water, purchased water allocations, or employed other farm management strategies to remain irrigating. As such, irrigators who had sold all of their water entitlements and left the farm in this time-period are not captured in the sample.

(Hoogesteger, 2013a). The relationships developed between NGO EWHs and the irrigation community can be conceived as bridging bonds that manifest as strategic alliances, which one party forms to solve a specific problem (Rydin and Holman, 2004). In the case at hand, the specific shared problem is the need for efficient and effective instream flows without compromising agricultural productivity. The investment in the *40 cups of coffee principle* and commitment to the development of social trust can therefore be viewed as the construction of bridging bonds. It is suggested that the existence of these bridging bonds, as well as the physical presence within the landscape, provide NGO EWHs with the legitimacy to carry out environmental water transactions. The social trust approach, involving collaboration between irrigators and EWHs in environmental water transactions, is consistent with the theory presented in existing literature. The interview results confirm the existing literature, suggesting that formal and informal interactions with a bridging institution can have the potential to increase policy effectiveness, rather than authority-driven change (Schneider et al., 2003). This is achieved by the ability for bridging institutions to potentially span across conflicting ideologies at the sectorial, federal and traditional lines of advocacy, in favour of a development of common perspectives and cooperation on shared problems (Schneider et al., 2003). Overall, the results of these interviews and the existing literature confirms what is known about how American irrigators perceive collaborative and authority-driven change: that the success of policy and the success of collaboration have different yard sticks – policy effectiveness is measured in private benefit, while collaboration is measured by social capital and public community benefits (Lubell, 2004).

In addition to the bridging relationships between irrigators and EWHs, the interview data reveals the importance of strong vertical linkages with higher order agencies (e.g. state and federal government) for the legitimacy and scope of NGO water transfer operations. These linkages are particularly evident in the *Columbia Basin Water Transaction Program* where engagement with federal funding agencies has had a positive impact on the ability for NGO EWHs to achieve substantial conservation goals and the water rights purchase expenditure in the order of US$20 million dollars (Garrick, 2015). Expenditure of this kind without the support of government agencies and access to federal coffers would be unlikely. In cases with fewer vertical funding linkages, financial wealth and sustainability may become an increasing challenge for NGO EWHs engaged in acquiring rights, especially in arid and semi-arid regions where water will become more scarce and variable due to climate change (IPCC, 2007). More generally, it is noted that engagement with wider power structures may also improve the creditability of NGOs acting in the water market, who may otherwise suffer from the criticisms of lack of democratic legitimacy by critics of third-party governance arrangements (King, 2004). The strong vertical linkages between NGO EWHs and federal and state agencies, particularly in the Pacific North West, reinforces wider observations in the literature that localised water resource decision-making and local partnerships are rarely absolute.

8.2.2 The Limits to Localism

Although interview and survey data highlight the importance of localism and the role of social capital from an irrigator and non-profit perspective, there is considerable literature to suggest there are dangers in the devolution and decentralisation of responsibilities. For example, Brown and Purcell (2005) argue that there is nothing inherently more ecological or socially just in local scale management, and that in fact such arrangements can become susceptible to the *local trap*. The *local trap* refers to the danger of decentralising and downscaling responsibilities to the subsidiary without the necessary resources to uphold them (Brown and Purcell, 2005; Born and Purcell, 2006; Bakker and Cook, 2011). In decentralizing water management to the subsidiary scale, there is also a risk that local institutions will be dominated by the powerful minority, such as those more financially or socially empowered to lead collective decision-making (Mustafa et al., 2017). In the case of Australia's MDB, this is somewhat evident in the *local voice* being appropriated by agricultural lobby groups to direct policy towards their own best interest (Grafton, 2007; Loch et al., 2014). In cases where there is extreme opposition between stakeholders (e.g. between agricultural and environmental interests), Hanemann and Dyckman (2009) argue that a bargaining arrangement among water user groups is unlikely to be reached and advocate for stronger state organization to address water management conflicts. Hanemann and Dyckman's (2009) argument for state organisation to resolve an ongoing conflict is reminiscent of Australian federal *Water Act* (2011), which required the transferal of some state powers to the federal government in order to better manage water competing water resources in the MDB. In addition, there are also water-specific issues that may affect the localised engagement of environmental water, for example: the physical and technological challenges posed by modern and large scale irrigation infrastructure; conflict between upstream and downstream watering priorities; and a poor alignment between localised and basin scale objectives (Garrick et al., 2012). These water-specific issues are particularly pertinent when considering the impacts of climate change on water resources in arid regions, such as the western US, where current physical and institutional water resource systems are highly susceptible to the impacts of a more variable and less wet climate future, with adaptation needed at all scales and boundaries (Doremus & Hanemann, 2008). As a result of these challenges, among others, localism is rarely absolute (Born and Purcell, 2006) and perhaps nor should it be, given the essentialness of water and the complexities in its management.

8.2.3 Flexibility and Multi-Functionality

Closely linked with aspects of localism and collaboration is the type of transfer tools provided by NGO EWHs. In rural farming communities it is reasonable to assume that a common goal involves the maintenance of irrigated agriculture. In undertaking water reallocation programs, the problem therefore becomes one of structuring environmental water transfer tools that achieve ecological flow restoration targets, without imposing undue externalities on the irrigation community. While NGO EWHs engage with a range of transfer tools, including permanent purchases and long-term arrangements, it appears

flexible, short-term, low-risk and multi-functional transfer tools are the dominant strategy implemented by NGO EWHs. This approach reflects the need to engage the irrigation community in the environmental water transfers, in order for reallocation to be successful. By lowering the stakes of the transaction (e.g. through short-term agreements) it appears that this approach also has the benefit of providing a means for individual irrigators to experiment with environmental water transfers and gain water market experience. These type of arrangements can help to avoid the somewhat socially unpalatable problem of leaving an irrigator with land but no water with which to irrigate. As has been suggested in the Australian water reallocation literature, shorter-term environmental water transfers can also aid in easing the structural adjustment within irrigation communities and reduce some of the negative externalities associated with permanent reallocation strategies (Wheeler et al., 2013).

As previously discussed, the principles of multi-functionality refer to the dual use of a single property right (in this case a water right) within the same year (see Section 8.5.2). To date, multi-functionality has been widely discussed and implemented in the land stewardship field. As with water rights, agricultural land has historically been managed for a single dominant private use (i.e. production). However, there has been increasing evidence to show that adjusting the rural landscape to meet consumer demands for nature-based activities can provide irrigators with an opportunity to diversity income, foster farm sustainability and promote the generation of ecosystem services (Schutz, 2011), such as recreation, tourism and sense of place. Additional literature suggests there may be considerable ecological benefits to be gained by incentivising a minimum level of environmental stewardship in multi-functional rural landscapes (Dobbs and Pretty 2004; Dobbs and Pretty 2006; Cocklin et al, 2006).

The provision of multi-functional and flexible transfer tools provided by NGO EWHs in the western US suggests that this approach is being implemented, at least in part, in environmental water stewardship. However, unlike the western US reallocation effort, the concepts of multi-functionality in water ownership have not been fully integrated into the Australian water reform landscape, which is dominated primarily by permanent reallocations from irrigation to the environment. Given the results suggesting that irrigators have a preference for the sale of allocations rather than entitlement sales to the environment, or other short-term reallocation arrangements[80] (Wheeler et al., 2013), it may be the case that multi-functionality in the water policy landscape is a worthy objective. Potential benefits in the expansion of water market flexibility supports existing literature which demonstrates the environmental benefits of annual environmental water trade (Kirby et al., 2006, Connor et al., 2013, Wheeler et al., 2013). The results in Chapter 7 have also shown this. Commitment to multi-functional use of water rights may temper the 'agriculture versus the environment' mentality that has prevailed in MDB public policy to date. However, commitment to such a strategy would require a restructure of the reallocation approach and renewed commitment to monitoring and enforcement (see Section 8.6.1

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[80] For example, entitlement leasing, options contracts.

regarding the challenges of monitoring complex water transfers in uncertain hydrological systems), likely at considerable extra costs.

In addition, from the interview results it appears that informal forbearance arrangements go hand in hand with NGO EWH brokered environmental water transfers. The interview data reveals that informal forbearance arrangements (i.e. verbal agreements to not abstract water) are being employed in cases where administrative transaction costs or legal and hydrological uncertainties are high. While these informal arrangements may be preferred by irrigators as a means of experimenting with EWTs and by NGO EWHs due to the low transaction costs (i.e. limited administration and legal hurdles), they also pose the non-trivial issue that there is no legal protection for rights left instream under these arrangements. A related issue is the question of monitoring and enforcement of informal forbearance arrangements, as well as how these forbearance volumes are reported and accounted for in catchment water-use records (where they exist). As recently highlighted in the MDB case, where Commonwealth owned environmental water was allegedly abstracted illegally by NSW cotton growers (Besser, 2017; Dayman, 2017), monitoring and enforcement is a keystone practice in ensuring ecological outcomes and institutional integrity of water acquisition programs. The reliance on informal forbearance arrangements, while perhaps useful in the initial stages of instream flow protection, does not provide a viable large-scale method of environmental water conservation.

8.2.4   The Potential for Non-government Environmental Water Holders in the MDB

It has been argued that the federally led top-down approach to reallocation in Australia has not properly accounted for local values and community participation (Tan, 2012) and has failed to *'win the hearts and minds of rural communities'* (Evans and Pratchett, 2013, p. 541). As a result, existing literature has shown that irrigators in the MDB do not necessarily trust national water agencies to manage water in a way that helps their community solve environmental flow problems (Wheeler et al., 2017). The result of the survey analysis revealing that sMDB irrigators list the federal government among their least preferred buyers of environmental water is therefore not surprising. The lack of trust and low irrigator preference for engagement with the federal government in the MDB begs the question of what can be done to ensure the goals of the water reallocation are met and enforced.

Although the details described in Sections 8.5 and 8.6 are specific to the western US, the concept of NGO EWHs as stewards of environmental water could resonate in Australia's MDB. For example, the MDB survey analysis shows that there are a previously unidentified percentage of irrigators (between 5% in NSW and 11% in SA) who list environmental NGOs such as water trusts as their most preferred buyer of environmental water. This is a comparable figure to the percentage of irrigators who most prefer the federal government (between 4% in NSW and 10% in SA), noting that the options are mutually exclusive.

To a certain extent, this opportunity has already been identified by the international conservation community, namely; the MDB Balanced Water Fund administered by The Nature Conservancy and Kilter Rural (an investment fund), which aims to use 80% of allocations to derive profits for investors and dedicate the remaining 20% to instream use (Kilter Rural, 2016b). In 2015-2016, the Fund raised $22 million in equity and $5 million in debt to privately purchase water entitlements (Kilter Rural 2016a) – a sum that would unlikely have been committed if willing irrigators had not already been identified. The engagement of the private sector in the market for environmental water indicates that there is indeed a shift in the composition, roles and responsibilities of EWHs in the MDB.

Given the reported irrigator unwillingness to engage with federal environmental water agencies, as irrigator survey data may suggest, then it would be in the best interest for the federal government to realign with irrigator preferences. One way this may be achieved is through the renewed engagement with localised and non-governmental entities, to improve irrigator engagement in buy-back schemes. Drawing on examples from the international literature, it is evident that NGOs engaged in water management play a key role in assisting relationships between communities of water users and external agents (e.g. government agencies, donor agencies, other NGOs) by creating bridging bonds between the groups (Hoogesteger, 2013b). Isolated examples of federal government and local NGO partnerships already occur in isolated examples in the MDB. Two of these examples occur in SA, where irrigators reveal the highest preference for environmental NGOs as buyers of permanent environmental water. One such key example is the recent development of an agreement between the Commonwealth Environmental Water Holder (CEWH) and the Ngarrindjeri Regional Authority (NRA). The agreement provides for the transfer, delivery and monitoring of Commonwealth environmental water in the Lower Murray by the NRA (CEWH and NRA, 2015). In addition to developing partnerships between federal and local agencies, the relatively high preference for environmental NGOs as buyers of environmental water in SA indicates a potential scope for non-federally affiliated EWHs to engage with irrigators to acquire water for the environment, as previously suggested in Chapter 4. However, a number of institutional and hydrological challenges would need to be overcome to implement this. Overall, the increased engagement of federal bodies with local organizations is supported by existing literature, which suggests there is scope for increased subsidiary-style management of water resources in the MDB, subject to overcoming a number of challenges and hurdles (Garrick et al., 2012).

8.9 Conclusion

This chapter has reflected on the existing literature and presented the results from a series of 49 detailed qualitative interviews, as well as the analysis of water sale preferences of 1,000 irrigators in the sMDB concerning the role, challenges and irrigator perception of NGO environmental water holders engaged in buying water for the environment.

Interview data from the western US suggests that social-orientated influences are of significant importance when implementing market-based water reallocation transfers. In particular, it was identified that the ability of NGO EWHs to dedicate time in developing trust and norms of reciprocity with rural water rights holders contributes to the success of water transaction programs, as does the range of environmental water products available to irrigators that allow flexible and multi-functional transfer agreements to achieve environmental co-benefits of agricultural water-use. Decrypting and disseminating market information was also identified as a positive trait of successful NGO EWH engagement in the instream flow water market. Results therefore suggest that market transaction programs, state administered or otherwise, aimed at improving instream flows or environmental conditions may benefit from better incorporating avenues to build social capital (e.g. time to build relationships, collaboration, trust and capacity) to improve environmental outcomes. Further, as was demonstrated in Chapter 7, the proposed carbon-water trading strategy may provide a viable way for NGO EWHs in the MDB environmental water market to facilitate their annual allocation (i.e. short-term reallocation) purchases to return water to the environment.

In the MDB, survey results indicating irrigator preference for short-term and flexible arrangements is a key insight for the future design of market-based reallocation programs. In particular, irrigator willingness to engage with non-government EWHs has not been reported before, and provides insight into the possible role of NGOs in the MDB to increase irrigator participation in the sale of water to the environment. This result also provides an interesting avenue for Australian policy to explore vertical collaborations between NGO and government agencies under future reallocation programs, as has been implemented successfully to generate a range of benefits in the *Columbia Basin Water Transaction Program* in the US.

However, it is noted that while there are a number of potential gains from NGO engagement in the western US and potentially in Australia, the engagement of the private and non-profit sector is not a substitute for federal leadership in either case. Rather, the essential nature of reliable urban and irrigation supply and the complexities of water management, not to mention the considerable capital and operational expenditure of water management and delivery, indicate that restoring freshwater ecosystems is best achieved in a context of federal leadership and financial commitment to restoring the balance between consumptive water uses and the environment. However, in the current political environment in Australia and the US, which lacks consistent federal commitment to environmental restoration and protection, there may be scope for non-governmental organisations to act within the framework of market environmentalism, through environmental water transaction programs, to achieve positive social and environmental outcomes. As documented, there are a number of limiting challenges that remain before these gains could be made.

CHAPTER 9: Conclusions and Policy Implications

This thesis has presented a multi-method exploration of the use of water markets for positive environmental outcomes. This chapter provides a summary of the thesis along with key findings. The methods and results contained within this thesis make a number of contributions to the literature, which are detailed in this chapter. Based on these results a series of policy recommendations and considerations are made. Lastly, thesis limitations and avenues for future work are identified and discussed.

9.1 Overview of the Thesis and Key Findings

Table 9.1 summarises the key thesis findings, which are elaborated on further below. The relationship between these findings and the implications for policy development is discussed in the following sections.

Table 9.1 Summary of Key Thesis Findings

Chapter	Finding	Method
Two	MDB water management has transitioned from a regulation and engineering based approach to more pluralistic governance arrangements.	Literature review
Three	There are a number of unresolved issues associated with the federally led environmental water reallocation strategy in the MDB that warrant further consideration.	Literature review
Four	Environmental works and measures are not necessarily synonymous with sustainability and have been implemented at significant monetary and ecological cost in the CLLMM.	Case study
Five	Hydro-economic modelling has focused disproportionately on agricultural production and profitability and has not adequately represented environmental dynamics, or the ecological and economic benefits derived from improved environmental water availability.	Literature review
Six	Integrated hydro-economic modelling including dynamic representation of carbon storage and sequestration provides a valuable approach to evaluating annual reallocation trade-offs between environmental and consumptive water uses.	Hydro-economic modelling
Seven	Strategic annual water allocation purchases for the environment can have considerable benefits to floodplain environments and marginally alter native vegetation carbon dynamics.	Hydro-economic modelling
	An EWH may be able to offset or reduce the cost of annual water allocation purchases through the generation and sale of improved carbon storage in native vegetation caused by overbank flooding.	
Eight	When an NGO EWH buys water for the environment, social factors such as trust, localism and flexibility play a significant and over-looked role in the success of environmental water transaction.	Qualitative interviews
	MDB irrigators display a clear preference for localism when considering the sale of water entitlements and rank the federal government among their least preferred buyer of environmental water in Victoria and NSW.	Large-scale irrigator survey

Source: Own Table.

Following the introduction in Chapter 1, Chapter 2 provided a detailed description of the MDB and an overview of water resource policy in the area. It was shown that, overall, MDB water policy has undergone a transition from top-down regulation and engineering based approaches to a more pluralistic approach involving the use of markets to allocate water between users. However, infrastructure development still plays a role in the MDB, as is evident with the recent policy and budget emphasis on reallocation through irrigation infrastructure upgrades, along with the wide use of environmental works and measures in the CLLMM.

Having established the current state of water policy in the MDB, Chapter 3 provided a more detailed exploration of how water markets and property rights are currently used to reallocate water to the environment. This chapter included a rationale for market-based water reform and an overview of select international case studies where environmental water markets are currently implemented. It is shown that in the MDB, the primary market-based method of reallocation is federal entitlement recovery. A number of challenges to entitlement recovery were identified, suggesting further investigation in terms of where the potential expansion of environmental water markets is warranted.

To better examine the concepts presented in Chapters 2 and 3, Chapter 4 considered a detailed case study of the Coorong, Lower Lakes and Murray Mouth (CLLMM) at the terminus of the MDB. It is demonstrated that although water markets have played an increasing role in securing environmental flows for this end-of-system wetland, there is still a significant reliance on infrastructure solutions and environmental works and measures to achieve sustainability. Reviewing the history of the CLLMM it becomes apparent that infrastructure interventions have been used cyclically, first causing ecological degradation and then as the solution to it. This chapter concludes with possible future management solutions, which are examined further in subsequent chapters.

Departing from contextual information and the first case study, Chapter 5 presents a detailed literature review of hydro-economic modelling (HEM) in the MDB. It is shown that hydro-economic modelling has played a vital role in informing and interrogating MDB the water resource policy examined in this thesis. However, it is found that the literature focuses disproportionately on irrigated agriculture and does not give due attention to modelling the ecological changes occurring in the MDB. Further, when environmental representation has been included in the HEM approach, it has been largely conceptual in nature and has not provided adequate representation of the complexities of environmental dynamics. This therefore forms a gap in how water policy is modelled and evaluated.

Chapter 6 details the construction of the GAMS model, which is developed to address this gap. The HEM is made up of the sub-models: a) catchment hydrology; b) floodplain carbon storage dynamics and carbon valuation; c) water price; and d) an annual reallocation heuristic. The model is applied to the Murrumbidgee catchment and simulates the annual decisions of an EWH with a mandate to improve floodplain conditions through annual reallocation of water from consumptive users to the environment.

Chapter 7 presented the results of the model outlined in Chapter 6. It is found that in the Murrumbidgee catchment, the actions of an annually trading EWH have the capacity to reduce the period of time that environmental watering requirements are not met and improve modelled floodplain conditions relative to the baseline scenario. It is found that the increase in carbon storage in floodplain River Red Gums, caused by improved overbank flooding, may be of sufficient value when sold as credits on the carbon market to offset the cost of purchasing supplementary allocations for the environment. This uncovers a potential carbon-water trading scheme, which may provide a novel revenue stream for a self-financing EWH. The opportunities to implement the proposed scheme occur under specific hydrological and economic conditions, including a minimum carbon price of $30/tCO_{2e}$ and specific hydrological sequences leading into drought conditions, recovering from prolonged drought, and extending naturally occurring flooding. The viability of the proposed strategy is found to be sensitive to a number of assumptions and variables, including the price of carbon, surface water availability, and the ability of the modelled EWH to account probabilistically for future hydrological and economic conditions. To implement the proposed strategy in the MDB, a number of challenges would need to be overcome, specifically pertaining to higher prices of carbon and the inclusion of existing native forests in carbon credit accounting schemes.

The final chapter departs from the MDB context and examines the use of water markets by NGO environmental water stewards (e.g. water trusts) in the western US. It is shown that environmental stewardship in the western US is no longer solely a government pursuit and that NGOs play a unique and under-analysed role in the water market as a subsidiary actor. Specifically, it is shown that social factors such as trust, norms of reciprocity, flexibility and multi-functionality of water products are key characteristics of successful water transactions brokered by non-government environmental stewards. Applying insights from these results to the MDB case study and combining with a survey analysis of sMDB irrigators, it is found that irrigators display a clear preference for the sale of water to non-government environmental actors ahead of state and federal EWHs. This is in line with wider trends of farmer preference for localism and flexibility in water reallocation schemes - reflected in irrigator preferences for selling allocations rather than entitlements to the environment. Overall, it is suggested that in a climate of weak federal commitment to environmental protection, non-government EWHs may be able to access existing water market structures to achieve positive ecological outcomes and social gains, perhaps though employing the carbon-water trading strategy for self-financing, as proposed in Chapter 7.

9.2 Contributions to the Literature

This thesis makes a number of methodological and applied contributions to the literature. Firstly, in addition to providing a detailed understanding of water reform history in the MDB, the thesis provides two comprehensive reviews. The first is the review of policy and management in the CLLMM, which catalogues policy and infrastructure intervention since the construction of the initial South East

Drainage Scheme commencing in 1862. This review contributes to the literature by exploring the trends in policy emphasis and implications for the ecological health of the CLLMM, as well as calculating the costs of various interventions. It also contributes to the existing literature (e.g. Newman, 2000; Kingsford et al., 2009; Kingsford et al., 2011; Pittock and Finlayson, 2013) on water policy in general by examining the future management of the CLLMM – categorising all previously abandoned and newly proposed interventions which are likely to become relevant upon the return to drought conditions. The second major review of this thesis is a systematic analysis of MDB hydro-economic models including their key findings, identification of subject gaps and potential methodological avenues for advancement. To the author's knowledge, this is the first comprehensive review of all hydro-economic models in the MDB. It is therefore a useful contribution to the literature by demonstrating issues that remain to be addressed.

The formulation of the hydro-economic model was developed to understand the environmental benefits of potential trading of water allocations and carbon credits, and also makes a number of methodological contributions. As noted, hydro-economic modelling has traditionally not given full representation to ecosystem service dynamics beyond irrigation and farm profitability (Settre et al., 2017). The model developed in this thesis has extended traditional approaches to modelling biophysical responses by advancing the representation of ecological functions, and hence accounting for the complex, non-linear and temporally variable nature of forestry dynamics and carbon sequestration. Subsequent results using this more sophisticated representation of environmental dynamics provide a more variable reallocation scenario, better suited to the environmental watering requirements of native floodplains, thus demonstrating the applicability of the approach. Conversely, ecological modelling rarely attributes estimates of the economic value of changes in a biophysical system. Therefore this model makes a further contribution by valuing the annual incremental changes in the River Red Gum carbon stocks in monetary terms, allowing consumptive and non-consumptive opportunity costs to be valued and incorporated into annual trade-off decisions.

Further, while a number of studies have investigated the interaction between water supply and ecosystem service generation (e.g. CSIRO, 2012; Banerjee et al., 2013; Bark et al., 2016), to the author's knowledge this is the first model to look in detail at the trade-offs involved in optimizing water reallocation and the carbon storage and sequestration ecosystem service, in an annually dynamic simulation approach. In the few cases where carbon-water trade-offs have been examined (Schrobback et al., 2011), the focus has been distinctly on the negative implications of carbon farming on overland flows into the river. This thesis looks at the issue from a new angle and demonstrates the carbon benefits of existing forests rather than the cost of new carbon farming plantations. Overall, this approach has resulted in a very high level of integration of hydrology, economics and ecology, in a model capable of evaluating some of the complexities of water allocation decision-making. In doing so, it has also been demonstrated that simulation modelling of ecosystem service values in a hydro-economic model is both

a valid and useful means of investigating consumptive and non-consumptive water trade-offs in the MDB.

The results of the hydro-economic model also make a number of policy contributions. For example, it was found that annual environmental watering, facilitated through the purchase of water allocations for the environment, can have positive environmental benefits for vegetated floodplains (as has been found in previous studies such as Connor et al., 2013 and Ancev, 2014). Although other literature has identified the potential for a self-financing EWH to fund future allocation purchases through the strategic sale of allocations when not needed by the environment (Kirby et al., 2006; Connor et al., 2013); this thesis extends these findings by demonstrating that there may be new and previously un-identified avenues for a self-financing EWH to generate revenue, through the generation and sale of carbon credits by stimulating floodplain biomass growth. This demonstrates that purchasing water allocations for the environment can be a cost-effective means of reallocating water to the environment, if purchase costs are offset through the sale of other marketable ecosystem services. This result adds weight to the substantial literature querying the cost-effectiveness of current reallocation policy focus (PC, 2010; Adamson and Loch, 2014; Loch et al., 2014b; Grafton, 2016) and supports existing literature calling for diversification of water products for the environment (Wheeler et al., 2013a).

In considering the possibility for a self-financing EWH, although previous studies have demonstrated that cost-effective annual reallocation can be achieved, they have given limited attention to the specific conditions under which reallocation is most ecologically and economically effective (Kirby et al., 2006; Wheeler et al., 2013a; Ancev, 2014). This thesis has identified the economic and hydrological conditions that best enable cost-effective reallocation. Assuming a forward-looking EWH, this occurs when water prices are relatively low and environmental watering requirements are not fully met, namely in the years before drought, during drought recovery, and opportunities to extend floodplain inundation during wet periods. There appears to be limited opportunity to purchase allocations from the environment during protracted drought conditions as a result of high water prices reflecting scarcity. The results of the HEM were also subject to extensive sensitivity analysis and uncertainty management, which go beyond the usual current practice employed by models of the MDB (Settre et al., 2017). This demonstrated the considerable uncertainty faced by modellers and decision-makers in the MDB, and contributes to the discussion on the difficulty and implications of carrying out post-modelling testing of highly integrated and complex models (Van Asselt and Rotmans, 2002; Letcher and Jakeman, 2003).

Overall, the findings in this thesis add weight to existing research by demonstrating that in addition to providing improved social outcomes (Wheeler et al., 2013a) there are also significant environmental gains to be had from water markets. The findings also support the rationale for the recent development and implementation of the CEWH environmental water-trading framework, which informs the purchase and sale of environmental water allocations (CEWO, 2016).

In addition to the significant quantitative hydro-economic modelling, substantial qualitative research was also undertaken, which has resulted in a number of insights regarding the type and role of EWHs in environmental water markets. To date qualitative research examining environmental reallocation strategies has typically focused on the views and motivations of irrigation farmers (e.g. Thampapillai, 2009; Loch et al., 2012). Instead, this thesis undertook research from the perspective of the purchaser, rather than the seller, of environmental water. The western US was chosen as the main focus, given the vast number of NGOs in this space and the potential to generate key insight for Australian environmental water management. Consequently, the thesis research contributed significant insights into answering the non-trivial question of whom, or which institutions, are entitled to buy and manage water for the environment. To the author's knowledge, despite indications of the diversification of EWHs in Australia and the western US, there has been limited research conducted investigating the specific role of non-governmental organisations in the market for environmental water (a notable exception is King, 2004). This thesis contributes to the literature by exploring specific EWH roles through the lens of subsidiary and stewardship theory, to identify the advantages and challenges of this institutional arrangement. A further contribution is derived from introducing the concept of the non-government environmental steward, which reflects the observed EWH departure from the traditional land and water trust model (King, 2004) to better reflect the diversity of non-government actors owning and managing environmental water for the public benefit. Considerable insights were gained regarding the importance of social capital in facilitating successful environmental water transactions, namely – trust and reciprocity, localism, information dissemination, as well as flexibility and multi-functionality in water products for the environment. This result contributes to the existing literature, which has identified social factors as limitations to reallocation (Gross, 2011; Evans and Pratchett, 2013) however to date, existing literature has not explored how social capital can be developed to facilitate successful reallocation. When examining and comparing the type of water products available to the environment, Wheeler et al. (2013a) depicted a relatively small range of the options currently in use in the western US. This thesis depicts a wider range of alternative transfer tools that are worthy of further exploration than has been previously recognised in the academic literature.

Given the findings from the qualitative interviews undertaken in the western US, this thesis contributes to the ongoing comparative assessment literature between Australia and the US, designed to elicit novel avenues for policy development and learning opportunities between the two contexts (e.g. Garrick et al., 2009; Garrick et al., 2011; Garrick et al., 2012; Grafton et al., 2012; Lane-Miller et al., 2013; Wheeler et al., 2013a). It methodologically contributes to transferring policy questions between case studies to uncover previously under-examined ideas, and highlights additional insights that may be explored and applied. In particular, issues surrounding the diversification of environmental water products and EWHs are discussed at length. Existing literature has reported irrigator intentions in the MDB have largely focused on irrigators' intentions regarding water sales or recovery preferences (e.g.

Wheeler and Cheesman, 2013; Loch et al., 2014b) and has not addressed the question of the type of EWH to whom they would prefer to sell water. The results indicate that irrigators exhibit a clear localism bias and list the federal government as their least preferred buyer of environmental water. This result contributes to the literature concerned with understanding irrigator motivations and behaviour to better inform reallocation policy. Further, the survey of irrigators' willingness to sell allocations to the environment enhances understanding of irrigator water sales preferences and adds weight to the existing literature, demonstrating irrigator preference for short-term, low-risk and flexible trade options.

9.3 Policy Recommendations

Based on the results discussed within this thesis, a set of six key MDB water policy recommendations have been developed. The findings of this thesis largely support recent water policy initiatives, such as the CEWH trading of environmental water allocations (CEWO, 2016), and the strategic local partnerships between the CEWH and local water stakeholder entities in South Australia, including the Ngarrindjeri Regional Authority (CEWH and NRA, 2015). Key water management policy recommendations derived from this thesis are discussed below.

Recommendation 1

- A greater focus on demand-side water management, including the judicious use of the water market to reallocate water to the environment, should be implemented rather than a continued reliance on supply-side management.

Analysis of the CLLMM case study and the wider MDB indicates that supply-based methods of scarcity management, such as environmental works and measures and reallocation through irrigation efficiency improvements, are being implemented as solutions to environmental challenges. However, the implications of continued infrastructure reliance, such as further fragmentation of the ecosystem and consequences for return flows; suggest that the costs may outweigh the benefits, particularly during drought. Alternative demand-side management is a viable alternative to managing scarcity and reallocation.

This recommendation is derived from the CLLLMM case study in Chapter 4 and supported by the MDB literature reviews in Chapters 2 and 3. The CLLMM case study analysis demonstrates that supply-side methods of managing scarcity can be identified as both the cause (e.g. barrages) and solution (e.g. environmental works and measure) to environmental degradation in the Lower Murray region. However, infrastructure solutions are not necessarily synonymous with achieving sustainability, and the cost of intervention in the CLLMM is calculated and shown to have occurred at significant public expenditure and high opportunity cost. More widely in the MDB, investment in infrastructure as a means of acquiring environmental water has also been shown to be expensive (PC, 2010; Loch et al., 2014b; Grafton, 2016) for comparatively lower environmental outcomes. Infrastructure solutions to scarcity have been shown to create perverse policy outcomes for environmental water availability as a

result of over-investment in irrigation infrastructure, higher consumptive use, and reduced reliability for downstream users including the environment (PC, 2010; Ward and Pulido-Velazquez, 2008; Adamson and Loch, 2014, Loch and Adamson, 2015). Based on the extensive case study and literature review of the ecological and economic costs of supply-side development in the CLLMM and MDB, it is suggested that demand-side focused policies, including the use of the water market as discussed in the following recommendations, be implemented as the primary method of acquiring water for the environment.

Recommendation 2

- Environmental water allocation trade should be more widely employed by federal and locally based EWHs, as well as further diversification and use of environmental water products incorporating principles of flexibility and multi-functionality to meet dual farming and environmental objectives.

Model results indicating positive hydrological, ecological and carbon storage benefits of annually reallocating water to the environment supports existing policy initiatives allowing for the purchase and sale of environmental allocations. Further environmental and welfare gains may be achieved by employing an annually adaptive reallocation approach through use of the water allocation market to increase environmental water availability.

This recommendation is derived from Chapters 7 and 8. Results from Chapter 7 demonstrate that purchasing annual reallocations for the environment can have positive hydrological and ecological impacts, and potentially alter carbon storage dynamics in native vegetation. This was demonstrated for the Lower Murrumbidgee floodplain forest, which experienced a modelled increase in overbank flooding volume and frequency, and a subsequent improvement in ecosystem health and carbon storage volumes. This result supports the recent expansion of policy to allow the CEWH to annually trade environmental water allocations following the CEWH Environmental Water Trading Framework (CEWO, 2016). This modelling result leads to a questioning of the current infrastructure-based reallocation strategy, which since 2015 has focused exclusively on environmental water acquisition through infrastructure upgrades (MDBA, 2016b).

The ability to improve ecological conditions through annual water trade also demonstrates the applicability and usefulness of adaptive and flexible reallocation arrangements, which can be achieved through EWH participation in the water market. The applicability of an annual reallocation approach is supported by evidence from the case study of the western US presented in Chapter 8. Results from Chapter 8 demonstrate that increased irrigator willingness to participate in reallocation strategies, if short-term, low-risk and multi-functional water transfer arrangements are offered by EWHs. This result is consistent with existing MDB literature, which shows that purchasing annual allocations for the environment may provide additional flexibility to EWH arrangements, increase irrigator willingness to participate in reallocation programs, ease structural adjustment and increase irrigator welfare (NWC,

2004; Kirby et al., 2006; Qureshi et al., 2007; Wheeler et al., 2013a). The improved social and environmental outcomes of annual environmental reallocation therefore suggests that the existing entitlement and infrastructure approaches to acquisitions could benefit from the addition of alternative strategies, such as annual reallocation which can be both ecologically effective and cost-efficient.

Recommendation 3

- Renewed efforts should be given to the management of environmental flows to enhance and sustain native floodplain forests for carbon storage and sequestration as a result of their proven ecological and ecosystem service values.

To implement this recommendation, it is suggested that conservation management of existing native forests for carbon benefits should be adopted, as well as native forest inclusion in carbon accounting schemes, enabling active management initiatives (such as improved overbank flooding, see Chapter 7) to generate and sell carbon credits as a novel source of income for natural resource managers.

This recommendation is based on Chapters 6 and 7, which demonstrate that floodplains vegetated with native Eucalypt species store and sequester large volumes of carbon, which may be improved through strategic overbank flooding targeted to promote vegetation growth and maintenance. However, the role of native forestry carbon dynamics in climate change mitigation is controversial and contrasting academic conclusions have been drawn (Keith et al., 2015). As a result native forests are typically omitted from national carbon accounting schemes, due to difficulties in quantifying the carbon stored in highly heterogeneous existing native forests (Jonson and Freudenberger, 2011) and the assumption of limited additionality (Law, 2013). Carbon sequestration policy in Australia has instead focused on the plantations of non-native species planted for timber production (Jonson and Freudenberger, 2011), which also has had negative implications for Australian biodiversity (Conte and Kotchen, 2010). More widely, terrestrial carbon sink management for climate change mitigation has focused on sustainable harvesting practice (IPCC, 2014) rather than conservation management of existing forests. Based on analysis within this thesis, a comparison cannot be drawn between conservation and harvesting practices, although literature exists to suggest that conservation of Eucalypt species compares favourably to sustainable harvesting methods in managing carbon stocks for climate change mitigation (Keith et al., 2015). The substantial capacity for Eucalypt vegetated floodplains to sequester large volumes of carbon makes a persuasive case for the expansion of policy to include native forests in carbon accounting schemes. This would allow for the active management of existing native forests to accrue and sell carbon credits, which is likely to prove economically beneficial and ecologically valuable. Policy expansion to include native forests in national carbon credit schemes may also provide natural resource managers (i.e. native forestry owners) a novel source of income, as well as an additional economic incentive for the conservation of native forests. Further, the management of native forests for improved carbon dynamics is likely to generate an additional set of co-benefits, specifically

around biodiversity and habitat which is otherwise unrealised with non-native carbon farming initiatives (Conte and Kotchen, 2010). Improved management of native floodplain forests for carbon storage would likely constitute as a no-regrets action.

Recommendation 4

- Consideration should be given to opportunities for EWHs to offset annual environmental water purchases through the strategic trade of water allocations and carbon credits, or additional marketable ecosystem services.

Further, opportunities for expansion of the relevant carbon accounting policy and the CEWH environmental water trading framework, or other frameworks governing the actions of private EWHs, should be investigated to explore the potential for offsetting or reducing the cost of purchasing annual environmental allocations, through the generation and sale of carbon credits or other marketable ecosystem services.

This recommendation is based on the results presented in Chapter 7, indicating that if the value of carbon stored in vegetation native floodplains is recognised and tradable, there may be opportunities for EWHs to either reduce or offset the cost of environmental water purchases, through selling the carbon credits generated by improved overbank flooding regime. This result suggests that the CEWH EWT framework (CEWO, 2016) could benefit from further expansion, allowing the CEWH to use the generation of carbon credits as a means to finance additional environmental water purchases or other restoration efforts. The proposed carbon-water trading strategy may also have implications for non-government EWHs, due to the ability for the proposed carbon-water trading strategy to lower financial costs of buying water for the environment; which can be prohibitive for smaller EWHs reliant on external or philanthropic funding arrangements. In addition to potential benefits for EWHs, there are also policy implications for cities or concerned entities (e.g. companies) interested in offsetting their carbon emissions through the purchase of carbon credits. For example, the possible trade of water allocations for carbon credits may open up new avenues for cities and companies to buy carbon indirectly through the purchase of water allocations, which may be less expensive under certain hydrologic and economic conditions. However, this may be more institutionally complex and would require judicious policy design regarding the ownership of the carbon credits generated, as discussed in Chapter 7. Such an arrangement would, in essence, create a new type of EWH previously non-existent in the MDB, which is briefly discussed in Chapter 8. However, at present there is no policy framework in place governing the potential generation and sale of carbon credits stored in native floodplain forests, and significant institutional development would need to occur to effectively operate the proposed carbon-water trading strategy.

Recommendation 5

- Water reallocation policy should be informed by modelling that gives adequate consideration to the complexities and uncertainties inherent in water resource systems in order to better understand the risk of implementing new policy arrangements.

For hydro-economic modelling of the MDB to fully mature and inform robust water policy, it is recommended that greater attention to the representation of complex ecosystem service dynamics and the treatment of inherent environmental uncertainty be adopted as best practice in modelling and policy development.

This recommendation is derived from Chapters 6 and 7. Accounting for complex ecosystem dynamics allows the identification of reallocation opportunities and ecosystem service benefits, which would otherwise not be evident. To date, hydro-economic modelling in the MDB has focused disproportionately on the costs of reallocation to the irrigation sector, without comparative attention to the ecological and economic benefits derived from improved environmental water availability. Policy that is informed by hydro-economic modelling, undertaken without a clear understanding of the complex dynamics and significant economic value generated by flow-dependent ecosystem services, can run the risk of informing sub-optimal policy which understates the benefits of reallocation. Similarly, modelling ecosystem service benefit in a non-dynamic framework may overlook the value of annually adaptive reallocation approaches, such as the ecological and economic benefits of annually reallocating water to the environment as demonstrated in this thesis. Furthermore, MDB policy and hydro-economic modelling has generally not paid due attention to the full suite of uncertainties and sensitivities that are faced in water resource management (Settre et al., 2017) and this has translated into policy construction. For example, river flows and diversions are more sensitive to possible climate futures than they are to reallocation scenarios considered in the development of the *Basin Plan* (e.g. 2,400; 2,750; 3,200GL) (Kirby et al., 2014a), yet a 2,750GL water recovery target was implemented. For this thesis, the sensitivity analysis enabled the applicability and limitations of the proposed trading strategy to be identified, as well as the sources of uncertainty acting within the model (Chapter 7). This highlighted that under dry future inflow scenarios, opportunities to fully offset the cost of water with the sale of carbon credits are limited due to the higher cost of allocation water. Policy design based on the results of hydro-economic models that do not account for the inherent climactic, ecological and economic uncertainty affecting socio-economic systems will inevitably reduce welfare (Adamson, 2015; Settre et al., 2017). Conversely, models that undertake a thorough sensitivity analysis may assist in understanding the risk in policy making and the impact uncertainty will have in achieving optimal social solutions.

Recommendation 6

- Federal and state level EWHs should consider strategically partnering with local and non-government organisations to increase irrigator participation in environmental water reallocation programs and sustain local values in resource decision-making.

It is suggested there may be benefits for the federal government to continue exploring opportunities to partner with existing local and non-government institutions, in order to better include principles of localism in decision-making, overcome bias against federal engagement in the water market, as well as potentially increasing irrigator engagement and better responding to local river conditions.

This recommendation is derived from Chapter 8, which demonstrates that irrigator preferences in the MDB are aligned with principles of localism when considering who they would rather sell water to, placing federal and state governments last on their preference list, behind local irrigators, local councils and environmental organisations. This result may be partly explained by irrigators' preference for local control of resources or the perceived injustices perpetuated by the federal government against rural interests during the entitlement recovery[81] (Gross, 2011). While the *Basin Plan* currently has provisions for including local communities in decision-making[82] (MDBA, 2012a), implementation of this concept has been tense, often mixing emphasis on local knowledge and decision-making with the need for centralised river management (Parliament of Australia, 2013) and the principles of localism are not fully accounted for in current policy. However, the concept of localism is complex, as the most vocal and vested interests such as agricultural lobby groups often capture the 'localism voice'. Incorporation of localism through strategic partnerships must therefore proceed based on a factual understanding of rural community preferences, such as that achieved through representative irrigator surveys, to avoid facilitating rent-seeking behaviour from vocal minorities with vested interests in resisting or manipulating environmental reallocation policy.

Irrigator preference for the sale of water to local institutions suggests a potential for increased irrigator willingness to participate in reallocation programs if local values were sustained through EWHs acting at the local level. Engagement of local entities also provides an avenue for environmental watering decisions to more accurately respond to local environmental conditions in real time as well as providing avenues for EWHs to disseminate information and intention to buy water through existing information channels – which is particularly important in rural communities where family and local networks remain the primary source of trusted information (Sligo and Massey, 2007). While subsidiary style governance in the MDB is an unlikely reality, there may be scope for centralised EWHs to develop

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[81] Note that since 2015 federal water recovery expenditure has focused exclusively on irrigation infrastructure upgrades (MDBA, 2016b). Loch et al. (2014) show that there is split demand for infrastructure and market-based reallocation among irrigators. The weight of arguments citing poor inclusion of rural interests in the reallocation process is therefore somewhat reduced since this policy shift.

[82] Environmental watering should be undertaken having regard to the views of: a) local communities ….; b) persons affected by the management of environmental water (MDB Plan Division 6, Subdivision A, Principle 7)

strategic partnerships with local government institutions (e.g. local council) to improve irrigator perception of reallocation and increase irrigator engagement. Such a form of governance reflects the successful polycentric approach to environmental water management in the western US, which includes a middle path between top-down (e.g. centralised power) and bottom-up (e.g. subsidiary power) governance arrangements (Garrick et al., 2011). Within the MDB, the recent development of an agreement between the CEWH and the Ngarrindjeri Regional Authority is an example of a collaborative strategic local partnership which provides for the transfer, delivery and monitoring of Commonwealth environmental water in the Lower Murray by the NRA (CEWH and NRA, 2015). In addition to preferences for the sale of water to local institutions, the survey results reveal that irrigators marginally prefer to sell water to environmental organisations than high-level government. This result potentially demonstrates an opening in the water market for non-government organisations to actively engage with irrigators and buy water for the environment. Recent examples of this are provided by the MDB Balanced Water Fund operated by Kilter Rural, which aims to invest up to $27 million in water entitlements to be partly managed for environmental outcomes (Kilter Rural, 2016a); as well as the modest amounts of water owned by Nature Foundation SA in the Lower Murray (Nature Foundation SA, 2015). Insights from the US case study indicate that non-government EWH organisations are often able to provide flexible transaction arrangements, which are preferred by irrigators (Chapter 8), although this is yet to be examined in the case of the MDB. Nevertheless, the reported irrigator willingness to sell water to environmental organisations may indicate a potential growth of NGO engagement in the MDB water market in the future.

9.4 Limitations and Future Work

9.4.1 Limitations

As described within this thesis, the methodologies employed suffer from some weaknesses or limitations. This section highlights the challenges faced, how they were dealt with and the potential impact on the results.

9.4.1.1 Hydro-economic Modelling

Ecological behaviour is rarely fully observed and as a result time-series data is short, case studies are localised and findings may have limited generalizability (Webb et al., 2015). The relative lack of data has implications for how the hydro-economic model is built and calibrated. This was addressed in a number of ways. First, available data describing River Red Gum response to flood and water deficit was sourced from research covering a range of studies comparable to the Lowbidgee in order to increase data availability, and behaviour was corroborated using theoretical models of forestry dynamics (e.g. the von Bertalanffy Chapman-Richards growth model). Remaining unknown ecological parameters, such as time until tree death, were obtained from expert opinion, and sensitivity tested in the model. In rare cases of the complete absence of knowledge, such as the maximum carbon stored per hectare,

highly conservative estimates were made following expert advice. For example, as explained in Chapters 6 and 7, the observed mean carbon storage per hectare was implemented as the upper-bound storage capacity. The relatively small amount of data used to calibrate the forest dynamic impacts the results by potentially limiting their generalisability.

Because conservative estimates were always used when estimates were required, the implication of this is the likelihood that benefits from the reallocation results are underestimates of the total possible benefits. This is also the case given only one tree species is modelled and additional sources of carbon (which store significantly less, but not negligible volumes) are not included in the model, as described in Chapter 6. Further, native forest carbon sequestration dynamics are often poorly understood, although this is a continually evolving area of science. To overcome this deficiency, it was assumed that sequestration rates are proportional to tree health – a fact well supported in the literature. However, such an assumption omits the possibility of time lags between changes in tree health and changes in carbon storage capacity. The implications of this limitation are that the magnitude of change in carbon stocks may be overestimated in any one simulation period. However, for the data at hand and the current state of knowledge there is no known method of rectifying this assumption.

In hydrological models it is common practice to undertake a process of verification and validation to ensure the model is behaving properly. This is typically achieved by splitting time-series data into calibration (i.e. training the model) and validation (i.e. evaluating the model). However, due to the complexity of highly integrated modelling, the large degrees of freedom, varying time and spatial scales, and once-off policy changes which cannot be replicated, traditional validation approaches are a considerable challenge and potentially not applicable (Letcher et al., 2003; Jakeman and Letcher, 2003). These challenges have also been faced by other similarly integrated models of environmental reallocation (e.g. Mainuddin et al., 2007). Instead, it is recommended that each component be tested individually (Jakeman and Letcher et al., 2003), as has been implemented where possible in this thesis. Because the model is run using a historical time-series of flows, it was possible to qualitatively validate each model component against reporting and monitoring literature from that time. For example, in 2003 it was reported that up to 95% of River Red Gums in MDB floodplains were water stressed (MDBC, 2003), which is behaviour the model simulated. This indicates that the model was able to capture the broad dynamics of floodplain forests. In this way, each component in the model was validated to ensure it is behaving in an expected manner.

9.4.1.2 *Qualitative Analysis*

There are a number of stakeholders who play a role in environmental water governance in the western states of the US. In line with the research objectives, non-government environmental groups accounted for the vast majority of the identified and sampled participants. Interview participants from stakeholder groups including farming communities, agricultural lobby groups and government agencies at the state and federal level were also contacted, but declined to participate in the study. The absence of a balanced split of study participants from each stakeholder group could be used to argue that the results may suffer from sampling bias. In any qualitative research project, sampling bias is ever present as it is either impossible or financially prohibitive to undertake random sampling or obtain representative samples from all stakeholder groups (Oppong, 2013). Further bias may arise from the fact that interview participants inject their values, opinions and bias into the results through their interview responses, as well as values projected onto the respondents, data and interpretation from the perspective of the research (Norris, 1997; Anderson, 2010; Chenail, 2011). In the case at hand, purposive sampling was employed and pre-interview screening was used to omit participants who do not fit the sample selection. This was done because the purpose of this part of the research was not to find a representative sample of any particular population; rather it was to identify those who had specific knowledge of the subject under investigation (Patton 2002). Based on the results, it is evident that there is an overwhelming participant bias towards improving environmental conditions, perhaps even at considerable costs to the irrigation industry. There is also an evident bias towards the use of market mechanisms (as opposed to administrative or legislative methods) to achieve improved environmental outcomes. Systematic removal of the value bias introduced by the values of the participant is hardly possible (Norris, 1997), but needs to be acknowledged.

The impact of potential sources of limitations has been mitigated, as much as possible, by efforts to establish the validity (e.g. credibility and believability) of the findings. Due to the nature of the semi-structured interviews it was possible to identify the point of data saturation, which occurred when no new information was presented. Saturation occurred at 49 interviews, after which no further interviews were conducted. Reaching data saturation has implications for research quality and validity (Fusch and Ness, 2015). As saturation was reached in this case, this indicates that the data is valid, and therefore reliable, such that confidence can be had that the results are valid and replicable if the research was to be repeated with the same research design, irrespective of potential bias inherent in the results. Nevertheless, when interpreting the results it must be noted that these are representative of the non-government stakeholder group only.

9.4.2 Future Work

Several avenues for future work have been identified from this thesis. Specific areas where future work is especially warranted include: a) the consideration of additional ecosystem services in the HEM; b) greater consideration of the stochasticity of inflows and ecological parameters; and c) further exploration of irrigator attitudes to different types of EWHs. These are discussed in detail below.

Methodologically, the quantitative modelling thesis has demonstrated the value and insight that can be obtained through integrating diverse data sets and research methodologies to construct a dynamic hydro-economic model of water resource allocation. The results of this approach suggests that further work would benefit from integrating ecology and ecosystem services representation into a hydro-economic modelling framework, to achieve a more holistic understanding of water resource systems and the various interactions therein. To progress this line of work in the future, this thesis highlights the need for improved monitoring data of ecosystems in the MDB with particular focus on ecological thresholds, responses to hydrological events, and the length of time-series data. This would facilitate improved representation of ecosystem dynamics in hydro-economic modelling and allow a greater level of socio-ecological integration to be achieved in the modelling. The importance of uncertainty analysis has also been highlighted and it is concluded that further work to better address the full suite of uncertainty faced in hydro-economic modelling would help to improve confidence and applicability of hydro-economic modelling results.

From the qualitative research it has been shown that there are considerable insights gained through the study of EWHs with regard to how they operate and engage with farmers. This suggests that additional research in this area could, when paired with considerable literature regarding farmer preferences and attitudes, provide a more complete picture of the environmental water market landscape and actors. The evidence suggesting that irrigators list the federal and state as their least preferred buyers of permanent water indicates that considerably more research is required to unpack irrigator perception of government actions and the potential implications for water recovery. Conversely, this thesis demonstrates the clear principles of localism in irrigator preferences for water sales, which has been unrealised in policy and under-examined in academic literature. A greater understanding of how principles of localism can be effectively integrated into state and national level water policy planning is required in the future. Further work in this area would likely provide great benefits in informing the appropriate mix of top-down and bottom-up style governance in environmental water policy, and potentially have implications for irrigator engagement in reallocation programs.

Further, the integration of both qualitative and quantitative research approaches throughout this thesis demonstrates the value in undertaking a mixed-methods approach to best understand the posed research questions from a range of perspectives. Future research in water resources may benefit from integration of various research traditions in a mixed-method approach, especially considering the multi-faceted

and value-plural nature of water resources, along with the highly complex and wicked nature of future water resource challenges.

From a perspective of the results, this thesis has demonstrated the ability for an EWH to use the water allocation market in achieving positive environmental outcomes. This result has been proven quantitatively in only a small number of other studies (e.g. Connor et al., 2013; Ancev, 2014). Future research would benefit from further exploring both this potential and the use of methods for self-financing EWHs to offset water purchase costs through trade in other markets. This may be addressed through the investigation of additional marketable ecosystem services, as described in the following section. There is also scope for future work to quantitatively model the hydrological and ecological impact of additional water products (beyond allocations and entitlements), which has been qualitatively discussed but not quantitatively investigated in a hydro-economic framework to date. Quantifying the effect of additional water products for the environment may help to determine the appropriate mix of water products to achieve optimal environmental outcome and an effective use of public expenditure. The use of additional water products for the environment is bolstered by the analysis of transfer tools used in the western US, which are wider in scope and complexity than previously reported. Additional research considering the newly identified transfer tools (e.g. water products) for the environment is warranted in Australia, especially considering the success with which water allocation trading for the environment (in addition to entitlement purchases) has been implemented by the CEWH.

In addition to the points described above, future work would best benefit by addressing the following three main areas.

9.4.2.1 Consideration of Additional Ecosystem Services

The HEM presented in this thesis includes only the carbon storage and sequestration ecosystem service. However, the Lowbidgee and the MDB more widely generate a vast number of ecosystem services (Bark et al., 2016), such as those listed in Chapter 6. Further modelling of marketable ecosystem services, such as biodiversity credits, would be of particular benefit in order to identify additional opportunities to finance environmental water purchases with the generation and sale of ecosystem services. Including more ecosystem services in the HEM would generate a more accurate picture of how ecosystem services interact or compete, which is a gap that needs to be addressed in ecosystem service modelling as a whole (Seppelt et al., 2011). Understanding the interactions between multiple ecosystem services, including provisioning services such as food production, also provides a more accurate picture of trade-offs faced by a water resource decision-maker.

It is identified that additional ecosystem services, such as native fish habitat, Blue Green Algae prevention, erosion prevention, and recreation ecosystem services could be integrated into the HEM presented in this thesis. Further elaboration and possible methods of incorporating these ecosystem services into the existing HEM framework are presented in Appendix C.

9.4.2.2 *Stochastic Dynamic Modelling*

The model presented within this thesis is a dynamic mathematical model, which converges on an optimal reallocation strategy of an EWH over an iteratively updated three-year decision horizon. Water resource management is an inherently uncertain practice and decisions must often be made with incomplete knowledge of future events and unknown parameter values. Stochastic dynamic programming is one method of modelling water resource decision-making under uncertainty (Yakowitz, 1982). An application of the stochastic dynamic modelling approach implemented in the MDB is provided by Rowan et al. (2011), who use the approach to estimate the impact of climate change and variability on irrigated agriculture.

The process of developing a stochastic dynamic model for water resource decision-making is non-trivial, and there are considerable challenges when applying this programing method to a highly integrated model with multiple and inter-temporal feedback loops. As such, this approach was not adopted in this thesis, although there is likely merit in accounting for stochastics in future iterations of the model. In the first instance, this could be achieved through the inclusion of probability distributions for uncertain model parameters; specifically highly uncertain ecological parameters that govern key ecosystem dynamics. An example of probabilistic representation of parameter uncertainty applied to a dynamic model of vegetation and carbon flux is presented in O'Hagan et al. (2012) and provides a useful guide of how probability distributions for uncertain parameters may be used in an integrated water resource modelling framework. Overall, future work would benefit from the inclusion of stochastic representation to improve model result robustness and better account for uncertainty in ecological and hydrological parameters.

9.4.2.3 *Irrigator Attitudes towards EWHs*

The irrigator survey responses present an interesting result regarding irrigator preferences for the sale of water to EWHs at various institutional levels. While aggregated results provide an overall indication of irrigator preferences, it is known that irrigators are non-homogenous in their water sales motivation and behaviour (Wheeler et al., 2012). Thus, further work is required to better understand the irrigator characteristics associated with preferences for water sales. In general, few studies have investigated irrigator water sales to the environment, and the majority of research has been carried out in the MDB to examine irrigator engagement with the *RtB* program (e.g. Wheeler et al., 2009; Thampapillai, 2009; Wheeler et al., 2010; Wheeler et al., 2012; Loch et al., 2012; Wheeler and Cheesman 2013; Haensch et al., 2016). Extending the work presented in this thesis would be best achieved using a more detailed econometric approach, to examine which land and water use, farming and socio-economic characteristics are most closely associated with irrigator trading preferences. Future work in this area is warranted.

9.5 Final Conclusion

Despite these limitations, this thesis has broken new ground and demonstrated the ability for water markets to be used as a mechanism to reallocate water to the environment for improved ecological outcomes. In particular, this thesis has shown that trading water allocations for the environment can have positive ecosystem service benefits by improving floodplain inundation. Under certain hydrological and fiscal conditions, the increase in floodplain carbon storage may be of sufficient market value to offset the cost of environmental water allocation purchases. This result suggests a novel carbon-water trading strategy which may provide an additional revenue stream for self-financing environmental water holder. In addition, this thesis has explored the role of non-government environmental water holders and demonstrated the importance of flexibility, multi-functionality and localism in reallocating water to the environment through the water market. Contributions to the literature have also been made through the finding that southern MDB irrigators have a clear preference for the local management of water resources and shorter-term (e.g. water allocation trade) environmental water transactions. Overall, the results of this thesis indicate that there is considerable potential for the expanded use of water markets to provide greater and more efficient environmental flows.

APPENDIX A: Discussion of Ecosystem Service Classifications

Frameworks commonly used to categorise and evaluate ES include the Millennium Ecosystem Assessment (2005); The Economics of Ecosystems and Biodiversity (2010); and the Common International Classification of Ecosystem Services (2011); UK National Ecosystem Assessment; Fisher and Turner (2008); Landers et al. (2013); SEEA (2012). All ES frameworks share the underlying philosophy of providing a means to make the value of the environment visible and quantifiable (Haines-Young et al., 2012). Methods of classifying ecosystem services is not the focus of this thesis and only the three most widely used frameworks (TEEB, MAE and the CICES) are discussed. These are shown in Figure

The Millennium Ecosystem Assessment (MEA) was launched by the UNEP in 2001 primarily to assess the consequences of changes in ecosystems on human well-being and the action needed to address these changes. The MEA framework has been applied with success in the UK where it was used to provide an assessment of the state and value of ecosystems at national level (Brown et al., 2011). TEEB is founded on the MEA concepts of ecosystem services and accelerated by the Stern Review of Climate Change (Stern, 2006; TEEB, 2013). The objective of the TEEB project is to recognise, demonstrate and capture the value of nature the value of nature which is often invisible in environmental decision-making (TEEB 2013). The TEEB approach is currently being implemented at TEEB Climate Initiative operating in Columbia, Kenya, Tanzania and Thailand to evaluate the benefits and costs of climate-friendly agricultural land management projects. The CICES was introduced as a means of standardizing how ecosystem services are described and valued in environmental accounting in order to be able to make meaningful comparisons between results and case (CICES 2016). CICES is the basis for the EU Mapping and Assessment of Ecosystems and their Services (MAES) project which aims to map ecosystem services at a continent wide scale to inform actions to meet the EU's Biodiversity Strategy (European Commission 2013).

Figure A.1 Millennium Ecosystem Assessment Framework

Provisioning Services	Regulating Services	Cultural Services
Products obtained from ecosystems	*Benefits obtained from regulation of ecosystem processes*	*Non-material benefits obtained from ecosystems*
• Food • Fresh water • Fuel wood • Fiber • Bio chemicals • Genetic resources	• Climate regulation • Disease regulation • Water regulation • Water purification • Pollution	• Spiritual and religious • Recreation Aesthetic • Inspirational • Educational • Sense of place • Cultural heritage

Supporting Services
Services necessary for the production of all other ecosystem services
• Soil formation
• Nutrient cycling
• Primary production

Source: MEA (2003), p. 57.

There is some overlap between the various methods of categorizing ecosystem services. For example, the Millennium Ecosystem Assessment (2005) has a 'supporting services' category which are necessary for the production of all other ecosystem services (MEA 2003). Supporting services have indirect impacts on people which occur over long periods of time and include soil formation, nutrient cycling, water cycling, atmospheric oxygen production and habitat provision (MEA 2003). In the TEEB framework, some supporting services are incorporated into alternative categories, for example water and nutrient cycling are classified as regulating services under 'waste-water treatment', which is an output of the water cycle, and 'maintenance of soil fertility' which is an output of nutrient cycling. Species habitat classified as supporting services in MAE are listed as their own category in TEEB which includes 'habitat services' and 'maintenance of genetic diversity' (TEEB 2010). Unlike TEEB and MEA, the CICES uses a five-level hierarchical structure to describe the 'final' (non-supporting) ecosystem services. The hierarchical system is used to as a means to identify which ES outputs are recognised as ecosystem services (CICES 2016).

Figure A.2 The TEEB Framework

Provisioning Services	**Regulating Services**	**Habitat Services**	**Cultural Services**
The material or energy outputs from ecosystems'	*The services that ecosystems provide by acting as regulators*	*The services that ecosystems provide by providing homes for species*	• Recreation, mental health and physical health • Tourism • Aesthetic appreciation • Spiritual experience and sense of place
• Food • Raw material • Fresh water • Medicinal resources	• Local climate • Carbon sequestration • Moderation of extreme events • Waste-water treatment • Erosion prevention • Pollination • Biological control	• Habitat for species • Maintenance of genetic diversity	

TEEB (2010). Available: http://www.teebweb.org/resources/ecosystem-services/, [Accessed August 4th, 2017]

The choice of framework is important in the valuation of ecosystem services and helps in addressing valuation challenges such as double-counting. The risk of double counting is especially a concern for the MAE framework which categorises supporting services as drivers of ES production, thus posing a risk of double counting benefits if both supporting services and ecosystem services were to be valued (Fisher and Turner 2008). Conversely, excluding supporting services from valuation to avoid double counting may provide an underestimate of the total ecosystem services generated by the environment. Further, while MEA provided the initial building blocks for considering ecosystem services, it is argued by some authors that it does not provide a comprehensive list (Haines-Young et al., 2012). As such the MEA is not employed in this thesis. Another option is the CICES which is a useful means of combining the MEA and TEEB approach to address missing services in the MEA. However, the hierarchical structure of CICES adds complexity without adding additional insight in the current study. At the time of writing, the CICES is not in its final state and is currently in V4.3.

Figure A.3 Common International Classification of Ecosystem Services framework

Section	Division	Group	Class	Class type
Provisioning	Nutrition	Biomass	Cultivated crops	Crops by amount, type
			Reared animals and their outputs	Animals, products by amount, type
			Wild plants, algae and their outputs	Plants, algae by amount, type
			Wild animals and their outputs	Animals by amount, type
			Plants and algae from in-situ aquaculture	Plants, algae by amount, type
			Animals from in-situaquaculture	Animals by amount, type
		Water	Surface water for drinking	By amount, type
			Ground water for drinking	
	Materials	Biomass	Fibres and other materials from plants, algae and animals for direct use or processing	Material by amount, type, use, media (land, soil, freshwater, marine)
			Materials from plants, algae and animals for agricultural use	
			Genetic materials from all biota	
		Water	Surface water for non-drinking purposes	By amount, type and use
			Ground water for non-drinking purposes	
	Energy	Biomass-based energy sources	Plant-based resources	By amount, type, source
		Mechanical energy	Animal-based resources	By amount, type, source
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Bio-remediation by micro-organisms, algae, plants, and animals	By amount, type, use, media (land, soil, freshwater, marine)
			Bio-chemical detoxification / decomposition / mineralisation in land / soil, freshwater and marine systems including sediments; decomposition / detoxification of waste and toxic materials e.g. waste water cleaning, degrading oil spills by marine bacteria, (phyto)degradation, (rhizo)degradation etc.	By amount, type, use, media (land, soil, freshwater, marine)
		Mediation by ecosystems	Filtration/	By amount, type, use, media (land, soil, freshwater, marine)
			sequestration/	
			storage/accumulation by ecosystems	
			Dilution by atmosphere, freshwater and marine ecosystems	
			Mediation of smell/noise/visual impacts	
	Mediation of flows	Mass flows	Mass stabilisation and control of erosion rates	By reduction in risk, area protected
			Buffering and attenuation of mass flows	
		Liquid flows	Hydrological cycle and water flow maintenance	By depth/volumes
			Flood protection	By reduction in risk, area protected
		Gaseous / air flows	Storm protection	By reduction in risk, area protected
			Ventilation and transpiration	By change in temperature/humidity
	Maintenance of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool	Pollination and seed dispersal	By amount and source
			Maintaining nursery populations and habitats	By amount and source
		Pest and disease control	Pest control	By reduction in incidence, risk, area protected
			Disease control	
		Soil formation and composition	Weathering processes	By amount/concentration and source
			Decomposition and fixing processes	
		Water conditions	Chemical condition of freshwaters	By amount/concentration and source
			Chemical condition of salt waters	
		Atmospheric composition and climate regulation	Global climate regulation by reduction of greenhouse gas concentrations	By amount, concentration or climatic parameter
			Micro and regional climate regulation	
Cultural	Physical and intellectual interactions with biota, ecosystems, and land- /seascapes [environmental settings]	Physical and experiential interactions	Experiential use of plants, animals and land-/seascapes in different environmental settings	By visits/use data, plants, animals, ecosystem type
			Physical use of land-/seascapes in different environmental settings	
		Intellectual and representational interactions	Scientific	By use/citation, plants, animals, ecosystem type
			Educational	
			Heritage, cultural	
			Entertainment	
			Aesthetic	
	Spiritual, symbolic and other interactions with biota, ecosystems, and land- /seascapes [environmental settings]	Spiritual and/or emblematic	Symbolic	By use, plants, animals, ecosystem type
			Sacred and/or religious	
		Other cultural outputs	Existence	By plants, animals, feature/ecosystem type or
			Bequest	

APPENDIX B: Carbon Storage Model Supplementary Data and Outputs

B.1 Supplementary Input from Chapter 6 and 7

Above ground River Red Gums (*Eucalyptus camaldulensis)* store more carbon than other carbon components in native vegetated floodplain ecosystems (Smith and Reid, 2013). Table B.1 shows the volume of carbon stored in each ecosystem component for River Red Gums in the Namoi catchment. The Namoi is an MDB sub-catchment in NSW geo-hydrologically and vegetatively close to the Murrumbidgee catchment (CSIRO, 2008b).

Table B.1: Mean Carbon Stored in River Red Gums (tCha-1 ± s.e.m) in the Lower Namoi Floodplain in Northern New South Wales

Carbon Component	Carbon stored in River Red Gum ecosystem component (n=13) (tCha^{-1} ± s.e.m)
Woody	104.4 ± 20.0
Herbaceous	0.7 ± 0.1
Litter	1.6 ± 0.2
Coarse woody debris	6.2 ± 2.1
Dead standing	4.7 ± 1.5
Roots	26.5 ± 0.5
Total organic carbon (0-30cm)	71.7 ± 4.4
Total	215.9 ± 28.1

Source: Smith and Reid (2013).

During times of drought or anthropogenic changes to the hydrological regime (e.g. over-abstraction) impacted areas of River Red Gum forests experience symptoms of water stress and decline in tree productivity. River Red Gums reduce transpiration through stress-induced defoliation to conserve soil water and reduce heat load (Roberts and Marston, 2011; Doody et al., 2015). Table B.2 shows average annual transpiration rate and plant available water for River Red Gums in the Lowbidgee. An explanatory relationship between average annual transpiration rates and tree health is assumed.

Table B.2 Annual River Red Gum Transpiration (mm) between May 2010 and April 2012 with 95% Confidence Intervals for Selected Flooding Sites on the Lower Murrumbidgee Floodplain in Western New South Wales

Site	Annual Transpiration (mm)	Average Annual Transpiration per Site Grouping (mm)
1-2 years	566 ± 394	398.25
2-2 years	392 ± 117	
3-2 years	288 ± 150	
4- 2years	347 ± 118	
1-5 years	84 ± 49	167.25
2-5 years	208 ± 127	
3-5 years	285 ± 160	
4-5 years	92 ± 60	
1-10 years	17 ± 3	17

Source: Doody et al. (2015).

B.2   Supplementary Output from Chapter 7

B.2.1   Baseline Hydrological Conditions

Figure B.1 Baseline Modelled Consumptive Water Availability in the Lower Murrumbidgee Catchment
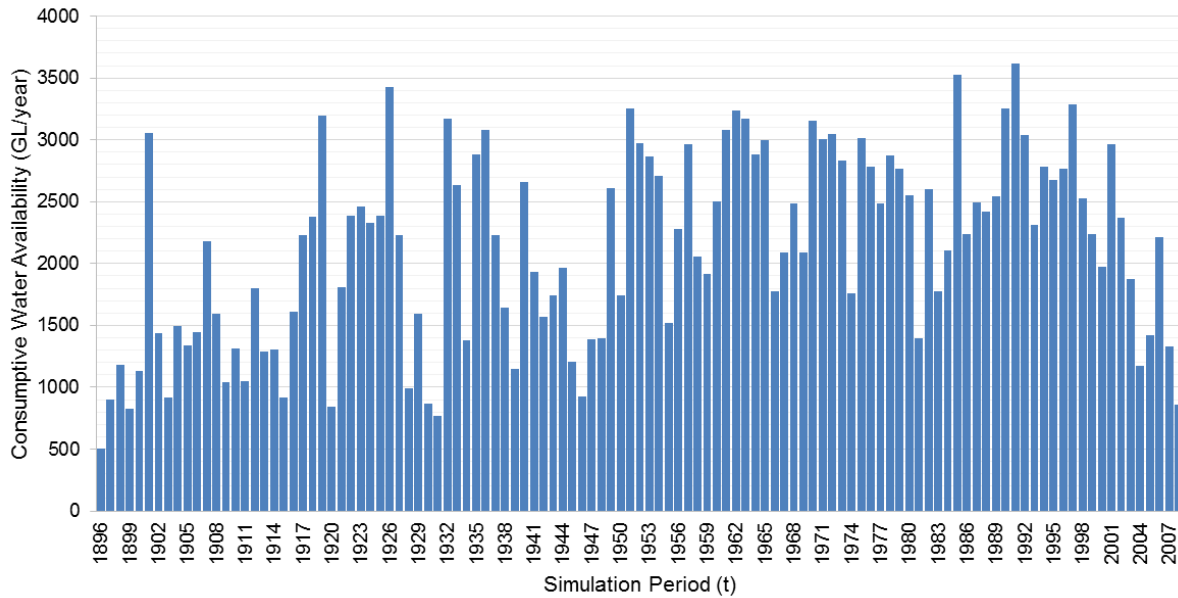
Source: Own Figure.

Table B.3 Descriptive Statistics of Baseline Modelled Consumptive Water Availability

Descriptive Statistics	Consumptive Water Availability (GL/year)
Average	2,127
Minimum	502
Maximum	3620
Standard Deviation	771
Total summed over simulation	24,0457

Source: Own Table.

Figure B.2 Baseline Modelled Environmental Water Availability in the Lower Murrumbidgee Catchment
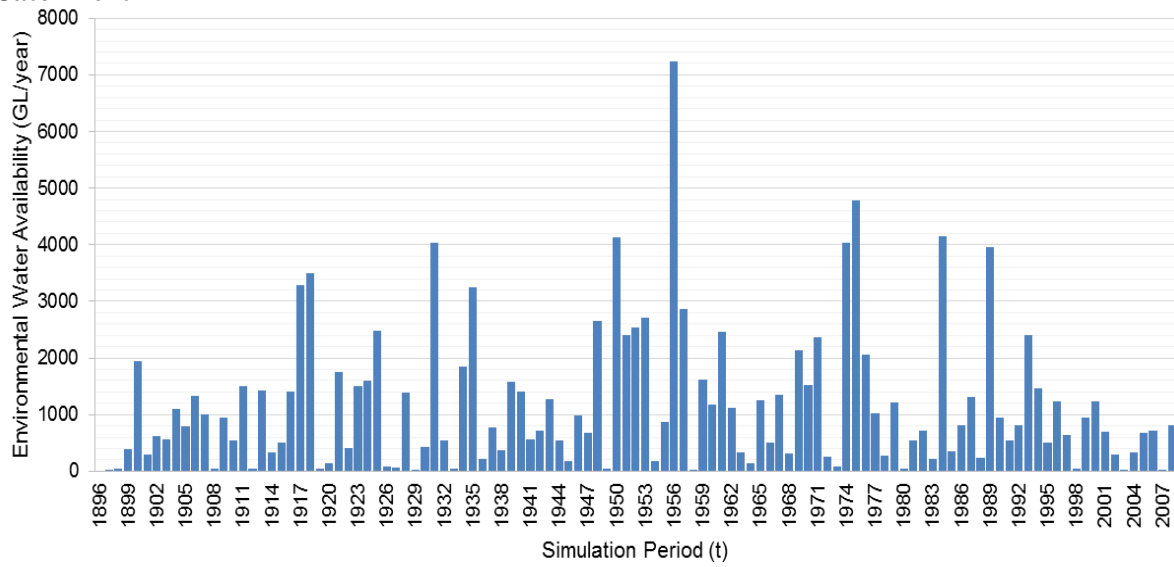
Table B.4 Descriptive Statistics of Baseline Modelled Environmental Water Availability

Descriptive Statistics	Environmental Water Availability (GL/year)
Average	1175
Minimum	0
Maximum	7230
Standard Deviation	1233
Total summed over simulation	132845

B.2.2 Baseline Modelled Water Allocation Price

Figure B.3 Baseline Modelled Water Allocation Price in the Lower Murrumbidgee Catchment
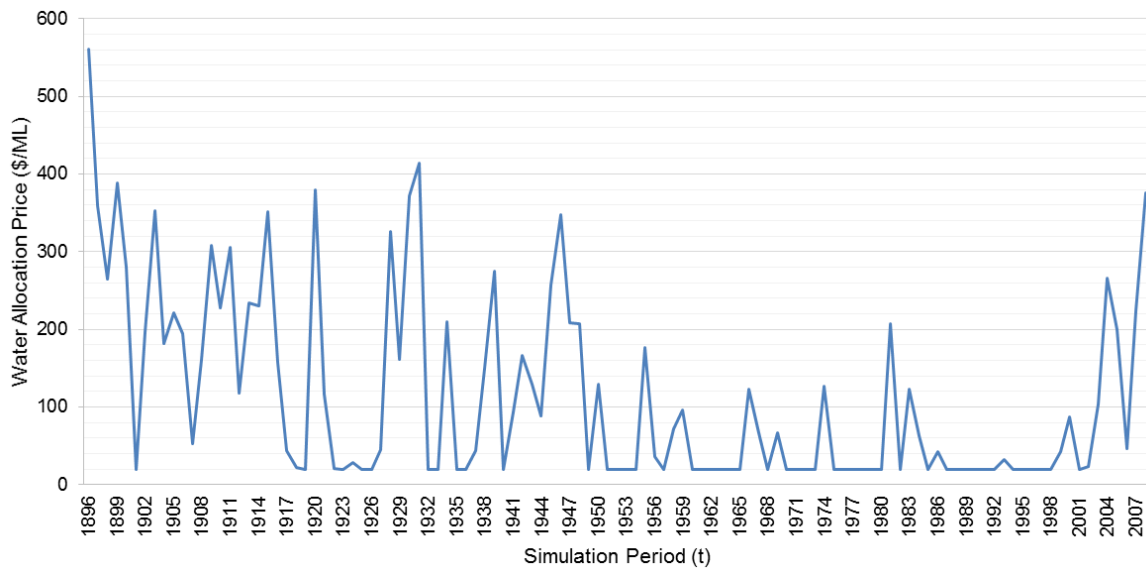
Table B.5 Descriptive Statistics of Baseline Modelled Water Allocation Prices

Descriptive Statistics	Water Allocation Price ($/ML)
Average	114.3
Minimum	20
Maximum	560.8
Standard Deviation	122.00

B.2.3 Baseline Modelled Carbon Storage and Carbon Value

Figure B.4 Baseline Modelled Floodplain River Red Gum Carbon Storage in the Lower Murrumbidgee Floodplain
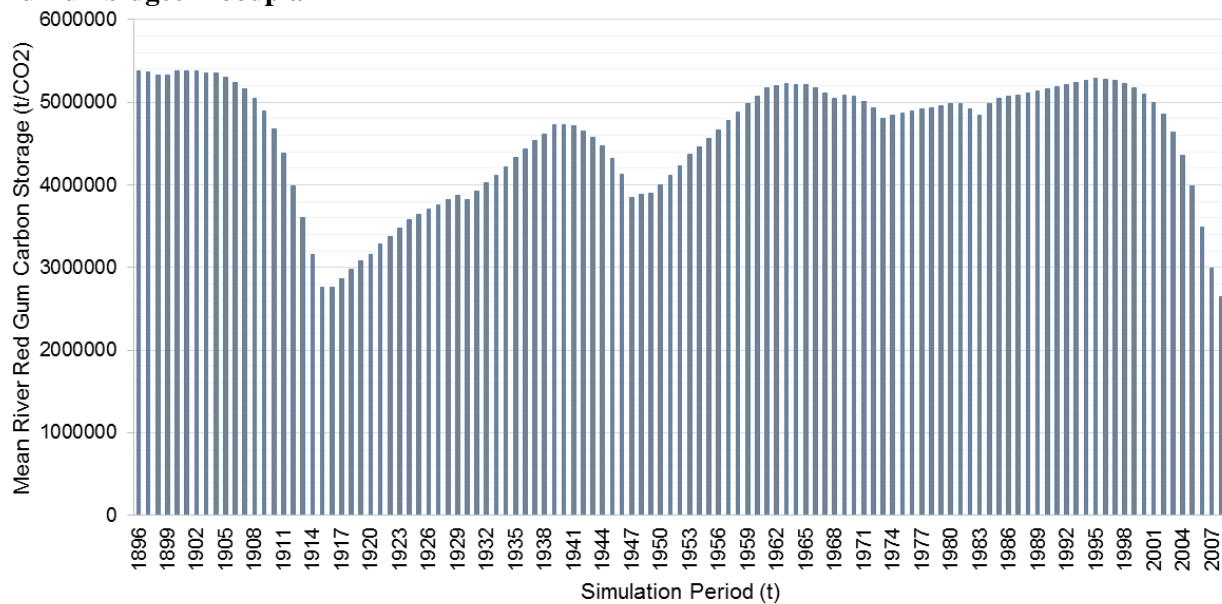
Table B.6 Descriptive Statistics for Modelled Floodplain River Red Gum Carbon Storage

Descriptive Statistics	Carbon Stocks (tC)
Average carbon storage	4558303
Minimum carbon storage [year occurred]	2641280 [2008]
Maximum carbon storage [year occurred]	5380324 [1896]
Standard deviation	724371
Total summed over simulation	515088321

Source: Own Table.

B.2.4 Flooding Frequency Results

Flowband Zonal Area (i)	Desired Flooding Volume and Frequency		Model Results
	Desired Inflow (GL) ($ER_{(i)}$)	Desired return interval (years) ($EF_{(i)}$)	Proportion of time desired return interval not met (%)
FB1	50	0.95	12.38
FB2	100	1.1	2.65
FB3	170	1.33	3.54
FB4	270	1.43	4.42
FB5	400	1.67	7.96
FB6	800	2	9.73
FB7	1700	4	33.62
FB8	2700	6.67	47.78
AVERAGE			15.26

Source: Own Table.

B.3 Output for Optimal Reallocation Scenario

B.3.1 Results Summary Table

	Reallocation Volume (GL/year)	Reallocation Cost ($ million)	Environmental water availability (GL/year)	Consumptive water availability (GL/year)	Water price ($/ML)	River Red Gum Carbon Volume ('000 tC)	River Red Gum Carbon Value ($ million)	Carbon Benefits Relative to Baseline ($ million)	Net Economic Benefit ($ million)
Baseline Conditions									
Average Annual [SD]	N/A	N/A	1175.6 [1233.7]	2127.9 [771.2]	114.3 [122.0]	4558.3 [724.3]	104.8 [16.6]	N/A	N/A
Minimum	N/A	N/A	0.0	502.0	20.0	2641.3	60.7	N/A	N/A
Maximum	N/A	N/A	7230.6	3620.0	560.8	5380.3	123.7	N/A	N/A
Sum	N/A	N/A	132845.1	240457.0	N/A	515088.3	11847.0	N/A	N/A
Annual Reallocation Conditions (CP=$30/tCO$_{2e}$, N=3)									
Average Annual [SD]	96.8 [160]	4.6 [14.9]	1272.5 [1232.6]	2031.0 [715.0]	121.5 [126.5]	4741.5 [768.4]	142.2 [23.0]	5.5 [4.2]	0.8 [15.6]
Minimum	0	0	0	487.5	20	2757.7	82.7	0	0
Maximum	624.5	147.8	7333.4	3389.1	570.9	5380.3	161.4	17.3	15.4
Sum	10948.4	526.9	143793.5	229508.5	13734.6	535792.7	16073.7	621.1	94.2
Annual Reallocation Conditions (CP=$48/tCO$_{2e}$, N=3)									
Average Annual [SD]	105.4 [176.2]	5.9 [16]	1281.0 [1233.3]	2022.4 [594.9]	122.8 [128.2]	4777.2 [778.5]	229.3 [37.3]	10.5 [13.2]	4.6 [21.1]
Minimum	0	0	0	719.7	20	2757.7	132.3	0	0
Maximum	791	147.8	7230.6	487.5	570.9	5380.3	258.2	54.7	50
Sum	11915.9	667.1	144761.1	228541.1	13885.4	539831.1	25911.8	1187.6	520.7

Source: Own Table.

B.3.2 Long-run Analysis Results

Table B.7 Baseline Modelled Carbon Storage in Short-run Analysis

Descriptive Statistics	Short-run (T=113) Baseline Carbon Storage (tC)	Long-run (T=113) Baseline Carbon Storage (tC)
Average	4558303	4431408.2
Minimum	2641280	2636359.2
Maximum	5380324	5380324.9
Standard deviation	724371	766900.0
Total summed over simulation	515088321	1001498274

Source: Own Table.

Table B.8 Baseline Modelled Carbon Storage in Long-run Analysis

Descriptive Statistics	Short-run (T=113) Optimal Carbon Storage (tC)	Long-run (T=113) Optimal Carbon Storage (tC)
Average	4741528.7	4531760.8
Minimum	2757749.3	2641280.0
Maximum	5380324.9	5380324.9
Standard deviation	768483.4	796862.6
Total summed over simulation	535792752.9	1024177957

Source: Own Table.

Table B.9 Difference Between Short and Long-run Analysis

Difference Between Short Run and Long-run Impacts	Average Annual Carbon Benefit [SD] (tC)	Total Carbon Benefit (tC)
Short-run	183225	20704431
Long-run	100353	22679683

Source: Own Table.

B.3.3 Comparison with Permanent Reallocation Strategies

Table B.10 Proportion of Time Desired Flood Interval Not Met for Annual Reallocation Strategy and Permanent Reallocations of 0-50%

Flowband	Proportion of Time Desired Flood Interval Not Met						
	Annual Reallocation	Permanent Reallocation Proportion					
	N=3, CP=$30/tCO$_{2e}$	0%	10%	20%	30%	40%	50%
1	12.38	12.4	5.3	1.8	1.8	1.8	1.8
2	2.65	2.7	1.8	1.8	1.8	0.9	0.9
3	3.54	3.5	3.5	3.5	2.7	0.9	0.9
4	4.42	4.4	3.5	3.5	3.5	0.9	0.9
5	7.96	8	8	8	6.2	0.9	0.9
6	9.73	9.7	9.7	8.8	8.8	0	0
7	33.62	33.6	33.6	33.6	33.6	3.5	0
8	47.78	47.8	47.8	47.8	47.8	28.3	22.1

Source: Own Table.

APPENDIX C: Estimating the Value of Environmental Water with Ecosystem Service Supply Modelling

Publication Details

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Publication Status	☐Published ☐Accepted for publication ☐Submitted for publication ☒Unpublished and unsubmitted work written in manuscript style
Publication Details	N/A

Principal Author

Name of Principal Author (Candidate)	Claire Settre		
Contribution to the Paper	Carbon model, Blue Green Algae model, Murray Cod Model, and erosion control model, re-allocation model development, economic valuation, sensitivity and uncertainty testing, model verification, model coding in GAMS, paper writing and editing.		
Overall percentage (%)	80%		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	28/8/2017

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

- the candidate's stated contribution to the publication is accurate (as detailed above);
- permission is granted for the candidate in include the publication in the thesis; and
- the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Jeff Connor		
Contribution to the Paper	Guidance in ecological and carbon modelling, water price model, assistance in paper structure, writing and editing.		
Signature		Date	21/8/2017

Name of Co-Author	Sarah Wheeler		
Contribution to the Paper	Assistance in paper structure, discussion, editing and revisions for publication.		
Signature		Date	21/8/17

C.1 Introduction

In a number of nations around the world increasing water abstraction to support growing agricultural and municipal demands has proceeded with little protection for the environment. This has resulted in the degradation of freshwater biomes (Abromovitz 1996; Dudgeon et al., 2006) and quality of ecosystem services (ES) they provide (Foley et al., 2007; Banerjee et al., 2013). Riparian restoration has become an important strategy for increasing the supply of ecosystem services in degraded freshwater ecosystems (Benayas et al., 2009; Bullock et al., 2011).

In the MDB, Australia, this is partially being addressed through market-based reallocation of water from consumptive users to the environment. Reallocation is governed by the Murray-Darling *Basin Plan* (Commonwealth of Australia, 2012), as discussed at length in Chapters 2 and 3. The Basin Plan stipulates that environmental water deliveries must be undertaken with regard to: i) *limitations on the effectiveness of environmental water, ii) cost effectiveness and ii) optimising economic, social and environmental outcomes* (*Basin Plan*, 2012, Chapter 8, Part 6, Division 1, Principle 5).

Despite these provisions, public policy debate has largely focused on the volume of environmental water as an indicator of environmental benefit and has implied a linear relationship between volume of water available and ecological outcomes (Crase et al., 2011). A lack of attention to the underlying ecological responses to environmental water availability can lead to environmental water targets which are not cost-effective nor efficient.

One way to evaluate the cost-effectiveness and efficiency of environmental water reallocation is to account for the marginal increase in flow-dependent ecosystem supply with respect to increasing reallocation volume. This can be achieved by quantitatively enumerating the linkages between environmental water availability, ecosystem health, ecosystem service supply, and economic value in an integrated hydro-economic modelling framework.

An extension of the hydro-economic model presented in Chapters 6 and 7 is presented here. The model is extended to incorporate a detailed representation of three additional ecosystem services in the Lower Murrumbidgee catchment. The ecosystem service considered are carbon storage and sequestration (detailed in Chapter 6), erosion prevention, freshwater provision and native fish habitat. As with the model shown in Chapters 6 and 7, the biophysical models of the four ecosystem services and the relationship to environmental water availability are linked to estimates of economic value. The model is then used to provide an estimate of the marginal benefit of increases in environmental water.

C.2 Literature Review and ES Supply Theory[83]

The state-of-the-art in ecosystem service modelling in the MDB is CSIRO (2012) and Bark et al. (2016)[84] which provide the first whole-of-MDB estimate of ecosystem service supply. Bark et al. (2016) simulate the provision of ES under natural, baseline, and *Basin Plan* scenarios to estimate the improvement in ES supply values achieved through improved environmental water availability. It was estimated that a total value between $3 billion and $8 bullion in ES benefits would be derived from improved water availability under the *Basin Plan* (CSIRO 2012). The majority of ES benefits will be derived from habitat services (between $3 billion and $8 billion) and improvements in carbon (between $120 million and $1 billion). An additional notable study is Akter et al. (2014) who link marginal willingness-to-pay (WTP) estimates for increases in waterbird species to changes in bird habitat suitability scores in the Macquarie Marches. A benefit of $0.15 million and $1.7 million was estimated for an increase of 104GL and 135GL river flow, respectively (Akter et al., 2014).

In these models, a static comparison approach using hydrological scenarios to calculate economic benefit was used. This involved modelling total ES supply for each hydrological scenario and comparing to the baseline to measure economic benefit. However, static comparison models are not fully applicable to determining the marginal improvement in ES supply for unit increases in environmental water or assessing the benefit of temporally variable reallocation strategies. Stakeholder engagement revealed a feeling of reduced credibility in the CSIRO (2012) modelling report due to lack of consideration of alternative flow scenarios or potential to achieve better environmental outcomes through alternative management of environmental water (Hatton-MacDonald et al., 2014). Fisher et al. (2008) and Haines-Young and Potschin (2010) argue that there remains a pressing need to improve capacity to account for the marginal change in value in response to alternative environmental conditions to evaluate trade-offs. In an extensive review, Seppelt et al. (2011) find that process-based simulation models are rarely used in ES modelling.

To address this gap in the literature and provide a means of estimating the marginal value of additional environmental water it is necessary to model the temporally dynamic links between water reallocation, ecosystem health and ES supply. In the MDB hydro-economic literature this has approached using predominately conceptual environmental representations rather than process-based models (e.g. Grafton et al., 2011b; Connor et al., 2013; Ancev 2013; Settre et al., 2017). In effect, this means that the first two aspects of the ecosystem services cascade ("biophysical structure or process" and "ecosystem function") (Haines-Young and Potschin (2010) are simplified to conceptual functions which do not necessarily fully capture the ecosystem dynamics. This also becomes a greater issue when

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[83] A wider literature review of ecosystem service modelling in a HEM framework is provided in Chapter 5. Further discussion of ecosystem service theory, modelling and valuation is provided in Chapter 6.
[84] Bark et al. (2016) report the results of an integrated ecosystem service assessment report published by CSIRO (2012).

ES supply is modelled with a linearly proportional to volumetric increase in environmental water, as is frequently assumed in the policy dialogue (Crase et al., 2011).

Ecosystem service value changes in proportion to condition of the environmental asset which may have non-linear response to water availability and may be subject to a number of ecological thresholds. This is shown conceptually in Figure C.1 which shows how assuming linear proportionality between the percentage of reallocation and ES generation can lead to a distorted picture of benefits from reallocation. ES supply models rooted in an understanding of the dynamics of the biophysical system are best placed to address complex issues on non-linearity, temporal distribution and marginality in integrated HEMs. By better accounting for the biophysical dynamics of ES supply, a more robust picture of environmental benefits can obtained and used to inform strategic cost-effective water reallocation decisions. Our case study of Lower Murrumbidgee attempts to do this by looking at environmental benefits of four flow-dependent ecosystem services: carbon sequestration, erosion prevention, freshwater and native fish habitat.

**Figure C.1 Conceptual Representation of ES Supply Value Diagram**
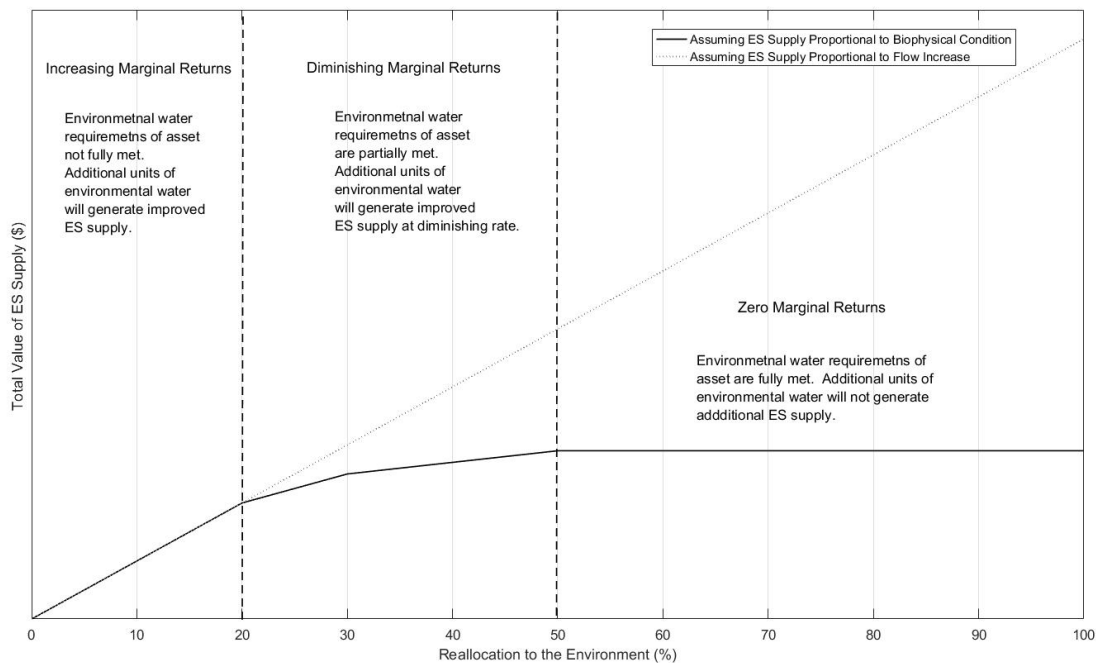
<u>Source:</u> **Own Figure.**

C.3 Methodology

C.3.1 Overview

An extension of the hydro-economic model presented in Chapter 6 is developed and shown in Figure C.2. The model uses the same catchment water balance, water price model and reallocation algorithm, as discussed in Chapter 6 and not repeated here.

Figure C.2 Extended Hydro-economic Model Schematic
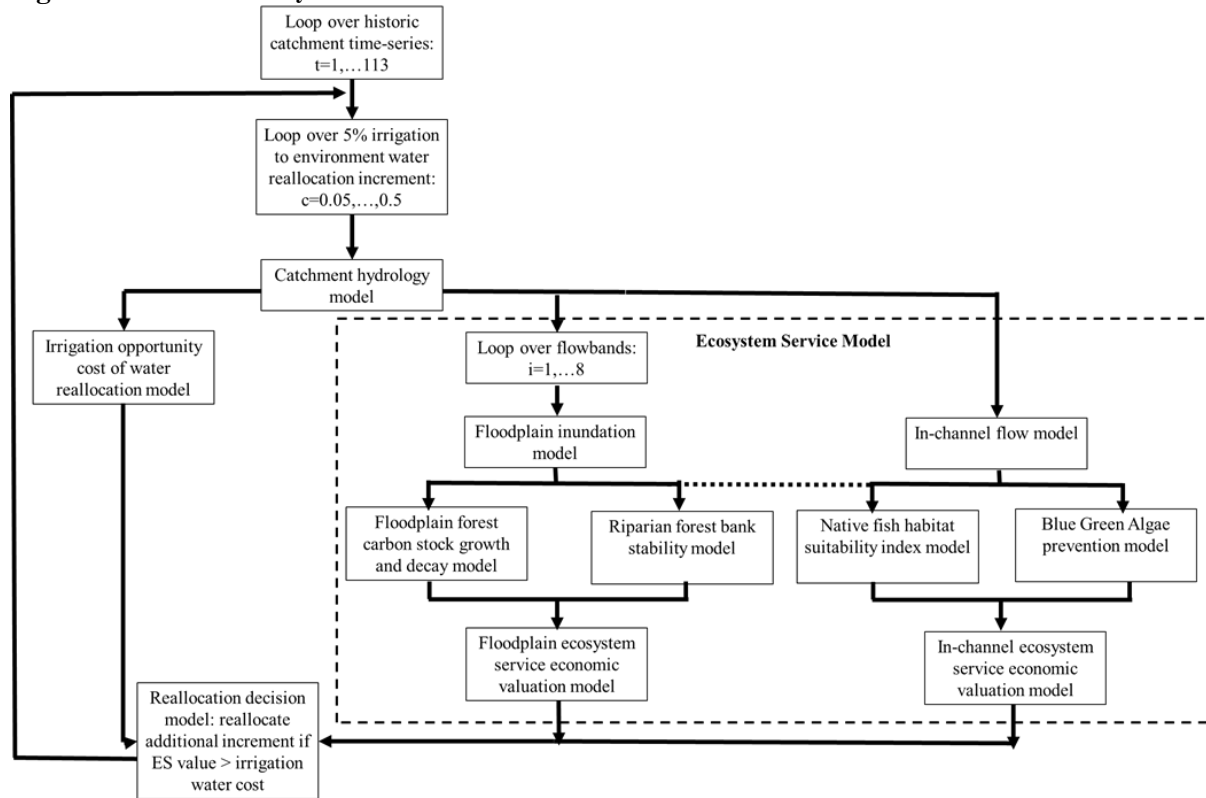
The extended model incorporates additional biophysical modelling and economic valuation for three new flow dependent ecosystem services. As can be seen in Figure C.2, the river flow output (volume, and timing, and flood and drought counters) is inputted into four separate biophysical models of ecosystem service supply. Two services generated from the floodplain or riparian area (carbon sequestration and erosion prevention) and two services generated in-channel (algae prevention and native fish habitat) are considered. Floodplain ecosystem services are driven by floodplain inundation modelling which determines the frequency and area of inundation by zonal areas of differing elevation and river connectivity. Inundation modelling results are input into models of River Red Gum carbon sequestration and bank stability dynamics. Monthly hydrological indicators (e.g. average velocity) of in-channel river flow are computed and tested against minimum flow requirements for algae formation to provide estimates of at-risk algae periods. In-channel flows are also input into the Murray Flow Assessment Tool (MFAT) Native Fish Habitat Model which stipulates ecological preference scores for in-channel generalists (e.g. Murray Cod) and outputs an annual habitat suitability index.

Two economic valuation components follow. First a regression-based water allocation price model, as described in Chapter 6. The second is the economic valuation of the four ecosystem services, which are valued using market pricing, avoided cost, defensive expenditure, and benefits transfer. Table C.1 provides an overview of the modelling and economic valuation approaches for each ecosystem service

considered. The economic value of each incremental increase in environmental water availability is estimated by summing the economic value of the improvements in ecosystem services supply.

Table C.1 Overview of Ecosystem Service Biophysical Modelling and Valuation

Ecosystem Service	ES Supply Indicator	Hydrological driver	Modelling Approach	Valuation Approach	Unit Value
Carbon storage and sequestration	River Red Gum carbon stocks	Floodplain inundation	Stock and flow modelling	Market valuation; social-cost	$10/tCO$_{2e}$ - $48/tCO$_{2e}$
Erosion prevention	Erodible streambank area	Riparian zone inundation	Critical threshold approach	Avoided cost	$25/m2 - $62/m2 man-made erosion control
Freshwater (Blue-green algae prevention)	Flow velocity	Minimum flow velocity	Critical threshold approach	Avoided cost	$200/ML added water allocations
Native fish habitat	Habitat suitability modelling	Flow volume, velocity, timing and flooding	Habitat preference modelling	Marginal willingness-to-pay	$395,884/1% increase - $119,556/1% increase

Source: Own Table.

C.3.2 Modelling and Valuation Assumptions

To achieve the level of integration required a number of assumptions were required to be made. The following section provides an overview of key modelling and valuation assumptions.

C.3.2.1 *Ecosystem Service Supply Modelling Assumptions*

Integrating multiple ES supply models into a single HEM involved considerable data manipulation which required a number of modelling assumptions to be made. For example, it is assumed that intra-annual supply profiles follow the historic pattern, although it is known that as the climate changes the pattern of intra-annual water supply has altered (Murphy and Timbal, 2008) and has become more variable overall (Connor et al., 2012; Grafton et al., 2014).

Habitat condition was used as a proxy for Murray Cod population, which implies a linearity between population abundance and the proxy indicator. This approach does not account for wider influences on population dynamics; including ecosystem carrying capacity, predation, competition and anthropological disturbance. Debate regarding the reliability of habitat suitability scores as a proxy for fish population is ongoing in the literature (Roloff and Kernoham, 1999; Favata et al., 2015).

The modelled ES derived from River Red Gum are assumed to be proportional to tree health which is driven by the overbank flooding regime. For example, modelled erodible area exceeds zero when riparian River Red Gums decay in health. Alternative predictors of stream bank erosion may provide a more nuanced understanding of bank erosion, including erosion due to high stream power during floods which is not included in this model.

Further, the HEM considered ecosystem services generated by biophysical structures in the Lowbidgee only, although there are likely to be significant downstream water quality benefits. A considerable underestimate of basin-wide ES benefits from reallocation is therefore provided. Greater consideration regarding the downstream co-benefits of upstream watering and the future flows of ES warrants future research.

C.3.2.2   Economic Valuation Assumptions

There are a number of valuation challenges stemming from uncertainty in the supply of ES, ES management options, and society's preferences, as discussed at length in Chapter 6. Valuation uncertainty is partially mitigated by providing a range of possible values (e.g. multiple carbon market prices) to assess sensitivities.

The market price of carbon is assumed to be constant throughout the simulation when in fact the price of carbon is seasonally variable and influenced by changing policies and laws. Highly variable carbon prices would likely influence the value of water reallocation and alter annual reallocation decisions. The social cost of carbon is likely to increase in the future (Interagency Working Group on Social Cost of Carbon, 2013) and there is considerable debate regarding the choice of discount rates for climate related policies (Fleurbaey and Zuber 2013). Therefore it is possible that the estimate is a substantial underestimate of carbon value obtained by reallocating water to the environment.

In application of the avoided cost methodology it is assumed that expenditure occurs in the year in which a biophysical threshold is surpassed. For example, modelled erosion control measures are deployed in years where the potential erodible area exceeds zero. In reality it is likely that action is precipitated by some social or economic trigger, such as when the anticipated damage costs exceed the cost of erosion control measures. It is unlikely that costs associated with erosion control measures will be incurred in single years but are likely to be distributed through time in line with project schedules and budgets. This nuance is not accounted for in the model and it is possible therefore that the annual cost of erosion is overstated. However, as the erosion prevention ES is not a driving factor in reallocation benefits and results are not sensitive to this assumption.

The avoided costs are modelled based on one management option. For example, the avoided cost of Blue-green algae prevention is determined using costs of in-situ bloom management through flow releases. Alternative management options for algae management exist, but cost estimates for these management strategies as data pertaining to algae prevention decision-making is not readily available and is largely dependent on local management targets. Therefore a single example of a multi-pronged management strategy which may differ under various conditions is provide. Estimates of avoided cost approximate capital expense savings only and the avoided cost of labour, maintenance or monitoring is not accounted for, therefore providing an underestimate of possible avoided costs.

Willingness-to-pay survey data transferred to estimate non-market value of fish populations is dated (2001) and it is possible that the value attributed to native fish by society have changed considerably, particularly as the originally study was conducted during a period of extensive drought and media coverage of native fish population decline was high. As the Lowbidgee is an important NSW asset, but also a water source for Canberra (ACT), it is unclear which population would be best used to aggregate the non-market value. Preferences in out-of-basin communities are likely to be considerably different to those within the MDB. However, Bark et al. (2016) argued that inclusion of non-MDB communities is a valid approach as it fits with the objectives of Australian water reform which seeks to generate benefits for the whole Australian community. The results reported aggregate over the MDB household population only, and therefore likely provide an underestimate of the total non-market value of improvements in native fish habitat. Further, because native fish populations are severely degraded in the Murrumbidgee (Lintermans 2007) it is assumed that modelled population improvement occurs on the section of the total utility curve where marginal utility is increasing and marginal are not yet diminishing. This implies that additional 'units' of fish will continue generate benefit.

C.3.3 Modelling Ecosystem Service Supply[85]

C.3.3.1 Erosion Prevention

Reinforcing streambanks with native vegetation reduces the risk of erosion and bank collapse by altering the bank hydrology and the geotechnical properties of the bank soil (Abernethy and Rutherford, 1999; Abernethy and Rutherfurd, 2000; Wynn, 2004). Controlling erosion in Australian rivers is best achieved by restoring native riparian vegetation to stabilise stream banks (Hansen et al., 2010). The addition of native vegetation such as River Red Gums to riverbanks improves bank stability even under worse case hydrological conditions and for a range of bank geometries (Abernethy and Rutherford, 2000).

Erosion rate models relating river flow volume, riparian tree health and potential erodible area is developed for three conditions of River Red Gum riparian vegetation. To achieve this, existing biophysical equations expressing bank erosion rates as a function of the flooding regime are first employed. Rutherfurd (2000) suggest the following:

$$BE_{(t)} = 0.016Q(1.58)^{0.6} \qquad \textbf{(Equation A.1)}$$

Where $BE_{(t)}$ is the bank erosion in metres per year, $Q(1.58)$ is the volume of flood event (m³/s) at the 1:1.58 year recurrence interval and $Q(1.58)$ is assumed to be the bankfull flow[86] (Hughes and

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[86] The term *bankfull* refers to the case when water in the river is high enough that it just begins to spill out onto the hydrological floodplain. *Bankfull flow* refers to the volume of flow that is required to achieve bankfull stage.

Prosser, 2003). This approach has precedence for use in Australia (e.g. National Land and Water Audit (1997-2008)). However, this equation was found to predict high erosion rates in catchments with high discharge and undisturbed vegetation (Hughes and Prosser, 2003). Hughes and Prosser (2003) amended the equation to:

$$BE_{(t,x)} = 0.008\left(1 - PR_{(t,x)}\right)\left(Q(1.58)\right)^{0.6} \qquad \textbf{(Equation A.2)}$$

Where $PR_{(t,x)}$ is the proportion of intact native riparian vegetation and $BE_{(t,x)}$ is the annual bank erosion and $Q(1.58)$ is bankfull flow given the condition and extent of vegetation in each year, governed by x. Bankfull flow in the Lowbidgee occurs significantly more infrequently than the 1:1.58 recurrence interval due to river regulation and altered flow regimes (Page et al. 2005). In the hydro-economic model, the volume and frequency of bankfull flows which generate overbank flooding can be altered by the decisions of an actively trading EWH who has the option to reallocate a percentage, c, of consumptive water to the environment. Further, the floodplain (including the riparian zone) is grouped by zonal areas inundated by flow categories, i. Therefore, the rate of erosion is modelled in the HEM as:

$$BE_{(t,i,c,x)} = 0.008\left(1 - PR_{(x,t,i)}\right)\left(Bankfull\,Q\right)^{0.6} \qquad \textbf{(Equation A.3)}$$

Where $Banfull\,Q$ is the flow at bankfull stage. Under baseline conditions bankfull flow at Balranald is a flow of 12,330 ML/day which occurs at a return period of 2.33 years and has a mean bankfull stage height of 5.92m, as shown in

Table C.2 Bankfull Flow in the Murrumbidgee Catchment, 1970-1998.

Site	Bankfull Stage (m)	Bankfull Discharge (ML/day)	Return Period (years)	
			Regulated Flows	Natural Flows
Gundagai	6.95±0.22	62,660	2.44	1.61
Wagga Wagga	6.88±0.23	52,090	1.95	1.47
Narrandera	6.64±0.41	38,220	2.13	1.45
Darlington Point	5.41±0.27	26,460	1.89	1.369
Hay	7.93±0.38	27,010	2.5	1.54
Balranald	5.92±0.09	12,330	2.33	1.19

Source: Page et al. (2005), p. 573.

The degree to which riparian vegetation is able to reduce erosion rates is a function of the riparian zone width (Figure C.3) and the vegetation condition in this zone Abernethy and Rutherford (1999). A range of minimum riparian zone width recommendations are available. Abernethy and Rutherford (1999) recommend a minimum 5 meters.

Figure C.3 Cross-sectional Channel Geometry.
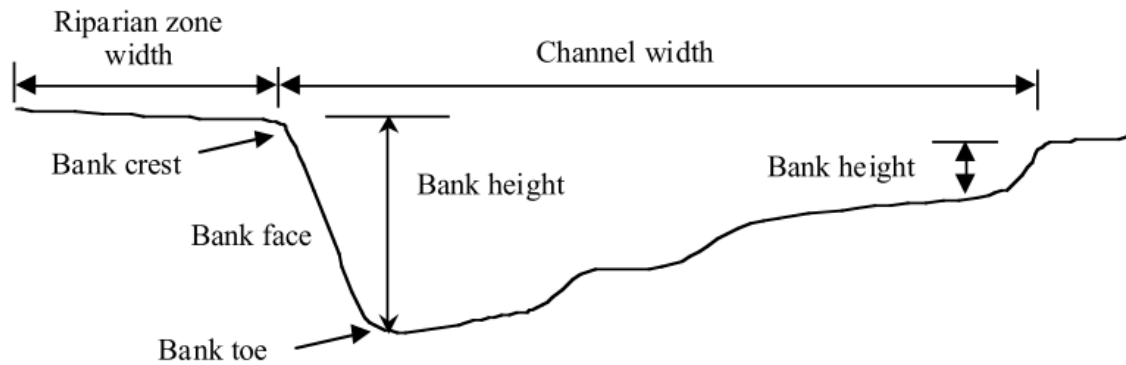
For vegetation which is immature, degraded or lacking a structural element the design width for riparian vegetation is calculated as the minimum width requirements plus the bank height (Abernethy and Rutherfurd, 1999). Using bankfull height as the bank height, the design width is 10.92m. To be effective the buffer must extend along the length of the stream. The length of the Lowbidgee floodplain from Hay to Balranald is found taking measurements from satellite imagery and is 390km (a total of 780km both sides of the river). The area of the minimum riparian corridor providing bank stabilization is therefore 851.76ha. From Table 6.11 (Chapter 6) it is known that a flow of 50GL per year (inundating flowband zone 1), 1,073ha of River Red Gum area is inundated at the desired return interval. Therefore, to maintain the health of River Red Gums contributing to erosion prevention in the riparian corridor a minimum flow of 50GL/year is required at a return interval of 0.95:1.

Erosion rates were calculated using Equation A.3 for three conditions of riparian vegetation: bare bank, River Red Gum decay, and River Red Gum growth/maintenance.

A 'bare bank' condition is considered which occurs when the bank is completely un-vegetated. This is an estimate of the maximum possible erosion rate. Under bare bank conditions:

$$BE_{(t,i,c,x=0)} = 0.156m / year \qquad \textbf{(Equation A.4)}$$

Where $BE_{(t,,i,c,x=0)}$ is the annual bank erosion, t is the annual time step, i is the flowband area containing the riparian zone , c is the reallocation proportion, and x is the proportion of bank of bank vegetation.

'Healthy' riparian vegetation made up of contiguous stand of mature riparian forest established with the minimum width and provides the highest level of erosion prevention (Abernethy and Rutherford, 1999). This occurs when environmental water requirements of the riparian area are fully met. Under these conditions it is assumed that $PR_{(x,t)} = 1$ (Equation A.3) such that the bank erosion for this case is 0m/year.

$$BE_{(t,i,c,x=1)} = 0.0m \, / \, year \qquad\qquad \textbf{(Equation A.5)}$$

When the desired environmental watering conditions are surpassed, the health of the riparian River Red Gums decay and is considered water stressed. Riparian vegetation which lacks structural elements or is generally degraded performs no stabilizing streambank role (Abernethy and Rutherford, 1999). It is assumed that there is a linear relationship between vegetation coverage and vegetation health such that $PR_{(x,t)} = 0.5$ (Equation A.3). Therefore, for cases where riparian ecosystem is in a decay dynamic, erosion is found to be:

$$BE_{(t,i,c,x=1)} = 0.078m \, / \, year \qquad\qquad \textbf{(Equation A.6)}$$

While it is known that the relationship between tree health and bank stability is likely to be non-linear, no empirical evidence relevant to the current study was identified that could otherwise determine the functional form of the relationship. Abernethy and Rutherfurd (2001) show that the relationship between bank stability and vegetation cover is dependent on the root strength, interface friction and root distribution within the soil. Such details are unavailable for the case of the Lowbidgee and the linear assumption is considered acceptable in absence of these details.

C.3.3.2 *Murray Cod Habitat*

When the data to undertake detailed stock and flow population modelling of fish populations is unavailable, habitat suitability scores as proxy indicators are a useful alternative (Stephens, 2015). Habitat suitability models use species preference curves which represent a relationship between a target organism and its response to gradual change in habitat conditions. Importantly, the Murray Cod habitat ecosystem service included in the HEM is a measure of habitat existence and suitability, not the population of species which use the habitat. However in the ecological literature there is an established precedence for using habitat suitability as an indicator of population health and *vice versa* alternative (Stephens et al., 2015).

The Murray Flow Assessment Native Fish Habitat Condition (MFAT NFHC) is used to model native fish habitat condition response to changes in the hydrological regime (e.g. flood timing, flow duration, maintenance flow etc.). The MFAT NFHC outputs habitat suitability scores for various species of native fish based on a series of user-established preference curves for a range of hydrological and biophysical events. Although MFAT has an established user interface, the preference curves were developed and re-coded in GAMS to allow the maximum level of integration with the existing HEM. The code was developed following the MFAT Technical Guidelines (Young et al., 2003). The preference curves are modelled for Murray Cod (*Maccullochella peelii*), which is an instream specialist fish and a Living Murray Icon species. The vast majority of the described preference curves are identical to the MFAT modelling, although some alterations have been made in the re-coding process to fully integrate the habitat suitability model into the HEM. In particular, preference curves expressed in days were scaled

up to be expressed in months, which is the minimum time-step in the GAMS model described in this thesis. This change from the original particularly affected temporal variables, such as; inundation duration, dry period and fish passage. The process used to determine monthly flows from the annual series, assuming equal distribution of flow between the days of each month, is shown in Appendix B. The following equations provide an overview of the model, and further technical detail is provided in Young et al. (2003).

Overall, native fish habitat condition (FHC) is given as:

$$FHC = x_1 AHC_{(c,t)} + x_2 RHC_{(c,t)} \qquad \textbf{(Equation A.7)}$$

Where $AHC_{(t,c)}$ is adult recruitment habitat condition in period, t, for annual reallocation proportions, c. $RHC_{(t,c)}$ is recruitment habitat condition and x_1 and x_2 are user-defined weightings (Young et al., 2003). Factors affecting $AHC_{(t,c)}$ and $RHC_{(t,c)}$ vary for different native fish species based on spawning and recruitment preferences for hydrological conditions.

For Murray Cod, adult habitat condition ($AHC_{(t,c)}$) is given as:

$$AHC = x_3 WD + x_4 FP_{(t,c)} + x_5 WT + x_6 CC_{(t,c)} + x_7 MF_{(t,c)} \qquad \textbf{(Equation A.8)}$$

The variables are defined in Table C.3

Note that some variables are annually variable (functions of t) and are influenced by environmental water management decisions (function of c). Other variables are not influenced by annual decision-making and are held constant across all reallocation scenarios.

Recruitment habitat condition ($RHC_{(t,c)}$) is a weighed sum of spawning habitat condition (SHC) and larval habitat condition ($LHC_{(t,c)}$). Variable definitions are summarised in Table C.4.

$$RHC_{(t,c)} = x_8 SHC_{(t,c)} + x_9 LHC_{(t,c)} \qquad \textbf{(Equation A.12)}$$

For Murray Cod, SHC is a function of spawning timing (ST) and rate of fall (RF_s).

$$SHC = \sqrt{ST + RF_s} \qquad \textbf{(Equation A.13)}$$

Note that spawning timing is a constant species preference and cannot be altered by annual EWH decision-making, hence is not expressed as a function of t or c indices which govern time and

reallocation proportion in the HEM. In high-flow recruitment mode[87] for in-channel specialists, $LHC_{(t,c)}$ is given by:

$$LHC_{(t,c)} = \sqrt[3]{IA_{(t,c)} * ID_{(t,c)} * DP_{(t,c)}}$$ **(Equation A.14)**

Table C.3 Variable Definitions for Adult Habitat Condition Equation

Variable	Definition
x_i	Weighting
WD	Woody debris
WT	Water temperature
$FP_{(c,t)}$	Fish Passage
$CC_{(t,c)}$	Chanel condition
$MF_{(t,c)}$	Maintenance flow

Source: Young et al. (2003).

Table C.4 Variables and Definitions for Recruitment Habitat Condition Equation

Variable	Definition
x_i	Weighting
$RHC_{(t,c)}$	Recruitment habitat condition
SHC	Spawning habitat condition
$LHC_{(t,c)}$	Larval habitat condition
ST	Spawning timing
RFs	Rate of fall (spawning)
$IA_{(t,c)}$	Inundation area
$ID_{(t,c)}$	Inundation duration
$DP_{(t,c)}$	Dry period

Source: Young et al. (2003).

C.3.3.3 *Blue Green Algae Prevention*

In low lying rivers such as the Murrumbidgee, low water discharge (Webster et al., 2000; Sherman et al., 1998), thermal stratification (Maier et al., 2001; Mitrovic et al., 2011) and long water retention times (Oliver and Gnaf, 2000) have been identified as factors causing the creation and persistence of blue green algae blooms. Under these conditions one approach to mitigate favourable Blue Green Algae conditions is to manipulate water releases to prevent prolonged periods of thermal stratification by providing a minimum flow velocity (Mitrovic et al., 2003; Mitrovic et al., 2011). The relationship between average daily discharge and maximum temperature difference (indicative of thermally stratified conditions) derived for conditions in the Lower Darling River is provided in Figure C.4.

Figure C.4 Relationship Between Discharge and Maximum Daily Change in Temperature in the Lower Darling River
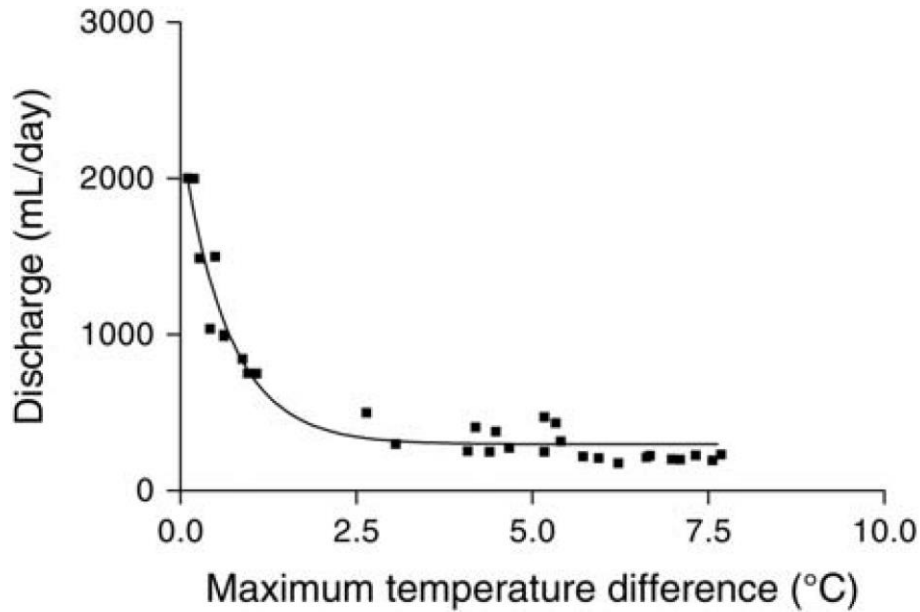
Thermal stratification tends to zero as daily discharge increases above 300ML/day (Figure C.4) which corresponds to a mean flow velocity of 0.03m/s. In the Lower Darling case study, a flow velocity of 0.03m/s was sufficient to prevent extended periods of thermal stratification and suppress the development of Blue Green Algae blooms. Comparable critical velocity thresholds have been identified in other case studies (Mitrovic et al., 2006) and a critical threshold of 0.03m/s is likely to have broader application to rivers in the MDB (Mitrovic et al., 2011).

This approach is therefore applied to the Lowbidgee. Daily flow is determined using the method described in Appendix B, assuming equal distribution of flow between days in the month. For each year in the model, t, the average monthly and daily discharge volumes for baseline conditions $Q_{0(t)}$ are determined. The flow required to be added annually to prevent the formation of favourable Blue Green Algae conditions, $\Delta Q_{(t)}$ is given as the difference between the average daily discharge $Q_{0(t)}$ for baseline conditions and the minimum flow $Q_{1(t)}$ able to achieve a flow velocity of 0.03m/s. Mathematically:

$$\Delta Q_{(t,c)} = Q_{1(t,c)} - Q_{0(t)} \qquad \textbf{(Equation A.17)}$$

Where:

$$\frac{Q_{0(t)}}{A} < 0.03 m/s \qquad \textbf{(Equation A.18)}$$

And:
$$\frac{Q_{1(t,c)} + Q_{0(t)}}{A} > 0.03 m/s$$
(Equation A.19)

Note that flow in Equation A.17 to A.19, flow is calculated in meters cubed per second. The annual flushing volume is calculated by summing over the intra-annual flow releases for 'at risk' periods to determine the total annual volume released.

C.3.4 Valuing Ecosystem Service Supply

To measure the ES benefit of reallocation and inform cost-effective reallocation it is necessary to value the marginal changes in ES supply in response to changes in environmental water availability. The following section details the economic valuation methods used. Carbon storage valuation is presented in Chapter 6.

C.3.4.1 *Erosion Prevention*

Erosion prevention is valued using an avoided cost approach. The avoided cost is estimated using the cost of employing man-made erosion control measures to prevent erosion in the absence of bank-stabilizing riparian vegetation. Vegetation cover is a recommended erosion control strategy (Abernethy and Rutherford, 1999) in Australian rivers, and therefore the cost estimate reflects the cost of constructing a native vegetated riparian buffer strip to control bank erosion. The cost of the buffer strip is sourced from Water Sensitive Urban Design Technical Guidelines (2004) which stipulates that the cost of a buffer strip consisting of native grasses and shrubs is between $25-62 per square meter (2015 dollars). The cost of ongoing maintenance is not considered, estimated to be between $212/m/year (2015 dollars) for the life of the buffer strip (Lloyd et al., 2002). The annual avoided cost of erosion prevention is:

$$AC_{erosion(t,c)} = BP\left(BE_{(t,i,c=1,x)} - BE_{(t,i,c>1,x)}\right)$$
(Equation A.20)

Where BP is the unit ($/m) cost of erosion control, $BE_{(t,i,c,x)}$ is the modelled potential erodible area of the riparian corridor in each simulation period, t, for reallocation percentages, c (where $c = 1$ represents the no reallocation scenario). Note that as only the immediate riparian corridor contributes to erosion prevention, it is not necessary to sum across the flowband areas.

C.3.4.2 *Blue-green Algae Prevention*

The avoided cost approach is also used to value the algae prevention by estimating the value of the volume of water released downstream to prevent bloom formation. The cost of water released is valued using the long-term average cost per mega-litre of water. As algae flushing water is released on an annual basis, it is appropriate to use the price of temporary (allocation) water rather than the price of permanent (entitlement) water sales. The average water allocation price in the southern MDB from 2002 to 2015 is approximately $200/ML. The average allocation price was used to avoid issues of endogeneity between water prices and avoided cost of blue-green algae prevention. This was required

because releasing a flushing flow removes water from the consumptive pool which drives modelled water price up, which would inflate the modelled avoided cost value of algal prevention. The avoided annual cost of blue-green algae prevention is:

$$AC_{a\lg ae(t,c)} = WP_{(t)}\left(Q_{(t,c=1)} - Q_{(t,c>1)}\right)$$ **(Equation A.21)**

Where $WP_{(t)}$ is the average water allocation price and $Q_{(t,c)}$ is the annual flow volume released to prevent the formation of favourable algal conditions each simulation period for each reallocation proportion.

C.3.4.3 Native Fish Habitat

The non-market value of native fish habitat is estimated using marginal willingness-to-pay survey results obtained from Whitten and Bennett (2001) and transferred to the current context. In this case native fish populations are used as a proxy indicator for habitat condition, as has precedent in applied ecology (Stephens et al., 2015). Whitten and Bennett (2001) use a choice experiment to elicit participant's preferences for improvement in ecological condition in the Murrumbidgee wetlands. Choice experiment methods involve the construction of a hypothetical market for environmental attributes to elicit willingness-to-pay (or accept) for improvements in asset condition (Louviere et al., 2000).

Benefit transfer is the method of transferring results of existing studies to new contexts by extrapolating value estimates from the original (study site) to a similar context (policy site) (Brouwer, 2000). Four options for transferring non-market values: a) unit transfer; b) adjusted unit transfer; c) value function transfer; d) meta-analytic function transfer. When high-quality primary valuation is available for catchments with very similar characteristics to the policy site, unit benefits transfer may result in the most precise value estimate (Pascual et al., 2010).

The primary study selected for transfer is a choice experiment conducted by Whitten and Bennett (2001). This study estimated the non-market value of improvement of Mid-Murrumbidgee wetland ecological assets including; native fish, native birds, area of native vegetation, and number of farmers exiting farming. The survey was mailed to participants in Griffith (800), Wagga Wagga (800), Canberra (800) and Adelaide (400) and there was a response rate of 30.2%[88]. All surveyed participants live within the MDB and Griffith, Wagga Wagga and Canberra participants are within the Murrumbidgee catchment. Survey participants were asked to choose a once-off levy of $0 (no management), $20 and $50 to achieve a range of wetland asset improvements (Whitten and Bennett 2001). It was found that the mean implicit price for a 1% increase in native fish population in the Murrumbidgee is $0.34 (95%

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[88] 2,800 surveys were mailed out, 378 were returned to sender, and 732 were completed and returned (Whitten and Bennett, 2001).

CI: \$0.24-\$0.45) in 2000 dollars. Accounting for inflation between 2000 and 2015 this results in a mean implicit price is \$0.47 per 1% increase in native fish populations.

The applicability of unit benefit transfer is largely dependent on the similarity between study and policy sites. The study site and policy site are both within the Murrumbidgee catchment, therefore the transfer must account for changes in condition and demographics over time only. Whitten and Bennett (2001) conducted their study during a time of the prolonged Millennium Drought. A valuation survey undertaken after the drought is likely to generate a higher implicit price because habitat conditions suffered considerably during the drought (Akter et al., 2014). It is also possible that environmental attitudes have changed with the passage of ecologically focused legislation (e.g. *The Basin Plan*). It is most likely that the CPI adjusted WTP estimate is an underestimate of society's current wiliness-to-pay for improvement in native fish population. In addition, the overall condition of the Murrumbidgee catchment has improved since the end of the drought due to improved environmental watering events in mid-2010 (Doody et al., 2015). Murray Cod populations in the Murrumbidgee were depleted during drought years (Davis et al., 2010) however no evidence to suggest their population has improved much since the time of the Whitten and Bennett (2001) study[89] (DoE 2017). Demographic data from the study site (2001) was compared to 2011 ABS Census data to determine relevant changes. Notably, the average proportion of tertiary education has increased by 2% and average personal income has decreased by \$4,000. There is negligible difference in the distribution of sex or age between current period and the survey year. Comparison of these demographics suggest that it is possible to use a direct unit transfer and adjusting for inflation between 2001 and 2015. A direct unit transfer derives comparable aggregate results to CSIRO (2012) and Bark et al. (2016) for the Murrumbidgee catchment.

Non-market Value Aggregation

Aggregating non-market values involves extrapolating the sample estimates across the population. However, the general population may not share the same characteristics of the sample population (Morrison, 2000). Morrison (2000) suggests two methods to address this; a) adjusting the sample means to account for differences between sample and population, or b) applying assumptions to account for the preferences of non-respondents. The second approach is adopted and has precedence in MDB valuation literature (Hatton-MacDonald and Morrison, 2005; Morrison and Hatton-MacDonald, 2010; CSIRO 2012). This involved extrapolating the non-market value over 30% of non-respondents to account for respondents which likely hold value for Murrumbidgee assets but did not return the survey due to external factors (e.g. 'too busy' (Morrison, 2000)). The resulting estimate is bound by a conservative estimate assuming non-respondents attribute zero value to Murrumbidgee assets (30.2% of population), and a least-conservative estimate that the entire aggregate population has the same

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[89] The International Union for the Conservation of Nature (IUNC) lists Murray Cod as 'critically endangered' in 2015 and state based Australian legislation lists Murray Cod as 'threatened' in 2015 (Department of the Environment and Energy, 2017).

values as those held by survey respondents (100% of population). Three household populations for aggregation were investigated: MDB, NSW, and NSW excluding Sydney. Sydney was excluded in one aggregation possibility because Bark et al. (2016) report that aggregating non-market values over the Sydney (a highly urbanised city outside of the MDB) population reduces stakeholders confidence in valuation results. Table C.5 shows aggregation values for the MDB population.

Table C.5 Aggregate Non-market Value

Aggregation approach (%)	Aggregation population (MDB households)	Native fish value estimate ($/1% increase)
100%	842,307	$395,884
51.1% (survey response rate +30% non-response rate)	430,418	$202,296
30.2% (survey response rate)	254,376	$119,556

Source: Own Table.

For the values shown in Table C.5, the non-market value of fish population improvements is given by:

$$FV_{(t,c)} = IP\left(HSI_{(t,c>1)} - HSI_{(t,c=1)}\right)$$ **(Equation A.22)**

Where $FV_{(t,c)}$ is the value of the change in fish population, IP is the implicit price for a 1% increase in native fish, and $HSI_{(t,c)}$ is the habitat suitability index for Murray Cod in each period, t, for reallocation proportion, c.

C.3.4.4   Economic Value of Modelled Ecosystem Service Supply

The annual aggregate incremental change in the value of the modelled ES supply for all reallocation scenarios (e.g. $c > 1$) relative to the baseline conditions is given by the addition of the use and non-use values described above. That is:

$$ES_{value(t,c)} = CV_{(t,c)} + AC_{erosion(t,c)} + AC_{\text{alg}ae(t,c)} + FV_{(t,c)}$$ **(Equation A.23)**

The total value over the simulation (113 years) relative to baseline conditions is therefore given by Equation A.24. ES values are also annualised to give an estimate of the annual flow of ES benefits in $/year.

$$ES_{value(t,c)} = \sum_{t=1}^{113} ES_{value(c)}$$ **(Equation A.24)**

C.4 Results

C.4.1 Baseline Ecosystem Service Supply

Despite considerable ecological degradation (Davies et al., 2010), the Lower Murrumbidgee catchment provides considerable ecosystem service supply under baseline (no reallocation) conditions. However, ecological condition and ES supply in the Lower Murrumbidgee is highly variable. In particular, extended drought sequences at the beginning and end of the hydrological series reduces ecological condition and ecosystem supply. Table C.6 summarises the indicators of ecosystem service supply and value and Figure C.5 to Figure C.8 demonstrate the distribution of ecosystem service supply over the simulation period.

**Table C.6 Indication of ES Supply under Baseline Conditions**

Ecosystem Service	Indicator of ES supply	Indication of ES Supply
Climate regulation	River Red Gum carbon storage	43.3% of years floodplain River Red Gums store less than mean carbon capacity
Bank stability	Area of erodible streambank	12.4% of years erodible streambank area exceeds zero
Freshwater (algae prevention)	Flow velocity in summer months	12.4% of years contain summer flows below velocity threshold
Native fish habitat	Native fish habitat suitability	Mean 0.72 native in-channel specialist habitat suitability score

Source: Own Table.

Under baseline conditions, there are significant decay in River Red Gum carbon stocks in the period following the Federation Drought (1895-1903) and during the Millennium Drought (1997-2010). During these periods the frequency and volume of environmental watering far exceeds the desired watering requirements for mid- and high-elevation floodplain areas and River Red Gum carbon stocks subsequently decay. Environmental watering requirements across the floodplain are not met on average of 15% of the time. For baseline conditions the River Red Gum forest stores a mean carbon volume of 4.5 million tC, corresponding to a market value of \$45-\$218 million a carbon price of \$10-\$48/tCO$_{2e}$, respectively. Summer months (January, February, March) containing flow below the minimum velocity requirements for Blue Green Algae prevention occur 12% for baseline conditions. These years are classified 'at risk' of Blue Green Algae formation and occur primarily during extended dry periods and intermittently in isolated dry year events. The modelled cost of purchasing additional allocation water to increase flow velocity is on average \$5.3 million per annum for a water allocation price of \$200/ML. Years at risk of Blue Green Algae formation coincide with periods where potential streambank erosion exceeds zero. This occurs in years where flow is consecutively insufficient to inundate riparian vegetation responsible for stream bank stabilization. Modelled annual expenditure to prevent erosion through human intervention is between \$4,900 and \$12,290 for a unit buffer replacement cost of \$25 - \$62/m². Modelled Blue-green algae and erosion prevention expenditure is highly variable and occurs primarily in the Federation Drought and Millennium Drought period. This results is consistent with

These results confirm findings by Banerjee et al. (2013) who identify considerable expenditure during the drought to maintain status-quo supply of provisioning and regulating services in the Lower Murray region. Native fish habitat suitability varies annually under bassline conditions and on average provide a habitat suitability score of 0.72. As native fish habitat is valued using an estimate of marginal WTP (e.g. $ per 1% increase) there is no monetary value attributed to baseline fish populations. Note that this does not imply zero value for baseline conditions.

Figure C.5 Modelled Erodible Riparian Area for Baseline Conditions
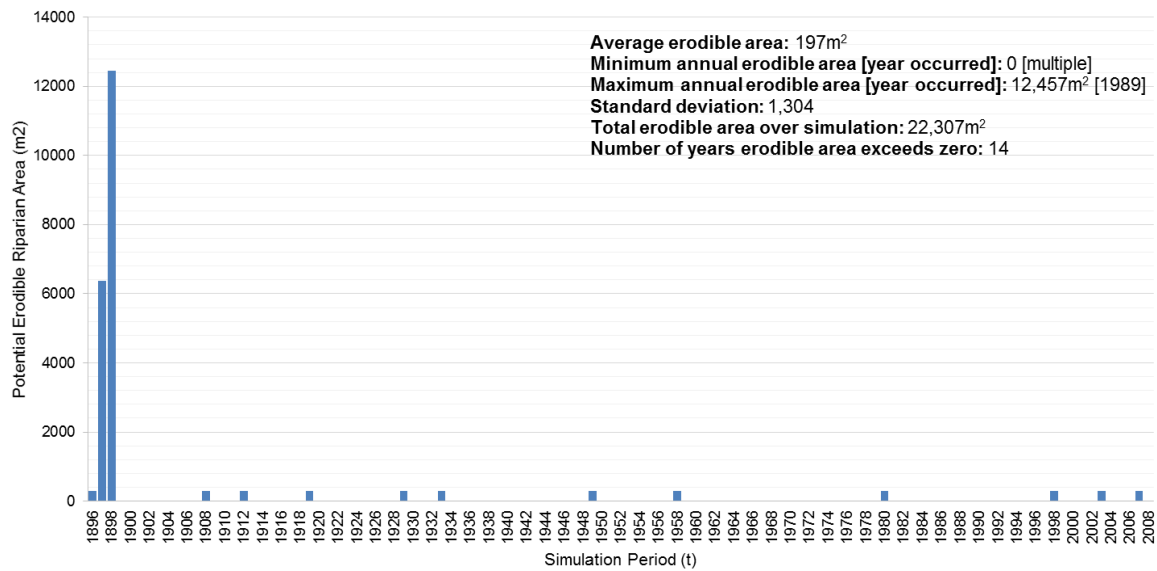
Source: Own Figure.

Figure C.6 Modelled River Red Gum Floodplain Carbon Storage for Baseline Conditions
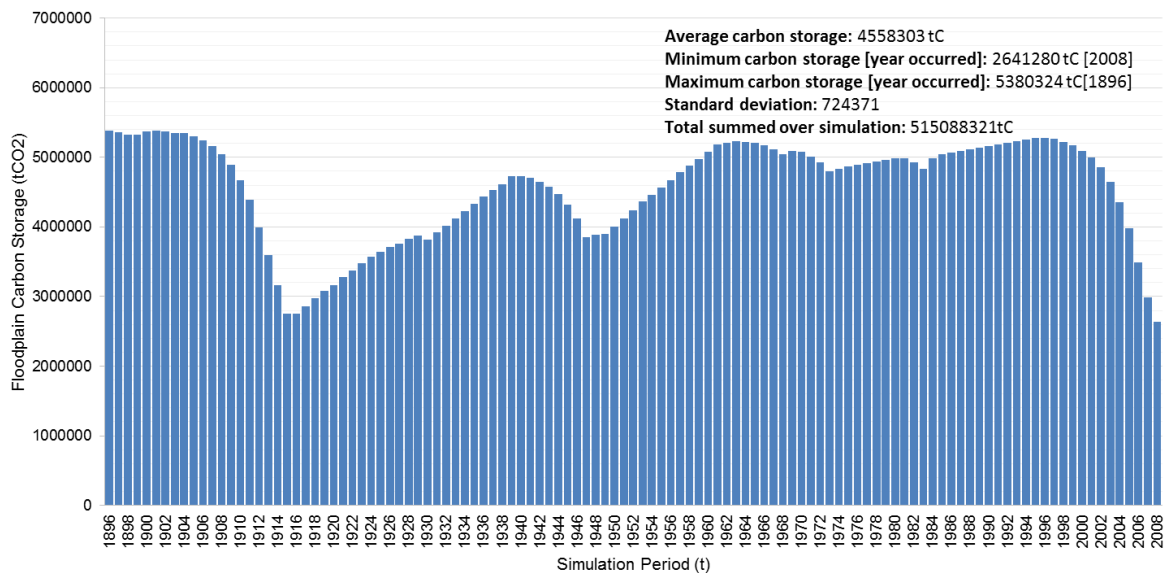
Source: Own Figure.

Figure C.7 Modelled Native Fish Habitat Suitability Scores for Native In-Channel Specialist (Murray Cod) Fish Under Baseline Conditions
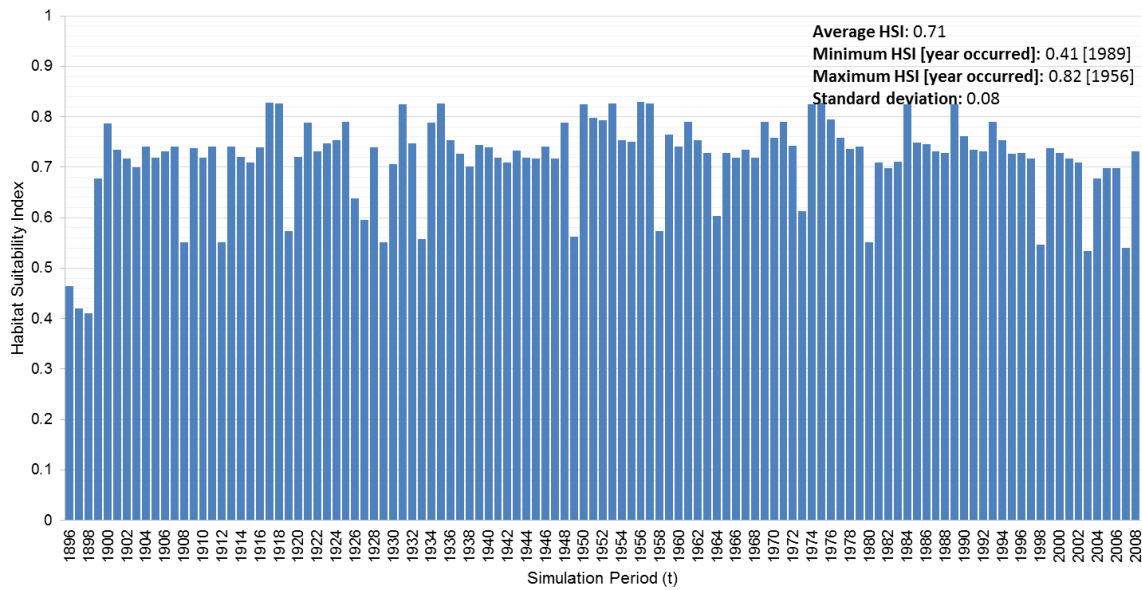
Source: Own Figure.

Figure C.8 Modelled Additional Annual Flow Required to Prevent the Formation of Favourable Blue Green Algae Conditions under Baseline Conditions
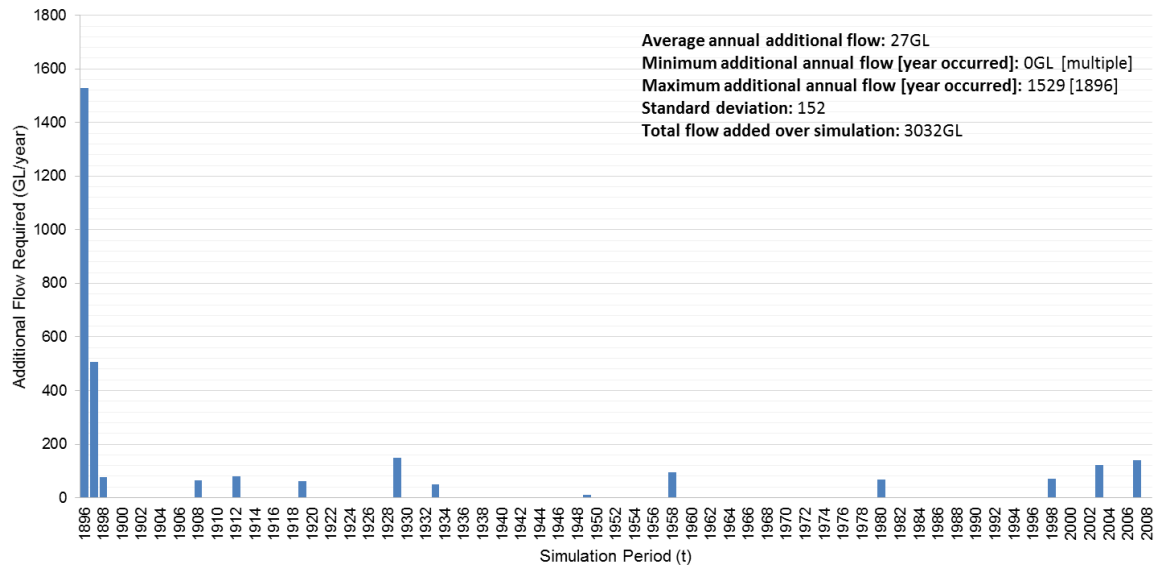
Source: Own Figure.

C.4.2   Ecosystem Service Supply for Environmental Reallocation Conditions

Unsurprisingly, as more water is initially reallocated from consumptive water-use to the environment there is an increase in the modelled ecological condition and supply of ecosystem services in the Lower Murrumbidgee catchment. The following section details the improvements in floodplain and in-channel ES supply for reallocation volumes from 5% to 50% reallocation from consumptive use to the environment.

C.4.2.1 Floodplain Ecosystem Services

Improvements in environmental water availability reduces the proportion of time River Red Gum environmental watering requirements are not met. This subsequently increases the supply and reduces the variability of carbon sequestration and erosion prevention ecosystem services. Figure C.9 demonstrates the reduction in variation for the supply of carbon storage ES as the proportion of water reallocated to the environment increases.

Figure C.9 Modelled Variation in River Red Gum Carbon Storage for Increasing Permanent Reallocations (CP=$23/tCO$_{2e}$)

Reallocation [S.D]	Mean millions
0%	$104.4 [16.6]
10%	$110.2 [16.9]
20%	$119.9 [5.7]
30%	$120.8 [4.4]
40%	$120.9 [4.4]
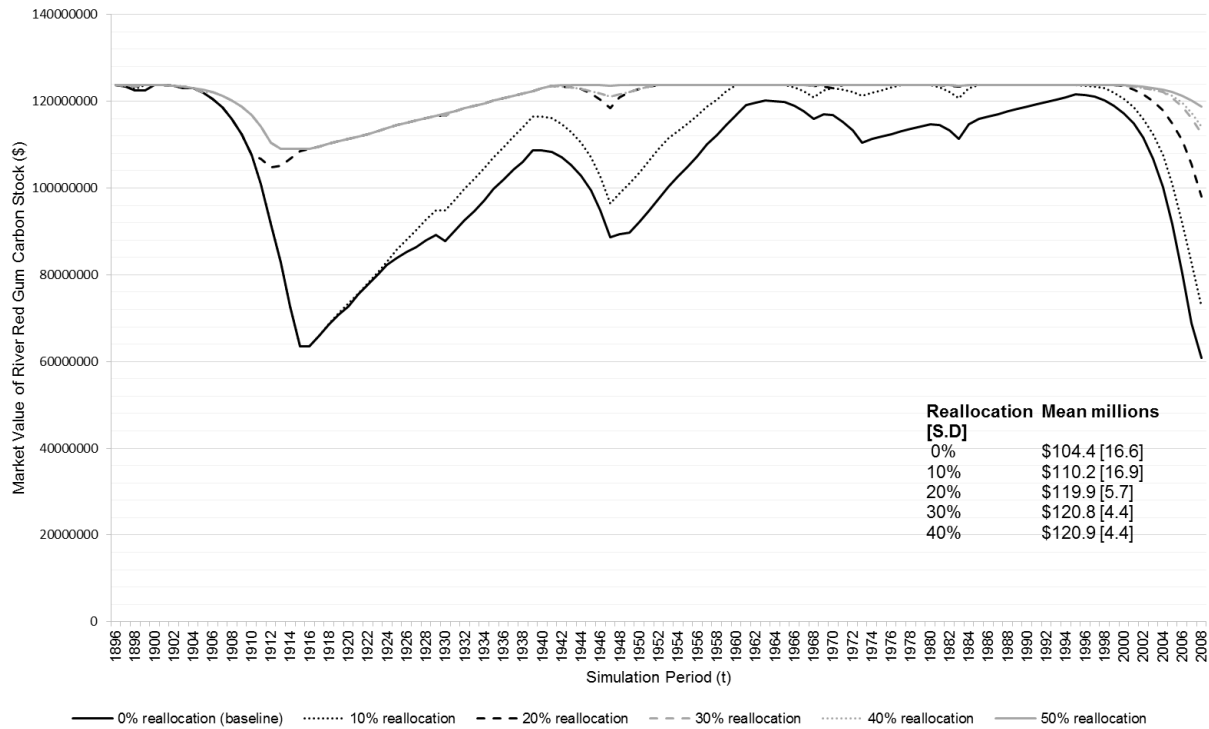
Source: Own Figure.

Relatively large volumes of reallocation are required to generate improvement in the flow of carbon storage and sequestration ES. The marginal increase in River Red Gum health and carbon storage ES supply increases for reallocation proportions between 0% - 20%. For a 20% reallocation proportion, the desired watering requirements of low elevation floodplain areas (flowband areas 1 to 4) are fully met. However, there is a considerable increase in watering requirements for flowband 6 (800GL), and high elevation flowbands 7 (1700GL) and 8 (2700GL). Reallocation volumes less than 50% make only a nominal difference in the improvement health of high elevation flowband River Red Gums which are inundated typically in 1 in 100 year flow events only for regulated river conditions. As a result of this biophysical discontinuity in the floodplain structure, the unit increases in reallocation proportion generates a smaller marginal improvement in carbon storage ES value for reallocation volumes greater than 20%. The supply of carbon storage ES increases at a decreasing rate for reallocation proportions greater than 20% (Figure C.10)

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Figure C.10 Modelled Marginal Increase in the Annual Flow of Carbon Storage Benefits for a Range of Carbon Prices
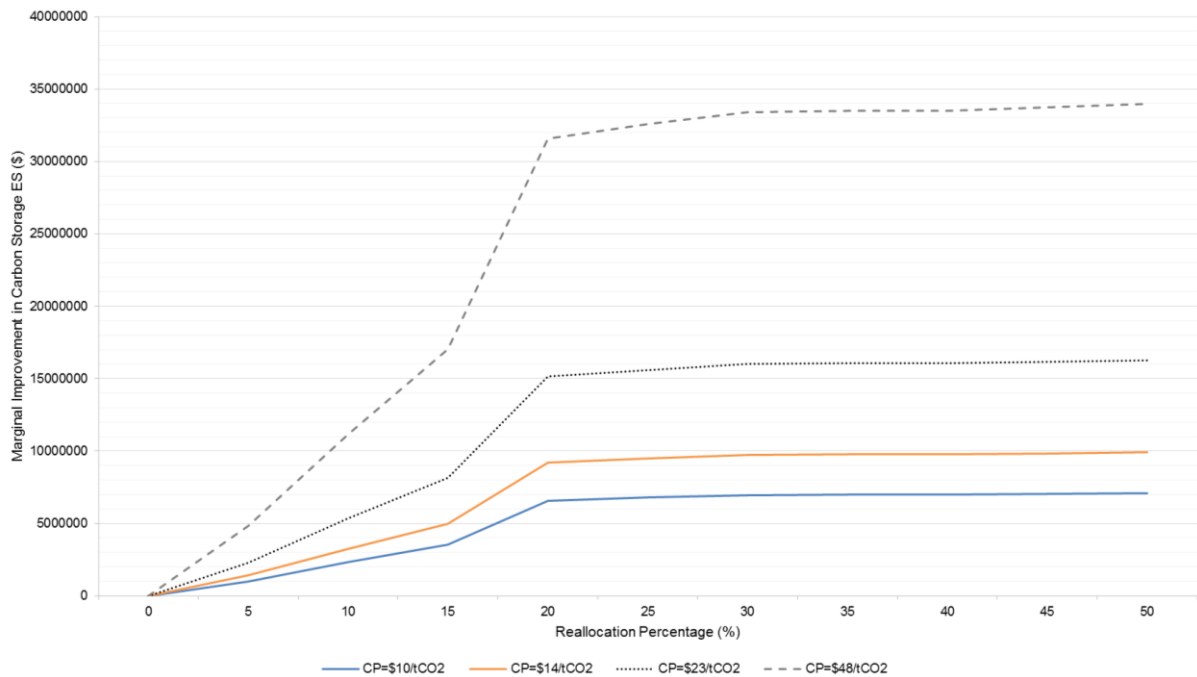
Source: Own Figure.

Importantly, however, the increase in reallocation volume generate a non-linear increase in marginal supply of floodplain ES values. The environmental watering requirements of the riparian corridor (flowband area 1) contributing to erosion prevention can be fully satisfied with relatively low reallocations to the environment (<10%). When the watering requirements of the riparian corridor are satisfied, modelled bank stability is increased and potential erodible area tends to zero, reducing the need for erosion prevention expenditure. For modelled reallocation proportions between 5% and 10% there are no periods within the simulation that would require man-made erosion control measures. There is therefore no modelled marginal improvement in the flow of erosion prevention ES benefit for unit additions of environmental water greater than 10% permanent reallocation, as shown in Figure C.11. However, the avoided cost benefits of erosion prevention ES are temporally distributed, such that benefits are pronounced in dry years and naught in wet years when critical thresholds are already met under baseline conditions.

Figure C.11 Modelled Marginal Increase in the Annual Flow of Erosion Prevention Benefits for a Range of Avoided Cost Estimates
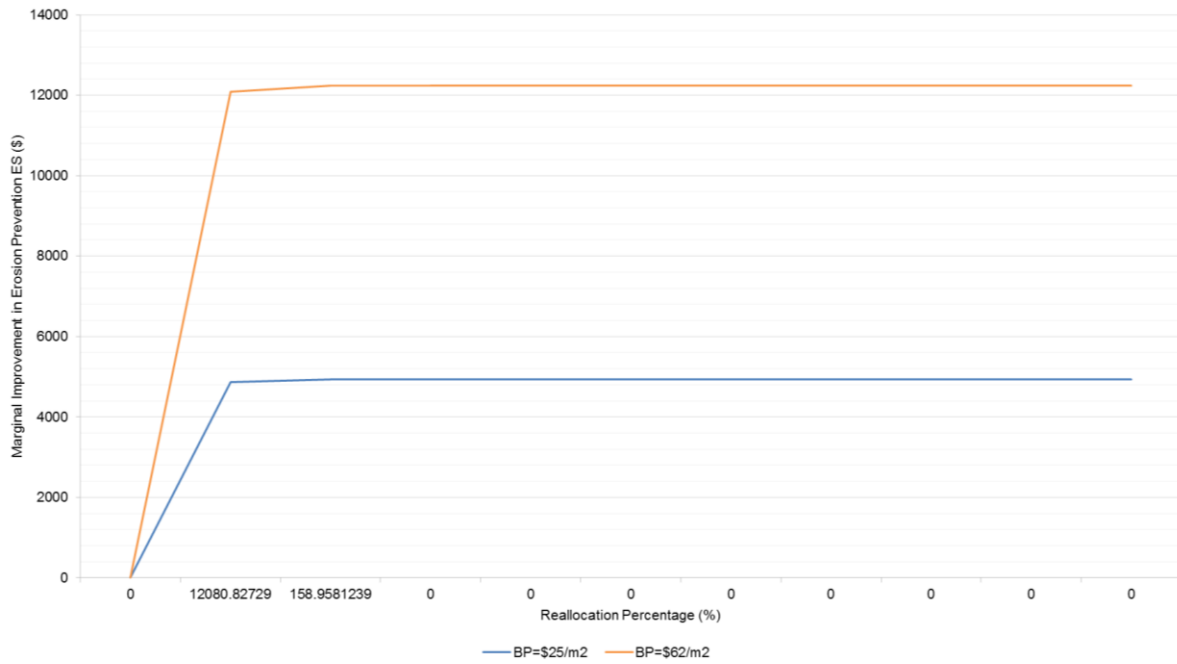
C.4.2.2 River Channel Ecosystem Services

All else constant, improvements in environmental water availability increased river velocity and the number of 'at risk' algae bloom years decrease, reducing the need for annual preventative expenditure. For a permanent reallocation >10%, the number of years requiring additional flushing flows tends to zero. This means that flow volumes do not drop below the critical threshold velocity in any year of the simulation for reallocation proportions greater than 10%. As such, the marginal benefit (avoided cost) of algae prevention is zero for additional units of reallocation greater than 10% of consumptive use. This is shown in Figure C.12.

Figure C.12 Modelled Marginal Increase in the Annual Flow of Blue Green Algae Prevention Benefits for the Average Water Allocation Price
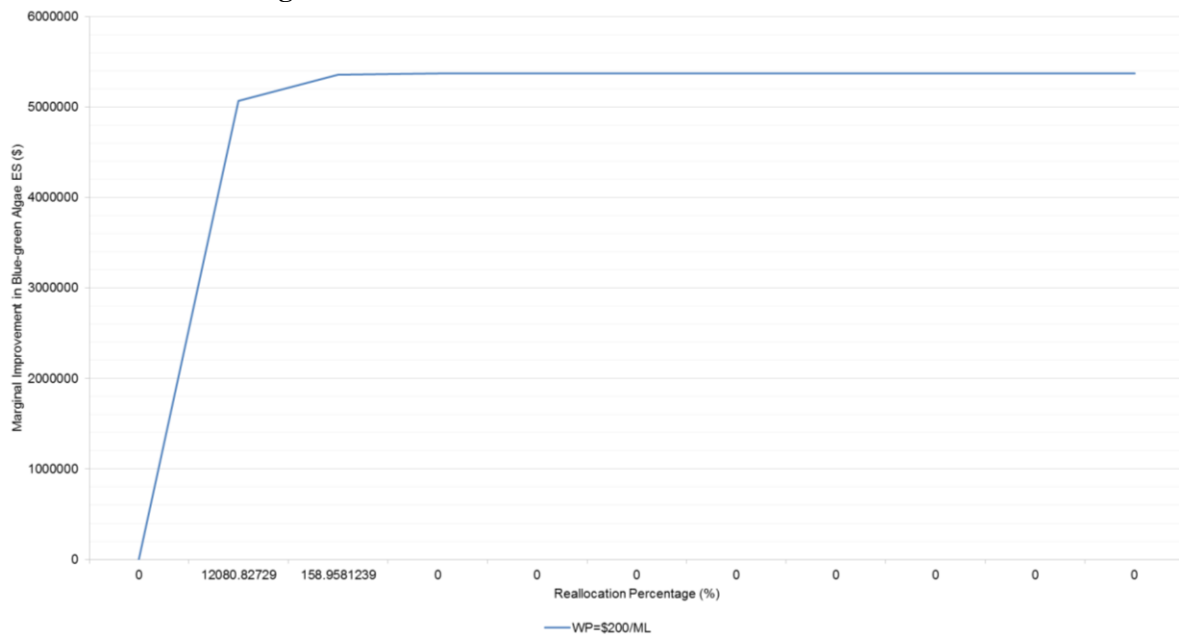
Improvements in environmental water availability improves flow dependent aspects of fish habitat indicators such as fish passage. As environmental water availability increases there is an initial near linear increase in the native fish habitat scores for small reallocation volumes (<10%). This is driven by improvements in modelled inundation of low-elevation floodplain areas and inundation duration which contributes to in-channel nutrient availability. Modelled fish habitat scores continue to increase at a decreasing rate for reallocation volumes greater than 10% until flow dependent habitat variables are fully satisfied. Improvements in fish habitat for higher volume reallocations is driven by improvements in fish passage during critical months. However, there are aspects of habitat condition which are not flow dependent (e.g. woody debris availability) and as a result modelled habitat condition will never be pristine (HSI=1) but rather tend asymptotically to a value less than 1 irrespective of increases in environmental water availability. As evident in Figure C.13, the marginal value of improved native fish habitat diminishes as it approaches maximum asymptotic HSI value that can be altered through changes to the flow regime[90]. Importantly, this behaviour is a result of the habitat suitability model rather than society's decreasing marginal WTP for increase in fish populations.

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[90] Other factors contributing to native fish habitat suitability, such as the presence of snags in the river, are not functions of annual river flows and are applied as constant reduction factors to the estimate of HSI. As such, the modelled HSI has asymptotic value is less than 1, irrespective of how much water is reallocated to the environment annually.

Figure C.13 Modelled Marginal Increase in the Annual Flow of Native Fish Habitat Benefit for a Range of Marginal Non-Market Value Estimates
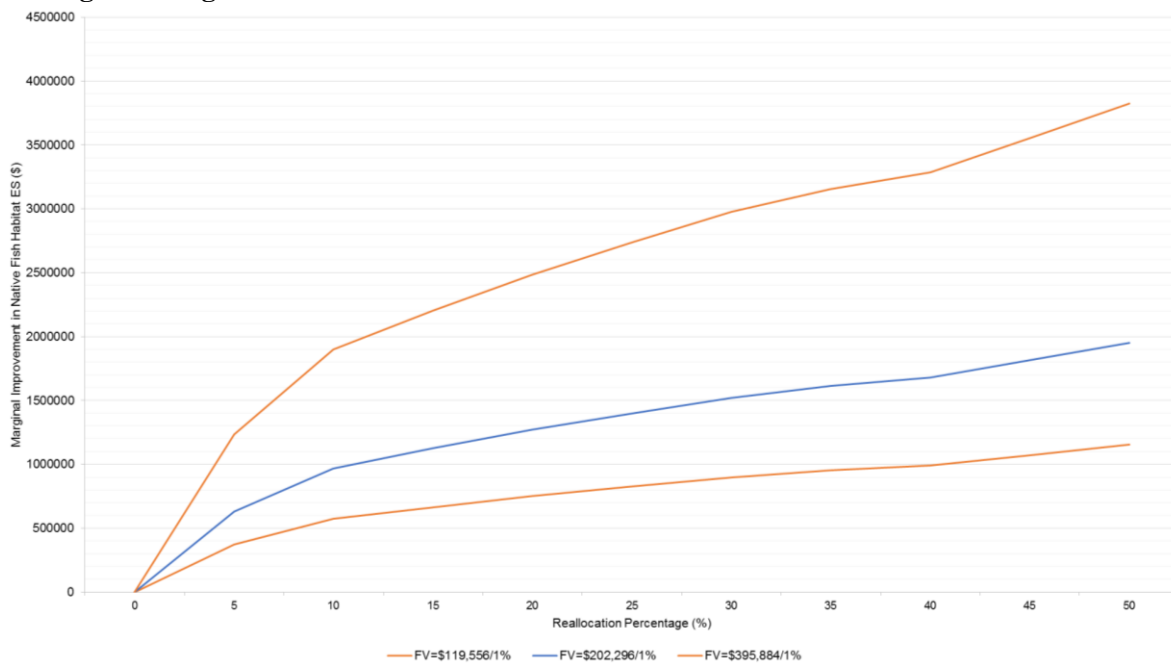
C.4.2.3 Total Ecosystem Service Benefits

The sum of the annual incremental benefits of ES supply provides an indication of the total ecosystem service value (column 6, Table C.7) generated by environmental reallocation. As the full suite of ES values are not accounted for this value is therefore not an estimate of total economic value (TEV), but rather an indication of the the potential magnitude and direction of change in ES supply that could be generated through reallocating water to the environment. The range in total ES value estimate is a result of the high and low estimates of individual ES supply value, as described in Section A.3.3.

Overall, it is found that relatively small volumes of reallocation (~10%) can reduce the need for algae and erosion preventative expenditure. The benefit of reallocation for these ES is therefore constant once environmental watering requirements have been met for their provision. As such the marginal increase in ES value for reallocation volumes greater than 10% reallocation is driven by improvements in carbon sequestration and native fish habitat ES values. This confirms findings by CSIRO (2012) who find that non-use values and carbon sequestration provide the largest ES benefits for reallocation proportion of 20%.

Table C.7 Improvements in Ecosystem Service Supply Value

Ecosystem Service	Carbon Sequestration	Erosion Prevention	Blue-green Algae Prevention	Native Fish Habitat	Total ES Value
Reallocation	($'000/year)				
5%	$1005 -$4826	$4.8 - $12.1	$5068	$373.1 - $631.3	$6451.6 - $10537
10%	$2352 - 11160	$4.9 - $12.2	$5356	$573.4 - $970.2	$8260.1 - $17499.7
20%	$6580 - 31587	$4.9 - $12.2	$5356	$751.1 - $1271	$12704.9 - $38239.1
30%	$6959 -$33407	$4.9 - $12.2	$5356	$898.7 - $1520.6	$13231.5 - $40308.0
40%	$6979 - 33503	$4.9 - $12.2	$5356	$991.7 - $1678.0	$13344.5 – $40561.5
50%	$7076 - 33966	$4.9 - $12.2	$5356	$1154.4 - $1953.3	$13603 – $41300.4

This is shown graphically in Figure C.14 which is an empirical version of the conceptual model hypothesised in Figure C.1. As can be seen, there is a non-proportional non-linear relationship between environmental water availability and ES supply. Instead, the incremental improvement in ES supply is contingent upon the health of the underlying biophysical system and the progressive satisfaction of environmental watering requirements as reallocation volume increases. This provides an alternative narrative to the existing policy dialogue which places a large focus on the volume of environmental water as an indicator of environmental benefit. As is evident by the range of values in Table C.7, the value of ES improvements is contingent upon the value society assigns to the environment. For example, if individuals place no value on the existence of native fish species the shape of the graph in Table C.7 would be considerably more horizontal and incremental marginal ES values would be diminished.

Figure C.14 Modelled Marginal Increase in the Summed Annual Flow of Ecosystem Services in the Lower Murrumbidgee (Carbon Storage, Blue Green Algae Prevention, Erosion Prevention, Native Fish Habitat)
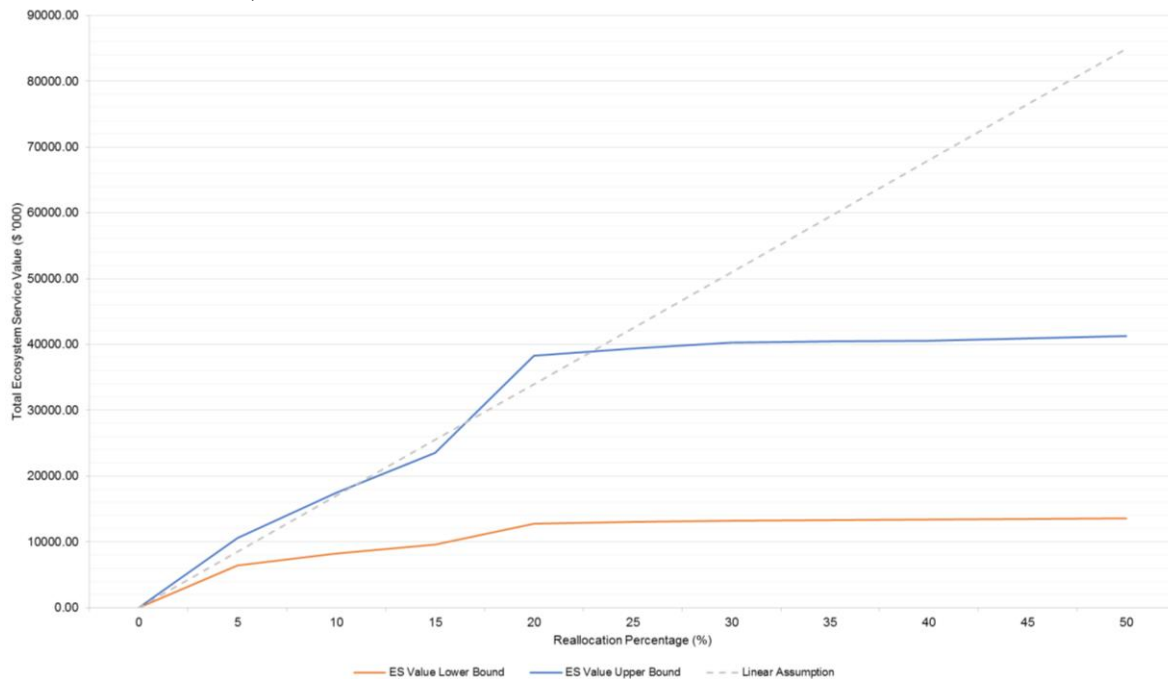
Source: Own Figure.

C.5 Discussion and Policy Implications

The public policy debate about reallocating water in the MDB has placed significant emphasis on the additional volume of environmental water availability achieved through reallocation (Crase et al., 2011). However, this policy approach does not fully account for the non-linear relationship between the volume of water reallocated to the environment and the provision of ecosystem services. In this chapter it has been shown that by integrating biophysical responses and economic values in a hydro-economic model it is possible to estimate the marginal improvement in ES supply for incremental increases in environmental water availability to generate a more nuanced understanding of the source, magnitude and direction of benefits. The non-linear response of ecosystem service supply has a number of policy implications for the design and discussion of reallocation policy.

First, improvements in ES supply are not necessarily proportional to the unit increase in environmental water availability owing to ecological thresholds and non-linearities inherent in the biophysical structure from which ecosystem services are derived. For the case study presented there is a near linear relationship for reallocation percentages between 0%-20%. This is due to the fact that under bassline scenarios the Murrumbidgee is in 'very poor' condition and therefore ecological condition benefits considerably from improvements in environmental water availability. However, beyond this reallocation the marginal improvements in ES supply diminish for reallocation volumes greater than 20%. Models which assume a constant and linear relationship between increased flows and ES supply without considering the nuanced nature of the relationship between water availability and

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environmental condition may miss-represent the value of environmental reallocation. This may lead to a distorted understanding of costs and benefits and could result in less cost-effective reallocation policy. When designing reallocation programs additional focus should be awarded to setting targets in line with ecological response and ES supply generation rather than the volume of water to be reallocated. Based on the modelling results the *Basin Plan*, which sets a reallocation target of 20% reduction in consumptive use, is likely to have considerable positive impacts on ES supply in the Murrumbidgee catchment. ES value improvements are predominately derived from non-market habitat value and carbon sequestration, confirming exiting research by CSIRO (2012) and Bark et al. (2016) which reach the same conclusion.

Second, the marginal improvements in ES supply generated by improved environmental water availability are temporally distributed. For example, the application of an additional one unit of environmental water to a floodplain in an ecologically appropriate sequence will provide a different unit increase in floodplain ES supply than the same volume of water delivered five years past the desired return period. This is due to the temporal dimension of desired environmental water requirements and the effect of biophysical thresholds on ES supply. Likewise, the targeted addition of environmental water to provide fish passage during native fish migration periods (September-December for Murray Cod) (InsideMFAT, 2004) will result in a higher marginal improvement in fish populations than the same volume of water delivered outside of these months. This provides evidence that there would be benefit in expanding the policy dialogue to better account for the temporal aspects of reallocation and environmental watering demands.

Lastly, the supply profiles for individual ecosystem services are vastly different depending on the watering requirements of the biophysical structure. For example, if the reallocation objective was to minimise years at risk of bloom formation, this would be likely achieved with 10% permanent reallocation strategy. Conversely, if the objective was to restore native fish habitat condition this would be achieved with significantly higher reallocation volumes. This implies that the choice of indicators when modelling or measuring the impacts of reallocation is of crucial importance. Consideration of additional ES supply may further improve representation of environmental benefits and improve accuracy of estimates of reallocation. While the choice of indicators will never completely describe the changes in a complex system (Functowicz, 1999), the variation in supply profiles of the ES considered suggests that the selection of indicators should be undertaken with care to best capture the dynamics of both floodplain and river channels and avoid a distorted picture which could be obtained through the use of a single indicator.

C.6 Conclusion

The results of a highly integrated hydro-economic model incorporating ecosystem service sub-models for carbon sequestration, erosion prevention, blue-green algae prevention and native fish habitat have been presented. In the Lower Murrumbidgee case study presented, it is found relatively small reallocation volumes (<10% irrigation supply) can improve the supply of Blue-green algae and erosion prevention, as well as reduce the variability of carbon storage and native fish habitat ES supply. A 20% reallocation in the Lower Murrumbidgee, consistent with the Basin Plan objectives, improves fish habitat condition and overbank flooding regime and is likely to generate considerable ES benefits. However, the aggregate marginal improvement in modelled ES supply is not necessarily proportional to increases in reallocation proportions. Rather, the each ecosystem service is dependent upon the satisfaction of the environmental watering requirements of biophysical structure which process the service. This result suggests that the marginal value of environmental water is itself non-linear; a trait which should be more widely considered in reallocation policy design and discussion. To date, the public policy debate has placed a large emphasis on the volume of water reallocated to the environment and little discussion has been awarded to the consideration of the marginal ES improvements gained by increased reallocation volumes. Further, it is shown that the choice of ES indicators is critical as a measure to inform reallocation objectives and success. Overall, it is suggested that progressing the both the academic and policy discussion beyond the reallocation volumes to better account for the relationship between watering volumes and ES supply dynamics may inform a more cost-effective reallocation approach.

APPENDIX D: Monthly Flow Analysis

The mean long-term monthly flow is determined using daily hydrological data for *without development*, *baseline* and *Basin Plan* conditions at Maude Weir, which is an upstream management point for the Lowbidgee floodplain. The hydrological data set spans 113 years and was provided by CSIRO. For each month the mean flow is calculated as the sum of flow per day divided by the number of days in the month. This is repeated for all twelve months to determine the monthly average for each month. The long-term monthly flow is determined by summing the monthly flow volume for each month and dividing by the number of years in the hydrological data set (113) to determine the long-term mean over the period of record. It is assumed that the flow release pattern at Maude weir is comparable to the flow release pattern at Balranald management point, which is slightly downstream. It is not necessary to normalise by catchment size because Maude and Balranald are contained within the same Lowbidgee sub-catchment and have the comparable catchment properties. The long-term monthly flow mean is indicative of the average intra-annual flow release pattern. The calculated long-term mean monthly flow release pattern is shown in Figure D.1. Descriptive statistics for the average annual monthly flow pattern are given in Table D.1.

Figure D.1 Mean monthly streamflow (ML), Murrumbidgee River at Maude Weir (1985-2009) (calculated)
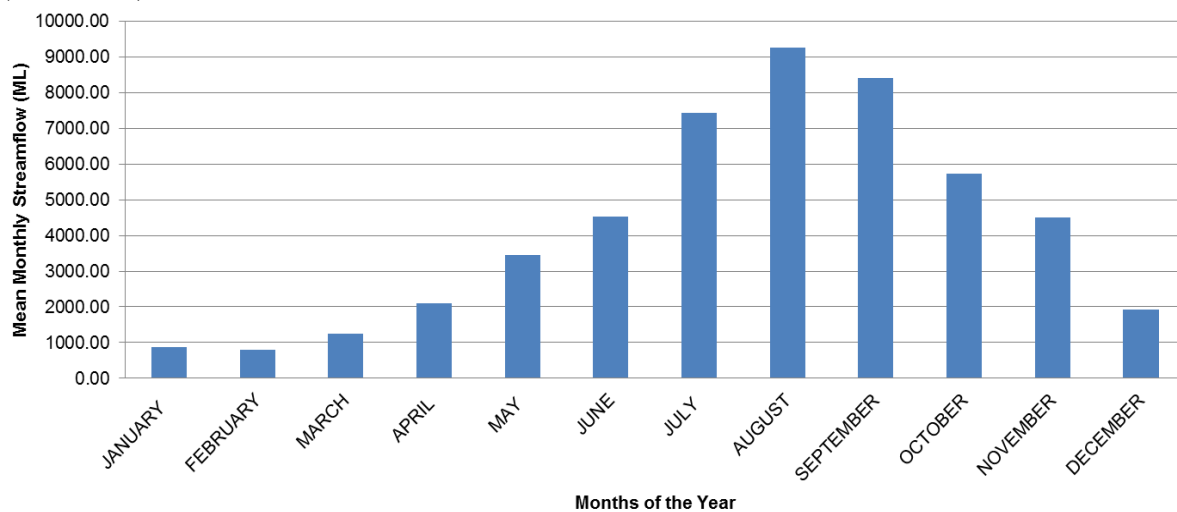
Source: Own Figure.

Table D.1 Descriptive Statistics of the Long-term Mean Monthly Flow Release Pattern in the Lower Murrumbidgee

MONTH	MEAN (ML)	MAX (ML)	MIN (ML)	STDEV
JANUARY	862.13	4298.06	441.46	793.18
FEBRUARY	802.28	17952.38	428.81	1659.53
MARCH	1252.21	10486.32	705.75	1368.16
APRIL	2090.99	57011.93	764.42	6153.36
MAY	3453.63	33261.90	349.88	5545.59
JUNE	4533.24	38602.64	415.36	6546.28
JULY	7428.75	70305.21	752.01	10186.99
AUGUST	9266.08	52631.18	1111.82	8795.04
SEPTEMBER	8415.15	47351.62	1615.12	7408.39
OCTOBER	5726.98	30109.18	1450.63	6523.34
NOVEMBER	4506.88	31624.82	1078.71	6968.20
DECEMBER	1935.52	12083.24	538.48	2582.86

Source: Own Table.

For ecosystem services which are dependent on velocity related hydrological indicators (e.g. Blue-green algae prevention), it is necessary to express the annual flow outputs in the HEM as an inter-annual distribution. To achieve this, the long-term mean flow were calculated as a ratio of the long-term average annual flow, which is found to be approximately 4201.1ML/year. Results are presented in Table D.3. The graphical representation of Table D.3 is shown in Figure D.2

Table D.2 Mean Monthly Flow as a Ratio and Percent of Mean Annual Flow

MONTH	MONTHLY MEAN (mmq) (ML)	ANNUAL MEAN (amq) (ML)	MMQ as percentage of AMQ	MMQ as PERCENT RATIO OF AMQ
JANUARY	862.13	4201.1	20.52	0.21
FEBRUARY	802.28	4201.1	19.10	0.19
MARCH	1252.21	4201.1	29.81	0.30
APRIL	2090.99	4201.1	49.77	0.50
MAY	3453.63	4201.1	82.21	0.82
JUNE	4533.24	4201.1	107.91	1.08
JULY	7428.75	4201.1	176.83	1.77
AUGUST	9266.08	4201.1	220.56	2.21
SEPTEMBER	8415.15	4201.1	200.31	2.00
OCTOBER	5726.98	4201.1	136.32	1.36
NOVEMBER	4506.88	4201.1	107.28	1.07
DECEMBER	1935.52	4201.1	46.07	0.46

Source: Own Table.

Figure D.2 Average Distribution of Monthly Streamflow (ML), Murrumbidgee River at Maude Weir (1985-2009).
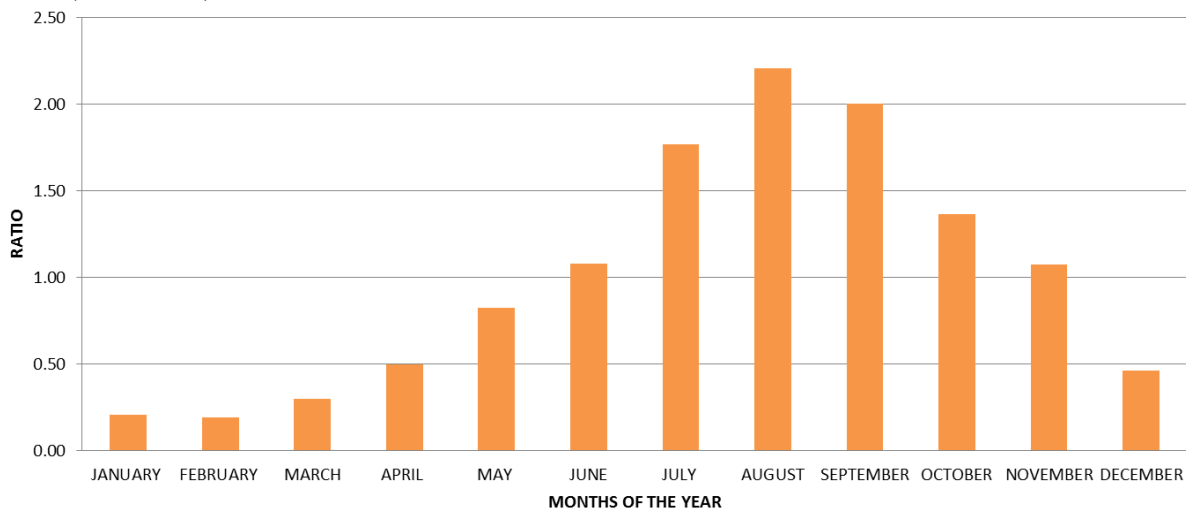
Source: Own Figure.

For each year, the average monthly flow is calculated by multiplying the long-term average ratio of *mean monthly flow:mean annual flow* by the annual output to provide an estimate of monthly flow volume. This is conducted within the HEM for each period of the hydrological time-series to develop an estimate of the intra-annual flow release patterns over the 113 year simulation. An example is shown Figure D.3 which driest years on record (2007) and that has a notably different intra-annual environmental water supply profiles compared to the average distribution shown in Figure D.2.

Figure D.3 Mean Monthly Streamflow (ML), Murrumbidgee River at Maude Weir (2007).
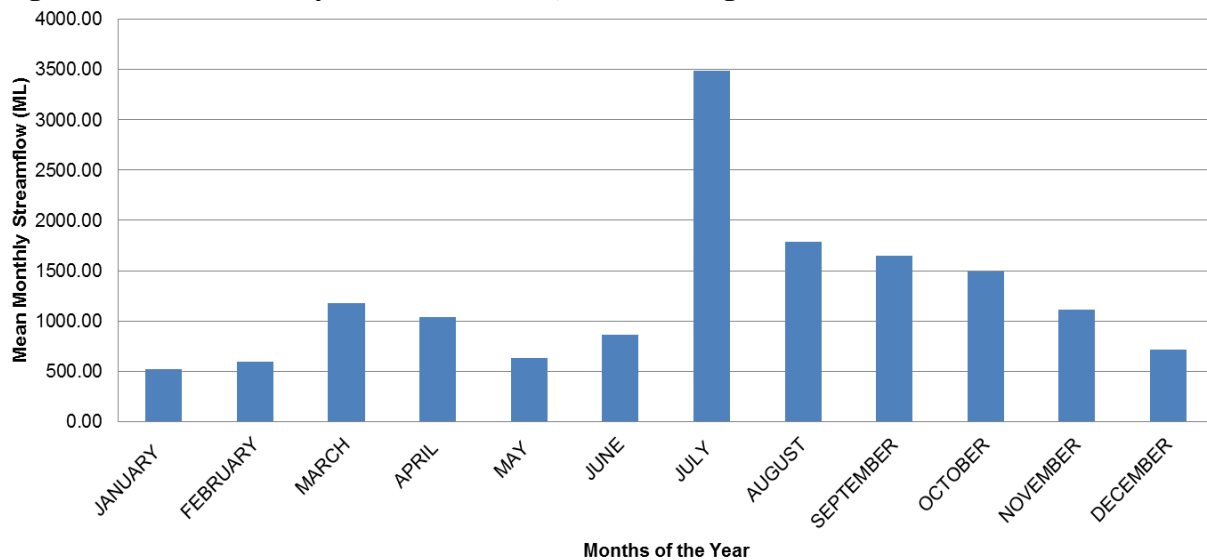
Source: Own Figure.

APPENDIX E: Supplementary Material from Environmental Water Stakeholder Interviews

E.1 Ethics Approval

THE UNIVERSITY
of ADELAIDE

RESEARCH BRANCH
OFFICE OF RESEARCH ETHICS, COMPLIANCE
AND INTEGRITY

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CRICOS Provider Number 00123M

1 July 2015

Associate Professor W Umberger
School: Global Food Studies

Dear Associate Professor Umberger

ETHICS APPROVAL No: H-2015-140

PROJECT TITLE: **Environmental Water Stakeholder Interviews**

The ethics application for the above project has been reviewed by the Low Risk Human Research Ethics Review Group (Faculty of Arts and Faculty of the Professions) and is deemed to meet the requirements of the *National Statement on Ethical Conduct in Human Research (2007)* involving no more than low risk for research participants. You are authorised to commence your research on **01 Jul 2015**.

Ethics approval is granted for three years and is subject to satisfactory annual reporting. The form titled *Annual Report on Project Status* is to be used when reporting annual progress and project completion and can be downloaded at http://www.adelaide.edu.au/ethics/human/guidelines/reporting. Prior to expiry, ethics approval may be extended for a further period.

Participants in the study are to be given a copy of the Information Sheet and the signed Consent Form to retain. It is also a condition of approval that you **immediately report** anything which might warrant review of ethical approval including:

- serious or unexpected adverse effects on participants,
- previously unforeseen events which might affect continued ethical acceptability of the project,
- proposed changes to the protocol; and
- the project is discontinued before the expected date of completion.

Please refer to the following ethics approval document for any additional conditions that may apply to this project.

Yours sincerely

PROFESSOR RACHEL A. ANKENY   PROFESSOR PAUL BABIE
Co-Convenor   Co-Convenor
Low Risk Human Research Ethics Review Group   Low Risk Human Research Ethics Review Group
(Faculty of Arts and Faculty of the Professions) (Faculty of Arts and Faculty of the Professions)

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E.2 Participant Consent Form

PARTICIPANT CONSENT FORM

1. I have read the attached Information Sheet and agree to take part in the following research project:

Title:	Environmental Water Stakeholder Interviews
Ethics Approval Number:	H-2015-140

2. I have had the project, so far as it affects me, fully explained to my satisfaction by the research worker. My consent is given freely.

3. Although I understand the purpose of the research project it has also been explained that involvement may not be of any benefit to me.

4. I have been informed that, while information gained during the study may be published, I will not be identified and my personal results will not be divulged.

5. I understand that I am free to withdraw from the project at any time throughout the study without any repercussions now or in the future.

6. I agree to the interview being audio-recorded during the interview.   Yes ☐   No ☐

7. I agree to be contacted at a later date to complete a short online survey.   Yes ☐   No ☐

8. I am aware that I should keep a copy of this Consent Form, when completed, and the attached Information Sheet.

Participant to complete:

Name: _____ Signature: _____ Date: _____

Researcher/Witness to complete:

I have described the nature of the research to _____
                                                                    (print name of participant)

and in my opinion she/he understood the explanation.

Signature: _____ Position: _____ Date: _____

E.3   Participant Information Sheet

THE UNIVERSITY
of ADELAIDE

PARTICIPANT INFORMATION SHEET

PROJECT TITLE: Environmental Water Stakeholder Interviews
HUMAN RESEARCH ETHICS COMMITTEE APPROVAL NUMBER: H-2015-140
PRINCIPAL INVESTIGATOR: Associate Professor Sarah Wheeler
STUDENT RESEARCHER: Ms Claire Settre
STUDENT'S DEGREE: PhD

Dear Participant,

You are invited to participate in the research project described below.

What is the project about?
This project is about water rights transfers within the agricultural community and from the agricultural community to third parties. It particularly focuses on the transfer of water rights to government departments, non-profit organizations, and water trusts, to be used for habitat improvement and environmental restoration. The project aims to gain an understanding of the agricultural community's attitudes towards water transfers, the role of non-profit organizations, tribal groups, and government in acquiring water rights for habitat improvement, and collaborative policy innovations used to manage shared water resources. The project aims to achieve an in depth understanding of the Western US water landscape, focusing on water transactions in the Colorado Basin, in order to identify key lessons relevant to policy development in Australian water resource management.

Who is undertaking the project?
This project is being conducted by Ms Claire Settre. This research will form the basis for the award of PhD at the University of Adelaide under the supervision of Associate Professor Sarah Wheeler.

Why am I being invited to participate?
You have been invited to participate in this project based on your professional or personal experience, current involvement, and/or ability to provide relevant policy information regarding water trade, water management, and/or environmental water management in the Western US. If you feel you are not able to or do not want to discuss this information, then you may choose not to participate.

What will I be asked to do?
You will be asked to participate in a semi-structured, face to face interview. If you consent, the interview will be audio-recorded.

How much time will the interview take and when will it occur?
The interview will take less than 60 minutes. The interview will be scheduled at a time and place of your convenience before the 15th November, 2015.

Are there any risks associated with participating in this project?
The risks associated with participating in the interview are minimal. These risks may include discomfort derived from being asked a range of questions about your professional experience and personal opinions. If you wish to end/change the interview, please let the interviewer know and an alternative arrangement will be made.

What are the benefits of the research project?

There are no immediate benefits from participating in this project. Participants may value being able to contribute their opinion and share their experience in their field.

Can I withdraw from the project?
Participation in this project is completely voluntary. If you agree to participate, you can withdraw from the project at any time with no repercussions now or in the future. If you choose to withdraw, your data will be withdrawn from the project up until the submission of the thesis or subsequent research articles. Beyond this time it will not be possible to withdraw your data.

What will happen to my information?
The information collected from you will be used for research purposes only. Data that could be used to identify you will be coded (made anonymous) and re-identifiable only to the researcher. Any information you provide will not be considered representative of the organization(s) you are affiliated with. Data from the interviews will be described thematically (for example: "a key theme emerging from the interviews was…."). When it is necessary to quote interview data, anonymous labels will be used (for example: "Stakeholder #7"). Results will be published in journal articles, conference papers and a PhD thesis. All project information will be confidentially stored on the University of Adelaide's password protected electronic database for a period of five years.

Who do I contact if I have questions about the project?
Claire Settre

Graduate Student, Global Food Studies, the University of Adelaide, Australia

Phone: 951-907-7270   Email: claire.settre@adelaide.edu.au

Associate Professor Sarah Wheeler

Global Food Studies, the University of Adelaide, Australia

Phone: +61 8 830 20698   Email: sarah.wheeler@adelaide.edu.au

What if I have a complaint or any concerns?
The project has been approved by the Human Research Ethics Committee at the University of Adelaide (approval number H-2015-140). If you have questions or problems associated with the practical aspects of your participation in the project, or wish to raise a concern or complaint about the project, then you should consult the Principal Investigator. Contact the Human Research Ethics Committee's Secretariat on phone +61 8 8313 6028 or by email to hrec@adelaide.edu.au if you wish to speak with an independent person regarding concerns or a complaint, the University's policy on research involving human participants, or your rights as a participant. Any complaint or concern will be treated in confidence and fully investigated. You will be informed of the outcome.

If I want to participate, what do I do?
If you would like to participate, please indicate to the researcher via return email your intention to participate in the interview. You will then be contacted by phone to schedule an interview time and place at your convenience.

Yours sincerely,

Ms Claire Settre
Associate Professor Sarah Wheeler

E.4   Sample Interview Talking Points for Semi-Structured Stakeholder Interviews

BACKGROUND

- In what regions does the Freshwater Trust work to acquire water rights?
- The Oregon Water Code seems to have many more provisions for environmental uses of water compared to California, for example, could you give me a bit of a background of why that occurred?
- How big of a part does purchasing water rights play in your conservation strategies?
- In Oregon does the state government (for example; The Water Resource Department) engage with purchasing water rights for the environment?

WATER TRUSTS

- The Freshwater Trust has a unique approach to engaging with irrigators, could you elaborate?
- Could you tell me a bit about how the Oregon Water Trust emerged into the Freshwater Trust?
- In other parts of the Western US, water trusts exist in conceptual form more than in practice. In Oregon, water trusts are a lot more popular. Could you discuss why this is?
    - PROMPT: enabling social/institutional conditions?

WATER RIGHTS ACQUISITIONS STRATEGY

- How does the Freshwater Trust identify water rights for acquisition?
- When you acquire water, from whom do you predominately acquire it?
    - PROMPT: do you engage with irrigation districts at the organizational level, or with individual rights holders, or urban municipalities?
- Typically what kind of water rights are you acquiring?
    - Instream leasing, time limited transfers, long-term, perpetual?
    - Do you target senior rights over junior rights?
    - Is this by design or by virtue of what irrigation districts are willing to sell to you?
- For acquiring water rights for conservation, do you think one type of right is more advantageous to have than anther? (i.e.: short term lease vs. long-term lease etc.)
    - What is the benefit of these types of rights?
- If acquiring short term leases, do you think this impacts your ability to manage water long-term?
    - How do you overcome this?
- When environmental demand is low, do you lease water back to rights holders?
    - Why/why not? (i.e.: is it legally possible?)
- Do financial constraints play a role in NGO ability to purchase environmental water?
    - Has this ever been the case for The Freshwater Trust?
    - When supply is scarce (drought) does your ability to purchase water decrease?
- If you don't mind me asking, how does The Freshwater Trust fund its water rights acquisitions?
    - Would you say this is a typical approach of many NGOs?
    - Could you give me an approximate amount of water that The Freshwater Trust holds in Oregon?

ENVIRONMENTAL WATER DELIVERY STRATEGY

- How do you identify environmental watering priorities?
    - PROMPT: Optimise ecological benefit? Optimise ecosystem service benefit? Fulfil legal contract? Optimise spiritual/indigenous use? Most benefit for dollars spent?
- Do your habitat/watering priorities dictate what type of water rights you try to buy?
    - If not, what motivates why type of water rights you buy?
- Once you have acquired rights, who manages them?

- o PROMPT: do you manage them yourself at The Freshwater Trust?
- o PROMPT: do you transfer them to a government body or state to be used in public trust?
- Private trusts and NGOs work towards their own objectives, could you discuss how you:
  - o Co-ordinate with other NGO's? (acquisition and delivery)
  - o Co-ordinate with state agencies? (acquisition and delivery)
  - o Co-ordinate with federal agencies? (acquisition and delivery)
- What is your strategy for measuring the effectiveness of your environmental watering/restoration efforts?
- In general is the amount of water you purchase able to meet your restoration objectives?

IRRIGATOR AND WATER RIGHTS HOLDER ENGAGEMENT

- Do you typically engage with irrigation districts at the organizational level, or with individual rights holders?
- In your experience, do the irrigators you engage with tend to trade water between themselves?
- What would you say is the general attitude of agriculturalists in your county to transferring water to the environment?
  - o Is there a different attitude if the water is sold to urban users?
- Does The Freshwater Trust maintain ongoing engagement with irrigators who sell water rights?
- The Oregon System has the clause 'use it or lose it'. Is this a hurdle to engaging irrigators?
  - o PROMPT: legal protections for instream transfers
- Have you noticed any common characteristics of rights holders that engage with transferring water to the environment?
- In your experience, do rights holders or irrigation districts engage more with private water trusts (i.e. The Freshwater Trust), state based agencies, or federal bodies when transferring water to the environment?
  - o Could you elaborate on why you think that is?
- Do you think irrigators or irrigation districts have a preference for the type of right that they transfer?
  - o PROMPT: Short term, long-term, permanent, point of diversion etc?
- Do you think irrigators in the regions you work would freely engage with the concept of mature market for water where they can buy and sell between each-other as they see fit?

GENERAL TALKING POINTS

- Have you seen an increase or a decrease in irrigators or districts willingness to sell their water or engage with the Freshwater Trust since drought conditions have begun?
- Some academic publications suggest that environmental water trade in the Western US will slow down. What are your thoughts?
- What primary constraints do you see restricting the emergence of a mature market for environmental water, where irrigators or districts can sell rights at will?

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