Identifying the water-related economic values of the Murray-Darling Basin and rating the quality of water economic studies

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Glossary

Adaptation	The response to major changes in the environment (e.g., global warming) and/or political and economic shocks. Adaptation is often imposed on individuals and societies by external undesirable changes.
Adoption (in agriculture)	A change in practice or technology.
Annual crops	Crops that go through their entire lifecycle in one growing season (e.g., cotton, rice, cereal).
Basin Plan	A high-level framework that sets standards (see sustainable diversion limits) for the management of Murray-Darling Basin water resources, balancing social, environmental, and economic outcomes.
Barmah Choke	A natural geological formation near the Victorian town Barmah. It restricts the daily flow of the River Murray to ~ 7,000 ML/day.
Broadacre	Broadacre cropping (a term used mainly in Australia) describes large-scale agricultural production of grains, oilseeds and other crops (e.g., wheat, barley, sorghum).
Carry-over	Arrangements which allow water entitlement holders to hold water in storages (water allocations not taken in a water accounting period) so that it is available in subsequent years.
Catchment (river valley)	An area determined by topographic features, within which rainfall contributes to run-off at a particular point.
Commonwealth Environmental Water Holder (CEWH)	An independent statutory office established by the *Water Act 2007* and responsible for making decisions relating to the management of the Commonwealth environmental water aiming to maximise environmental outcomes across the Murray-Darling Basin.
Consumptive water use	The use of water for private benefit (e.g., irrigation, industry, urban, and stock and domestic uses).
Council of Australian Governments (COAG)	Is the peak intergovernmental forum driving and implementing reforms in Australia (members are the Prime Minister, State and Territory Premiers and Chief Ministers and the President of the Australian Local Government Association).
Environmental asset	According to the Basin Plan, include water-dependent ecosystems, ecosystem services and sites with ecological significance.
Environmental water	According to the Basin Plan, comprises water provided to wetlands, floodplains or rivers, to achieve a desired outcome, including benefits to ecosystem functions, biodiversity, water quality and water resource health.
Farming water season	Describes a 12-month period from July 1 to 30 June (similar to the financial year in Australia).
Groundwater	The supply of freshwater found beneath the earth's surface (typically in aquifers).

High security water entitlement	Provide a highly reliable water supply (usually full allocation 90-95 years out of 100) with not much variation between the years (except during extreme drought).
IVTs	Inter-valley Trade restrictions. Restriction to the volume of water allocation that can be transferred/traded between catchments in a given season.
Irrigation Infrastructure Operators (IIO)	An entity that operates water service infrastructure to deliver water for the primary purpose of irrigation.
Long term average annual yield factor (LTAAY)	Expected long-term average annual yield from a water entitlement over a 100-year period.
Long-term diversion limit equivalent factor (LTDLE)	Proportion of long-term average annual water use per unit of entitlement. Also known as “Cap factors”, they were initially adopted in 2011, but have been updated in 2018/19 to consider recent water use information, climate patterns over the last 100 years, water trade patterns, and modelled Base Line Diversion Limits for each catchment.
Low/general security water entitlement	Provide a variable/uncertain water supply. General security provides LTAAY between 42-81%, and low security provides LTAAY between 24-35% in the Murray-Darling Basin.
MDBA	Murray-Darling Basin Authority, established by the Water Act 2007, formerly Murray-Darling Basin Commission
National Water Initiative (NWI)	The national blueprint for water reform, agreed in 2004 by the Council of Australian Governments (COAG), to increase the efficiency of Australia's water use, leading to greater certainty for investment and productivity, for rural and urban communities and for the environment.
Over-allocation	The total volume of water able to be extracted by the holders of water (access) entitlements at a given time exceeds the environmentally sustainable level of take for a water resource.
Regulated river system	Rivers regulated by major water infrastructure, such as dams, to supply water for varies uses.
Reliability	The frequency with which water allocated under a water (access) entitlement is able to be supplied in full.
River Murray Operations budget	The yearly operational expenses of running the River Murray system (e.g.. costs of damn, weir, lock operations) are shared between the Commonwealth and the New South Wales, South Australian, and Victorian government. The level of contributions by each party depends on the type of funded activity.
Permanent crops	Trees or shrubs, not grown in rotation, but occupying the soil and yielding harvests for several (usually more than five) consecutive years. Permanent crops mainly consist of fruit and berry trees, bushes, vines and olive trees and generally yield a higher added value per hectare than annual crops.
Surface water	Water that flows over land and in watercourses or artificial channels.

Sustainable diversion limit (SDL)	Maximum amount of water that can be taken for consumptive use reflecting an environmentally sustainable level of take (i.e., extractions must not compromise key environmental assets, ecosystem functions or productive base).
Transboundary water	A body of water that is shared by or forms the boundary between two or more political jurisdictions.
Unbundling	The legal separation of rights to land and rights to access water, have water delivered, use water on land or operate water infrastructure, all of which can be traded separately.
Unregulated river system	Rivers without major storages or rivers where the storages do not release water downstream.
Water Act 2007	An Act to make provision for the management of the water resources of the Murray-Darling Basin, and to make provision for other matters of national interest in relation to water and water information, and for related purposes.
Water allocation	A specific volume of water allocated to water (access) entitlements in a given season, according to the relevant water plan and the water availability in the water resource in that season (also known as temporary water).
Water buyback program	Known as *Restoring the Balance*, this was a government market-based instrument in Australia to produce environmental benefits in deteriorated sites across the Murray-Darling Basin by purchasing water entitlements from willing irrigators. In other words, water, previously allocated for consumptive uses, is reallocated back to the environment.
Water entitlement	A perpetual or ongoing entitlement to exclusive access to a share of water from a specified consumptive pool as defined in the relevant water plan (also known as permanent water).
Water for the Future	A 10-year initiative of the Australian government to better balance the water needs of communities, farmers, and the environment and to prepare Australia for a future with less water. Initially, the budget was set at AUD$12.9 billion, which allocated AUD$3.1 billion towards a water buyback program and AUD$5.8 billion towards Sustainable Rural Water Use and Irrigation Infrastructure (SRWUI) projects. Over the years, the budget was increased, primarily for the purpose of the infrastructure program.
Water recovery	Recovering water for the environment through investing in infrastructure to achieve greater efficiency and through the purchase of water entitlements.
Willingness to pay/accept	The acceptable bid amount that an individual is prepared to pay/receive for acquiring/giving up the good in question.

Abbreviations

ABARES	Australian Bureau of Agricultural and Resource Economics and Sciences
ABC	Australian Broadcasting Corporation
ABS	Australian Bureau of Statistics
ACCC	Australian Competition and Consumer Commission
ATO	Australian Tax Office
CEWH	Commonwealth Environmental Water Holder
COAG	Council of Australian Governments
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DAWE	Department of Agriculture, Water, and the Environment (Cth)
DAWR	Department of Agriculture and Water Resources (Cth)
DCCEEW	Department of Climate Change, Energy, the Environment and Water (Cth)
DI	Department of Industry (NSW)
DPI	Department of Primary Industries (NSW)
GDP	Gross domestic product
GL	Gigalitre (one billion litres)
GMID	Goulburn-Murray Irrigation District
GMW	Goulburn–Murray Water
GS	General security
GVAP	Gross value of agricultural production
GVIAP	Gross value of irrigated agricultural production
HS	High security
IIO	Irrigation infrastructure operator
IVT	Inter-valley Trade restriction
KL	Kilolitres
LMW	Lower Murray Water
LTAAY	Long term average annual yield
LTDLE	Long-term diversion limit equivalent
MDB	Murray–Darling Basin
MDBA	Murray–Darling Basin Authority
MI	Murrumbidgee Irrigation Limited
MIL	Murray Irrigation Limited
MINCO	Murray-Darling Basin Ministerial Council
ML	Megalitre (one million litres)

nMDB	northern Murray-Darling Basin
NWC	National Water Commission
NWI	National Water Initiative
PC	Productivity Commission
SDL	Sustainable diversion limit
SDLAM	Sustainable Diversion Limit Adjustment Mechanism
sMDB	southern Murray-Darling Basin
VWAP	Volume weighted average prices
WESA	Water for the Environment Special Account
WRP	Water Resource Plan

Executive Summary

Background

This report on the economic values of the Murray-Darling Basin (MDB) and their drivers was undertaken for the 2023 Murray-Darling Basin Outlook. It represents the first phase in the development of the economics chapter. Although the focus of the review was addressing the terms of reference, namely: a) what the main values are and how they are measured; b) what is known about the benefits and impacts of each value; c) what is known about the current condition and trend; d) what is known about the major risks, threats and recoverability and d) what is known about the future condition of each value, especially under climate change - what became clear in our initial review is the vast difference (both in terms of findings but also methodologies applied) that exists within the economic literature as to the actual impact of water recovery (or reduced water diversions) on economic values. Given that there is an ongoing concern, both politically and socially, regarding the impact recovery programs have on economic values, we therefore extended our report to undertake a systematic literature review, develop a quality assessment ranking framework and apply this framework to the relevant literature. The quality assessment allows quantification of the quality of previous studies and assists in establishing confidence in their findings in order to make robust policy decisions.

Therefore, the following chapters focus on a) a literature review of the economic values in the Basin; and b) providing a systematic literature review on drivers of water-related economic values and quality assessment of studies on the impacts of water reduction on such values in the MDB. **Chapter One** provides a broad overview of the MDB and water reform and policies to date. It emphasises that economic values in the MDB are variable and driven by various factors, including climate, external influences (terms of trade on agricultural profitability), public service investment, community variables, technology, trade and, to some extent, water recovery programs.

Economic Values

Chapter Two introduces the economic values in the Basin, broken down into three broad groups (Table on the following page provides a summary):

- **Direct economic values in the MDB**
    - Agricultural economic values (e.g., irrigated and dryland output – hectares, gross value of production, farm numbers, profitability, exports, capital stock)
    - Community economic values (e.g., jobs, income, gross state/regional product, local service provision)
    - Recreational, fishing and tourism economic values (e.g., boating, camping, commercial fishing, recreational fishing, water activities on river, accommodation, hiking, birdwatching, skiing)
    - Mining and energy economic values
    - Water market economic values (e.g., permanent and temporary trade (volumes traded and prices paid), participation)

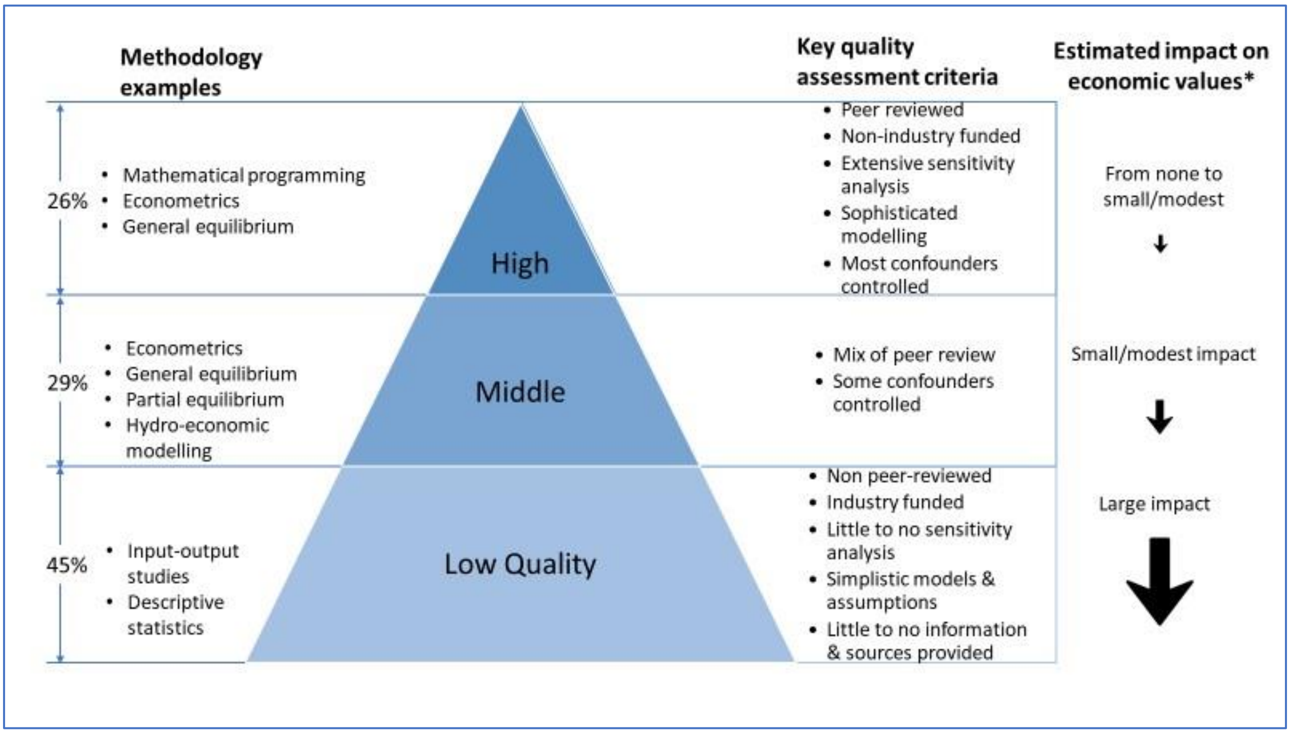
Economic Values of the Basin: A Summary

Categories	*Measures*	*Values*	*Trends*	*Major risks, future conditions*
Agricultural economic values	GVAP/GVIAP (data source: ABS)	About $29,000 million in 2020-21 for GVAP, and $8,500 million in 2020-21	GVAP increasing from 2007 to 2021, and GVIAP increasing in general from 2005-2021	The climate will be hotter and drier, with rainfall more variable. Australian agricultural productivity as a whole is projected to decline significantly by 2050 due to climate change. Farmers terms of trade important, along with other economic factors
	Profit (data source: ABARES)	Rate of returns highly variable, 3% average 2013-2015	Increasing slightly over time, but highly variable	Climate change will influence profits and reduce net revenue from irrigation significantly without adaptation. Farmers terms of trade important, along with other factors
Community economic values	GRP/GDP/GSP (data source: ABS)	$232 bill of GRP in MDB, 2020-21	Increasing since 2010	Drought, social services, climate change, terms of trade
	Employment rate (source: ABS population census)	About 1.63 million in 2020 to 2021	Increasing from 2010 onwards	Drought, social services, climate change, terms of trade
Recreational, fishing & tourism economic values	Recreation & Tourism (source: Tourism Research Australia, ABS, BLADE, consultancy & academic research)	Variety of different benefits found for skiing, boating, tourism	Increasing from 2001 onwards	High dependency on flow regime of ecosystem services makes the region highly vulnerable to climate change
	Economic values of fishing (source: NSW Recreational Fishing Survey, consultancy reports)	Gross output of recreational fishing in the Basin worth $108 million in 2018	Increasing during from 2014 onwards	Projected climate changes, declining river flows, and increasing salinity levels will be main threats and thus better management is required
Mining & Energy economic values	Mining & energy (sources: state gov data; ABS, private industry reports)	Substantial revenue, wages, and jobs	Uncertain	Climate change emissions, soil and water degradation issues, groundwater contamination
Water market economic values	Water prices and trade volumes (data sources: BoM; state water registers, academic and consultancy research)	$5+ billion turnover in the MDB in 2020-21	Increased over time	Water markets can be controversial, and therefore one potential risk is loss of public confidence and participation. Other risks to water market values are associated with water scarcity and climate change
Indirect value: ecosystem service values	Water quality (e.g. salinity, sediments) (source: water quality monitoring program - MDBA), biodiversity (irregular), carbon sequestration, consultancy and academic research	The MDB has high ecological values, with water supply, diverse species and other ecosystems providing diverse values	Salinity quality improved. Other ecosystems services decreased and experienced ecological value loss	Large water extraction and changed land use by expanded agricultural land, combined with climate change have affected water quantity, quality, and threat to aquatic species, thereby resulting in ecosystems value loss
Non-use economic values & cultural values	Option, bequest and existence & cultural values (estimated by methods such as choice modelling, travel cost method) (consultancy and academic literature)	Values vary by different study areas	Unknown	Climate change, over-water allocation, decreased water quality and quantity are main threats

- **Indirect economic values in the MDB**
  - Water quality and supply economic values
  - Ecosystem service economic values (water quality benefits from reduced salinity, reduced bank collapse, health benefits, improved water quality, carbon sequestration)
- **Non-use economic values in the MDB**
  - Option, Bequest and existence economic values (e.g., willingness to pay for environmental improvements)
  - Cultural economic values (e.g., First Nations values of water, businesses and employment).

Chapter Three provides the methodology of the systematic literature review. 106 relevant economic MDB studies were included in our review. **Chapter Four** shows that studies incorporating quantitative analysis of water economics issues in the MDB have increased over time, peaking in 2011, with relatively stable publications from 2012 onwards. Most studies have looked at the MDB as a whole, followed by the southern MDB, NSW, Victoria, northern MDB and SA. **Chapter Five** focussed on the studies that have sought to investigate the impact of various water recovery programs on economic outcomes. It also provided a quality assessment of such studies, looking at both internal and external validity issues. The figure below provides a summary. It highlights that the majority of studies conducted are classified as low quality (and dominated by input-output and descriptive statistics studies), and these are the studies that tend to find a large negative impact on various economic values.

Overview of water recovery studies by quality assessment and impact on economic values



Note: * Economic values include GDP, GRP, GRIAP, employment numbers, farm production, farm gross margins (which may decrease with water recovery). Other economic values such as water market prices have the opposite sign as some studies suggest they increase under water recovery. Diagram is not to scale.

Key Findings – Chapter 6

The impacts of water recovery on economic outcomes in the MDB remain one of the most contentious areas of interests to public policy in Australia. A cut in water allocated to agriculture is often portrayed as the root cause of reduced irrigated areas, reduced irrigated production, lower irrigated value of production, fewer jobs in irrigated agriculture and local communities due to decreased spending overall, together with dwindling populations in rural communities due to out-migration. Although it is much easier to quantify these direct costs of water recovery than trying to quantify the environmental benefits, our review of studies has highlighted that even quantifying the direct costs of water recovery to rural communities has been difficult to achieve, and extremely variable in quality.

The most significant problem that all analysts face with trying to model the impact of water recovery is to identify its causal impact, and disentangling true relationships between reduced water and various economic outcomes. Many studies assume that a 1% decrease in water extractions leads to an equal 1% decrease in irrigated hectares, which subsequently results in an equal 1% decrease in irrigation production – which in turn is associated with simple input-output multipliers, to suggest a loss of regional economic value and jobs. However, this ignores reality. A summary of key findings in the literature suggests:

- Overall, reducing consumptive water extraction does have a negative impact on irrigated agriculture. However, the magnitude of the relationship between water extracted and farm economic outcomes (irrigated area and revenue) can be substantially less than many studies have predicted where they have assumed a unit elastic response of production to water extractions;
- The positive impacts of buyback expenditure within the local economy have often been ignored;
- Not all farmers who sold water entitlements left farming, or suffered changes in production;
- Climatic and socio-economic factors are often a lot more important than water allocations for socio-economic outcomes;
- Negative buyback impacts are often overstated, whilst irrigation infrastructure subsidies are often understated; and
- Healthy rural communities depend on many other factors than water for irrigation.

The review identified many internal and external validity issues in the economic modelling studies. These issues included: small sample sizes; statistical modelling issues; causal policy impacts; sample selection biases; inadequate documentation; and no independent peer review. What was also clear was that studies that predicted significant impacts from water recovery were rated as low quality in our quality assessment, versus studies that suggested the impact was far less – which were much more likely to be rated as high quality.

The final chapter concludes with three main recommendations for further research, which, if implemented, will increase the robustness and validity of any assessment of water recovery on economic outcomes.

1 Introduction and Overview

1.1 Murray-Darling Basin (MDB)

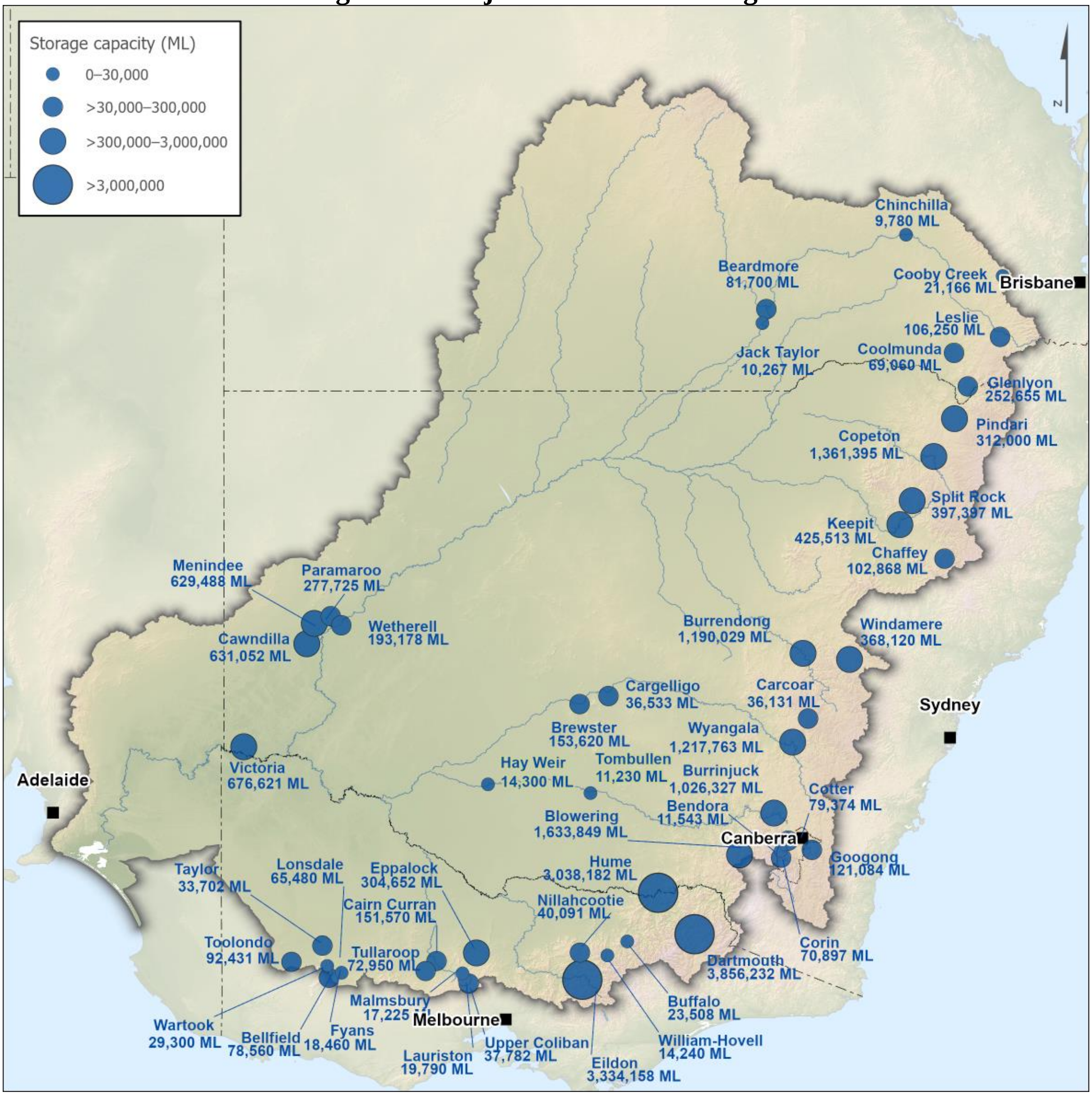
The MDB is the catchment for Australia's longest rivers, the Murray and the Darling Rivers. The Basin covers an area of more than 1 million square kilometres (14% of Australia's total surface area), and includes 75% of New South Wales (NSW), more than 50% of Victoria (VIC), 15% of Queensland, 8% of South Australia (SA), and all of the Australian Capital Territory (ACT). There are 22 major catchments (or sub-Basins) within the MDB. The northern MDB (nMDB) consists of the catchments draining into the Darling River, and the southern MDB (sMDB) is formed by the catchments draining into the River Murray (see Figure 1.1) (MDBA, 2016a).

Figure 1.1 Boundary of MDB

Source: MDBA (2016a)

As on average more than 40% of total surface water runoff is diverted for consumptive uses, water resources in the MDB are highly developed (see Figure 1.2). The nMDB, despite consisting of mostly unregulated river systems, encapsulates major storages along the Border, Gwydir, Namoi and Macquarie Rivers. The southern MDB is dominated by regulated river systems, with some of the biggest storages (Lake Dartmouth, Lake Hume, Lake Eildon and Lake Victoria) along the Rivers Murray and Goulburn, providing reliable water supply for communities and agriculture (BOM, 2021).

Figure 1.2 Major MDB water storages



Source: BOM (2021)

From the mid-1980s, uptake of private dams and off-river storages (also called ring-tanks) increased widely on the floodplains, with the purpose of these storages to ‘capture’ flows from unregulated tributary rivers, spills of major dams and floodplain inundation. As heavy rainfall and tropical cyclone events are predicted to become more frequent, rainfall variability in the north-eastern regions of the MDB will increase. Furthermore, with reductions in water availability and increased temperatures, drought frequency is predicted to increase in the southern and south-eastern regions (CSIRO, 2012a).

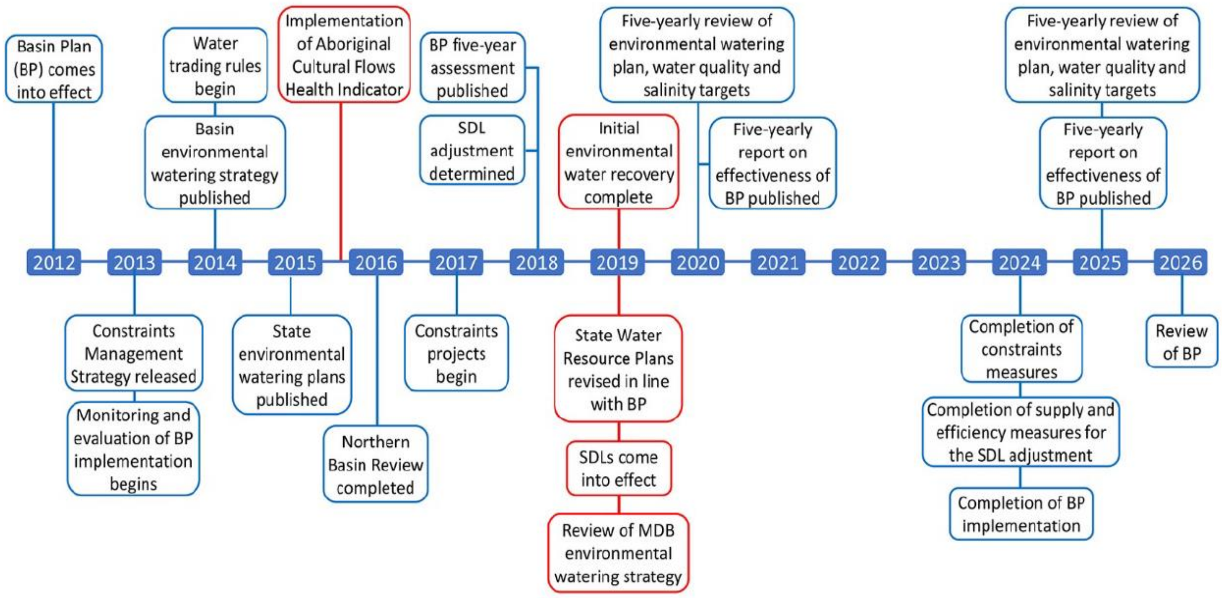
The MDB is Australia's most important agricultural region and also an area of great ecological, cultural and recreational significance (MDBA, 2009); it is home to 2.8 million people of which 75,000 are indigenous (Taylor et al., 2016; Sefton et al., 2020). Yet, many indigenous communities are experiencing poor socio-economic conditions, with recent causes identified as drought, environmental decline, and being excluded from water reform benefits (Sefton et al., 2020). Agriculture is of high importance in the Basin, as outlined further in Chapter 3: Economic values of the Basin.

1.2 Water Policy Reform

Although early water governance and water policy reform in the MDB began in the late 1890s (Guest, 2016), this report focusses on the period since 2007, which is dominated by environmental water recovery and water market governance initiatives under the Basin Plan framework. As legislated by the *Water Act 2007* (Cth), the Basin Plan is to determine a sustainable level of water extraction from the MDB, the Sustainable Diversion Limit (SDL), improve water market governance, environmental management, and indigenous and cultural values. As the MDB as a whole was (and still is) over-extracted at the time of the Water Act, setting the SDL under the Basin Plan includes implementing mechanisms and initiatives that recover water for the environment from the consumptive pool (Walker, 2019).

Figure 1.3 shows a timeline of Basin Plan implementation. This report does not provide detailed commentary on Basin Plan issues, for further detail and issues associated with the Basin Plan, see Grafton and Wheeler (2018); AAS (2019), Wheeler et al. (2020a) and Grafton et al. (2020).

Figure 1.3 Basin Plan implementation timeline



Source: Grafton et al. (2020, p. 5)

The Murray-Darling Basin Authority (MDBA) is responsible for Basin-wide planning and Basin Plan implementation. The states are responsible for managing water use within legislated agreed limits and to contribute to Basin Plan implementation. Authorities with regulatory responsibilities with regards to MDB water include the Australian government, state governments, the MDBA, the Inspector General of Water Compliance, the Productivity

Commission (PC) and the Australian Competition and Consumer Commission (ACCC) (which enforces and monitors water market and charge rules).

1.2.1 Water recovery for the environment

One key aspect of the Basin Plan is its mandate to recover water for the environment to achieve an MDB-wide SDL (MDBA, 2011). Water recovery started in 2007-08, with the National Plan for Water Security (renamed Water for the Future in 2008), with originally $12.9 billion allocated over ten years to support water reform. The largest program funding was for water infrastructure subsidies ($5.8 billion was originally allocated for this – known as *Sustainable Rural Water Use and Irrigation Infrastructure* program), followed by water entitlement purchases ($3.1 billion allocated for this – known as *Restoring the Balance* program) (ANAO, 2011).

As at 31st August 2022, current estimates are that the water recovered via buyback cost $2,109/ML through the buyback program, and water recovered via irrigation infrastructure cost $6,557/ML (3.1 times more), measured on a long-term average annual yield (LTAAY) (DCCEEW email communication 3rd Nov 2022).

Further ongoing (and unpublished) analysis on the water recovery by irrigation infrastructure reveals huge disparity in $/ML. Even on a LTAAY (e.g., apples are apples) basis, some farmers were paid as little as $55/ML, while other projects received on average over $30,000/ML for water recovery. How is such disparity possible? There were at least 13 different irrigation infrastructure programs that were developed to recover water and funded under the Sustainable Rural Water Use Infrastructure Program. Each program had different criteria, eligibility, objectives, budgets and methods/activities allowed. At least one of these schemes – the scheme SA River Murray Sustainability Program, allowed for other (non-irrigation infrastructure) farm activities to be subsidised instead. For example, irrigators could use the money to subsidise various farm productive activities (e.g., netting fruit/nut trees), and giving up some of their water entitlements as part of the program.

Wheeler et al. (2020a) provides a summary of the main justifications put forward for subsidising irrigation infrastructure in order to recover environmental water over buying water directly back, namely: a) farm productivity: increases farm productivity (Hughes et al., 2020) and hence makes recovery more politically acceptable; and b) water quality arguments: upgrading irrigation infrastructure can reduce saline return flows into the rivers (Wang et al., 2018). However, there are many other potential negative consequences of irrigation infrastructure subsidies. As summarised in Wheeler et al. (2020a), with reference to other literature in the MDB on this topic, they include: Cost – actual direct recovery and transaction costs; Governance issues; Return flows and additionality issues; Rebound effect on irrigated land area; Utilisation; Substitution; Equity; Floodplain harvesting; and Resilience.

Given ongoing opposition to both the Basin Plan and water recovery, since 2013, buying water from willing irrigators through open tender was shelved, with instead some focus given to 'strategic purchases' via closed and direct negotiations with large agri-corporates. Such purchases have been criticised due to their lack of transparency, potentially inflated values, and negative environmental externalities (Grafton, 2019; Seidl et al., 2020a). Indeed, flawed evaluation of water entitlement values has been at the centre of the controversies for both the Broken Hill pipeline business case, and the strategic purchase of overland flow water entitlements for environmental purposes from Eastern Australian Agriculture in 2017 (DPI, 2016; The Senate, 2018; Slattery & Campbell, 2020).

In 2015, legislation was introduced capping buyback purchases of water entitlements at 1,500GL (AAS, 2019). In 2018, further policy amendments were introduced that reduced entitlement recovery to 2,680 GL (plus a further 605GL reduction, subject to the implementation of 36 'supply measure' projects that are meant to offset water that would otherwise have to be recovered under the Plan in exchange for 'equivalent environmental outcomes') (Grafton, 2019; Productivity Commission, 2018). Suffice to note that these projects have been heavily criticised and have a very high probability that they will not achieve their predicted savings (Colloff & Pittock, 2019; Productivity Commission, 2018). Section 1.2.2 below provides more detailed discussion on some of these projects.

While the reemphasis on irrigation infrastructure water recovery is the preferred option for many farmers - though note, many do prefer market-based options (Loch et al., 2014), it is not cost-effective, and as outlined above may not meet long-term sustainability goals of being able to flexibly respond to uncertain and variable future water supply.

Although overall environmental water ownership has increased substantially since Basin Plan inception, critics argue that not enough water has been returned to the environment as environmental recovery has been less than expected (Wentworth Group of Concerned Scientists, 2020), and that there are considerable questions surrounding water extraction and consumption that need to be addressed (Wheeler et al., 2020a).

As part of the Basin Plan negotiations over the additional 450 GL, New South Wales and Victoria were only to accept the additional volume if it could be achieved through infrastructure upgrades, and crucially only if these were possible with neutral or positive socio-economic impacts. Of those 450 GL, 62 GL are simultaneously part of the 605 GL of 'supply measure' projects and were (partially funded) from the Water for the Environment special account (DAWE, 2021). The remaining 388 GL can be achieved in part or in full provided there are no negative socio-economic impacts involved (MINCO, 2018). This is commonly referred to as the socio-economic neutrality test (Aither, 2017b), and is considered to be highly dubious and illogical for a variety of reasons (Walker, 2019). The 450 GL have been a contentious topic between the states in so far as New South Wales and Victoria regard them as optional to the 2,750 GL of recovery, whereas South Australia views them as integral to be able to achieve a 3,200 GL Basin Plan recovery target.

1.2.2 Sustainable Diversion Limit Adjustment Mechanism (SDLAM) projects

The Basin Plan allows for catchment sustainable diversion limits to be adjusted (+/- 5%), in case (infrastructure) projects can be implemented that achieve Basin Plan environmental outcomes with less water. This is called the Sustainable Diversion Limit Adjustment Mechanism (SDLAM), and projects eligible to be considered are supply, constraint, and efficiency projects. Hence, it allowed for actual direct water recovery to be reduced from the 2,750GL (as outlined above). Supply projects represent river management initiatives that deliver water for the environment more efficiently, whereas constraint projects aim to overcome physical barriers to environmental water delivery, for example physical infrastructure that allows water to reach a wetland without having to flood the floodplain. Efficiency projects aim to change water use practices and thus save water for the environment, such as lining of irrigation channels (MDBA, 2022). The 2,750GL of water to be recovered from consumptive purposes can be reduced, if supply measures lead to equivalent environmental outcomes with less water, or if efficiency measures make water delivery for irrigation more efficient (MDBA, 2015).

Arguably the most prominent and controversial example for a supply project is the Menindee Lakes Water Savings project. Under the project, the NSW government claims water savings of at least 72 GL per year, through reduced evaporative losses from the Lakes and reduced delivery losses in the Lower Darling River, and seek an equivalent reduction in its environmental water recovery target (DI, 2018). Critically important to the success of the Menindee Lakes Water Savings project is securing town water supply for Broken Hill, which relied on the Lakes for supply, and would be negatively impacted by changed operating rules under the project (.i.e., keeping the Lakes at lower levels more often to reduce evaporation). To this end, the NSW government decided to supply Broken Hill with water directly from the Murray River via a $500 million pipeline of 270 km in length, constructed in 2018 (Jackson & Head, 2020), which has also been controversial.

More broadly, the SDLAM and associated projects have experienced significant delays with 19 projects completed by December 2021, delivering only 2.2 GL of the targeted 450 GL (MDBA, 2021), and been criticised for lack of transparency or absent project information and for funding initiatives that do not provide any environmentally beneficial outcomes (McBride, 2021). A review commissioned by the MDBA, Indec (2021) provided a status assessment for the SDL Adjustment Mechanism Program, finding:

- 81% (30) of individual projects should be able to be delivered before 30 June 2024 without major intervention, delivering 73.5% (444.7GL) of SDLAM offset volumes.
- 19% (7) of individual projects are unlikely to be delivered before 30 30 June 2024 without major intervention and were categorised as being ‘At Risk’. These projects account for 26.5% (160.5GL) of SDLAM offset volumes.

If the remaining 160.5GL of offsets through SDLAM projects are not achieved by 30 June 2024, there will be a new water recovery liability and the options identified by Indec (2021) included:

- Allowing more time for projects to be completed;
- Developing replacement projects with extended timeline;
- Finding further off-farm efficiency measures;
- Further on-farm efficiency measures that satisfy agreed socio-economic criteria; and
- Buying back more entitlements.

Other options could also include use of temporary trade, and mandatory cuts (compensated or uncompensated) to the long-term allocations to water entitlements.

2 Identifying economic values of the Basin

2.1 Why economic values of the Basin are important

2.1.1.1 What are the main values and how are they usually measured – data, indicators?

To help guide our framing of key economic values of the Basin that are relevant to water uses, we apply a framework of the total economic value of water, which is made up of direct and indirect use values. *Direct use values* are benefits that directly accrue to individuals who use/interact with the water resource including householders, rural and urban producers, and recreational users.

Indirect use values arise where there is no direct contact with actual water, but indirectly people benefit from it. For example, water views and the aesthetics it generates frequently increase property values. *Non-use values* are attributed to people knowing that rivers have decent flows, the environment is healthy; for now, and into the future, but direct contact is not required for any valuation. For example, such values might be ascribed to people in North America who care about the quality of water resources in the MDB, despite having no plan to ever visit or any links with people or land there.

Non-use values include *option values*, *quasi-option values*, *existence values* and *bequest values*. Option values are values for future use, existence values are associated with knowing about vegetation's presence and bequest values arise from the desire to preserve the public good aspect of vegetation for future generations (Grafton & Wheeler, 2015).

In sum, a healthy MDB has many other dimensions beyond its utilitarian purposes. To effectively manage water and land resources, the community's beliefs, perceptions and values should be taken into consideration in policy-making.

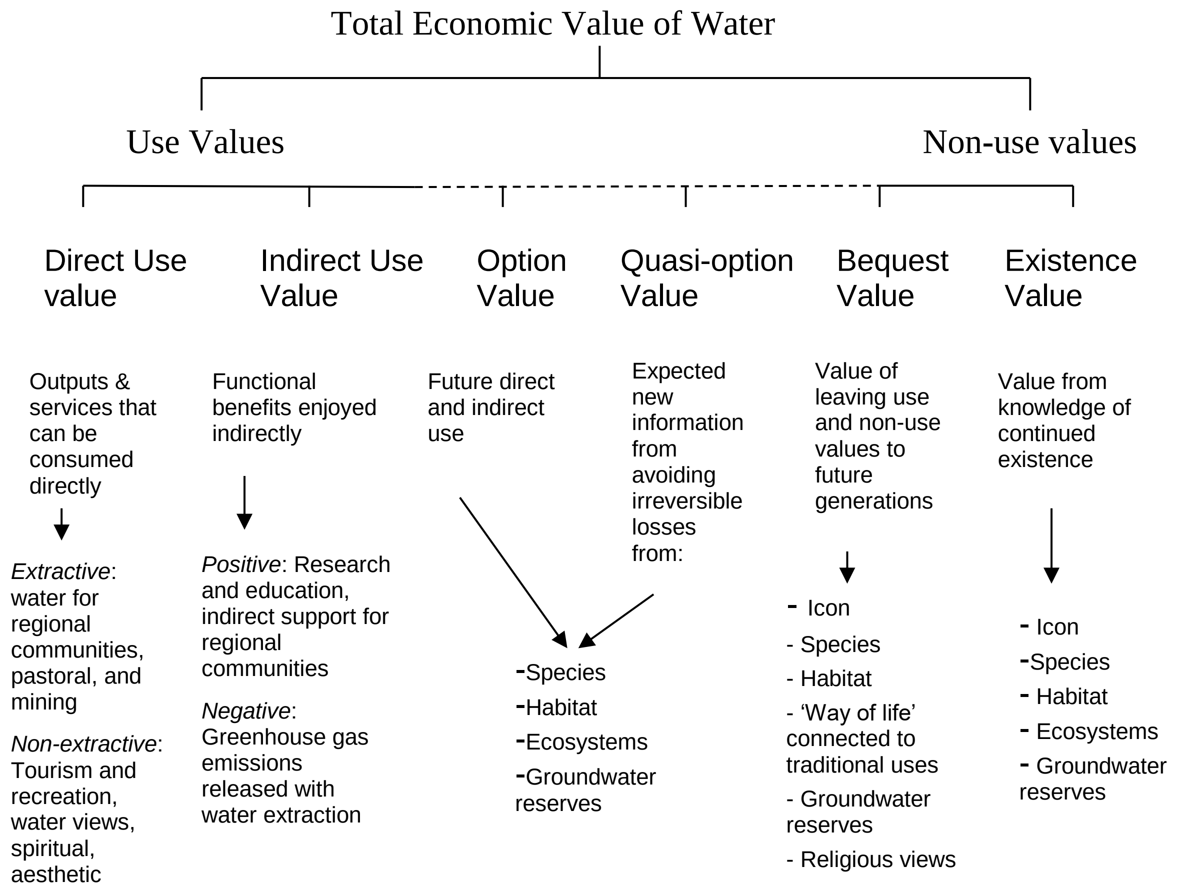
2.2 What are the main values and how are they usually measured - data, indicators?

What are some of the economic values in the Basin? Figure 2.1 provides an overview of the values that as a society we attach to water. These values are all obviously important as they provide the livelihoods upon which humans exist. In order to provide a detailed literature review on these economic values in the Basin, we break them down into three broad groups:

- **Direct economic values in the MDB**
    - Agricultural economic values (e.g., irrigated and dryland output – hectares, gross value of production, farm numbers, profitability, exports, capital stock)
    - Community economic values (e.g., jobs, income, gross state/regional product, local service provision)
    - Recreational, fishing and tourism economic values (e.g., boating, camping, commercial fishing, recreational fishing, water activities on river, accommodation, hiking, birdwatching, skiing)
    - Mining and energy economic values
    - Water market economic values (e.g., permanent and temporary trade (volumes traded and prices paid), participation)

- **Indirect economic values in the MDB**
    - Water quality and supply economic values
    - Ecosystem service economic values (water quality benefits from reduced salinity, reduced bank collapse, health benefits, improved water quality, carbon sequestration)
- **Non-use economic values in the MDB**
    - Option, Bequest and existence economic values (e.g., willingness to pay for environmental improvements)
    - Cultural economic values (e.g., First Nations values of water, businesses and employment).

Figure 2.1 Values of Water



Source: Adapted from Grafton & Wheeler (2015).

Available economic techniques for valuing the above values include:

- **Market value approaches:** use observable market data for prices to estimate costs/benefits of direct and indirect water values of the Basin. Some of the most used market value approaches include change in productivity, replacement cost and defensive expenditure;
- **Surrogate Markets - Revealed preference approaches:** uses price or cost of surrogate goods or services to reveal willingness to pay for water quality. Some of the most used techniques include travel cost, hedonic pricing and benefits transfer; and

- **Simulated Markets - Stated preference approaches:** construct information from respondents to propositions that ask them to state their preference for different outcomes. These are usually benchmarked against a plausible monetary outcome. Some of the most used surrogate market approaches include contingent valuation and choice modelling (conjoint analysis)

Obviously, the easiest values to measure are extractive direct use values – as these uses are traded in a marketplace and the price paid for a good provides one measurement of its value, and many market value approaches can be applied. Non-extractive and indirect values are the next easiest, with a mix of market and surrogate revealed market approaches, followed by simulated markets using stated preference approaches. More discussion on the economic methods that can be used to evaluate changes in economic values are provided in Appendix B. They are grouped into three broad categories: 1) optimisation and mathematical models such as general equilibrium model, partial equilibrium model, and input-output model; 2) econometric models; and 3) descriptive and qualitative analyses. Appendix B describes these economic models, their strengths and weaknesses and provides additional references for more detail.

Table 2.1 Common Measurements/Indicators of Economic Values in the Basin

Type of Economic Value	***Description and Source***
Agricultural economic values	Gross value of agricultural production (GVAP) and gross value of irrigated agricultural production (GVIAP) by industry (data source: ABS)
	Number of farms (irrigated and dryland), Various forms of farm profitability (data sources: ABS, ABARES, ATO data sources)
Community economic values	GRP/GDP/GSP (data source: ABS)
	Employment numbers, unemployment rate (data source: ABS)
Recreational, fishing & tourism economic values	Recreation & Tourism (sources: Tourism Research Australia, ABS, BLADE, consultancy & academic research)
	Economic values of fishing (sources: NSW Recreational Fishing Survey, consultancy reports, academia research)
Mining & Energy economic values	Mining & energy (sources: state gov data; ABS, private industry reports)
Water market economic values	Water prices and trade volumes (data sources: BoM; state water registers, academic and consultancy research)
Indirect value: ecosystem service values	Water quality (e.g., salinity, sediments) (sources: water quality monitoring program -MDBA), Biodiversity and ecological values, carbon sequestration, (data sources: consultancy & academic research)
Non-use economic values & cultural values	Option, bequest and existence & cultural values (estimated by methods such as choice modelling, travel cost method) (consultancy & academic literature)

Following the typology of economic value put forward here, we provide a breakdown of economic values into use (direct and indirect) and non-use values in the Basin.

2.3 Direct economic values in the MDB

2.3.1 Agricultural economic values

2.3.1.1 What is known about the benefits and impact of agricultural economic values?

Irrigated agriculture in the MDB makes a significant contribution to both national and regional economies (ABARES, 2014) but experienced a range of structural challenges the past few decades, such as droughts, over-allocation of resources, economic depressions, inefficient government subsidy schemes, rising water tables and salinity levels, and increasing water prices (Hallows & Thompson, 1995).

Irrigated agriculture produces about a quarter of the total agricultural value from less than 1% of the area of agricultural land (ABS, 2018). Much of Australia's irrigation occurs in the Murray–Darling Basin where over two-thirds of Australia's irrigation water is used for food production (MDBA, 2022). The gross value of MDB irrigated production is strongly influenced by water availability, with broadacre crops, such as rice and pasture, contracting the most in times of drought (see Table 2.1).

The data on agriculture can be obtained from a variety of sources. Estimates of GVIAP and GVAP are based on production, commodity price and water use data derived from ABS' Rural Environment and Agricultural Commodities Survey, and from non-ABS sources, including marketing authorities and industry bodies. GVIAP is available annually from 2000-01 to 2017-18 (ABS, 2019). The latest year's data is by NRM, state and available for the whole MDB as one region; the industries included are rice for grain, cereal for grains and seed, cotton, sugar cane, other broadacre crops, hay, nurseries/cut flowers/cultivated turf, vegetables, fruits/nuts (exclude grapes), grapes, dairy production, meat cattle production, sheep, and other livestock production. GVIAP is available annually from 2018-19 to 2020-21 from the ABS Water Account, Australia. GVAP is available annually from 1981-82 to 2020-21 (ABS, 2022a). The latest year's data is by State, ASGS regions, NRM, LGA, and for the whole MDB as one region. More detailed industry data is available at a specific crop or livestock level, such as wheat and various livestock. Issues persist in trying to get consistent small area level data over time, given differing boundary changes, as well as changing minimum values attached to agricultural businesses.

The production of agricultural commodities and agriculture land use including cereal and broadacre crops, fruit and vegetables and livestock on Australian farms are contained in ABS Agricultural commodities data (ABS, 2022b). Agricultural commodities data are available annually from 1997-98 to 2020-21. The latest year's data are primarily from the 2020-21 Agricultural Census. In non-census years, data in this publication are from the Rural Environment and Agricultural Commodities Survey (REACS), conducted annually in non-census years. The latest year's data is by State, ASGS regions, NRM, LGA, and for the whole MDB as one region. More detailed industry data is available at specific crop or livestock level. Again, issues persist in trying to get consistent small area level data over time, given differing boundary changes.

Other farm financial data including farm business profit, profit at full equity, change in farm debt, total family income; production data and agricultural labour data can be gained from ABARES Farm Data Portal (ABARES, 2022a). These data are by state or ABARES region and estimates from ABARES surveys of broadacre and dairy farms, covering the period from 1989-1990 to 2021-2022. However, the MDB is not a region separately listed. Moreover, ABARES conducted other surveys of irrigation farms in selected industries and regions in the MDB since 2006–07 (ABARES, 2022b). The MDB Irrigation Survey includes cotton, rice, dairy and horticulture farms in ten regions of the Basin. Most estimates are for the MDB, but individual farm-level data are unavailable. In this dataset, at the time of writing, farm cash income (total cash receipts minus total cash costs) and rate of return (average annual rate of return to capital, excluding capital appreciation) are available from 2006-07 to 2015-16. Rate of return for some industries are available in years prior to 2006 from other sources. Furthermore, business net income is available in ABS Microdata (BLADE, the Business Longitudinal Analysis Data Environment) for every farm business at the business level from 2001-02 to 2020-21 (ABS, 2022c). The geographic level of data is at the mesh block level. BLADE also includes merchandise export data, available from 2003-04 to 2018-19, for every exporting farm business at the business level. Data suggests that the average rate of return from 2011-12 to 2018-19 for cotton, dairy and rice farms in the MDB were 5.1%, 2.6% and 2.5% respectively. Cotton farms had the highest rate of return compared to dairy and rice farms. The rate of return in broadacre trended downward between 2014-15 and 2018-19 but usually remained above 0% (Aither, 2022).

Finally, data such as the number of farmers can be obtained from the population census in ABS, which covers the later census years of 2001, 2006, 2011, 2016 and 2021 (ABS, 2021). BLADE also provides the number of FTEs and headcount of employing businesses, including farms, at the business level from 2001-02 to 2020-21. ABS agricultural census also contains water use data (e.g., statistics on irrigation, including pastures and crops irrigated, and water sources) in census years. Moreover, farmland value can be obtained from Rural Bank (2022). Median price ($/ha) is reported by region of states (for example, North West, South West, Northern, Gippsland in Victoria), and by municipality. Median price by state and region is also available in figures from 1995 to 2021. Transactions weighted median price in municipalities in the MDB may be used to calculate the price for MDB as a whole. Previous years' reports are also available.

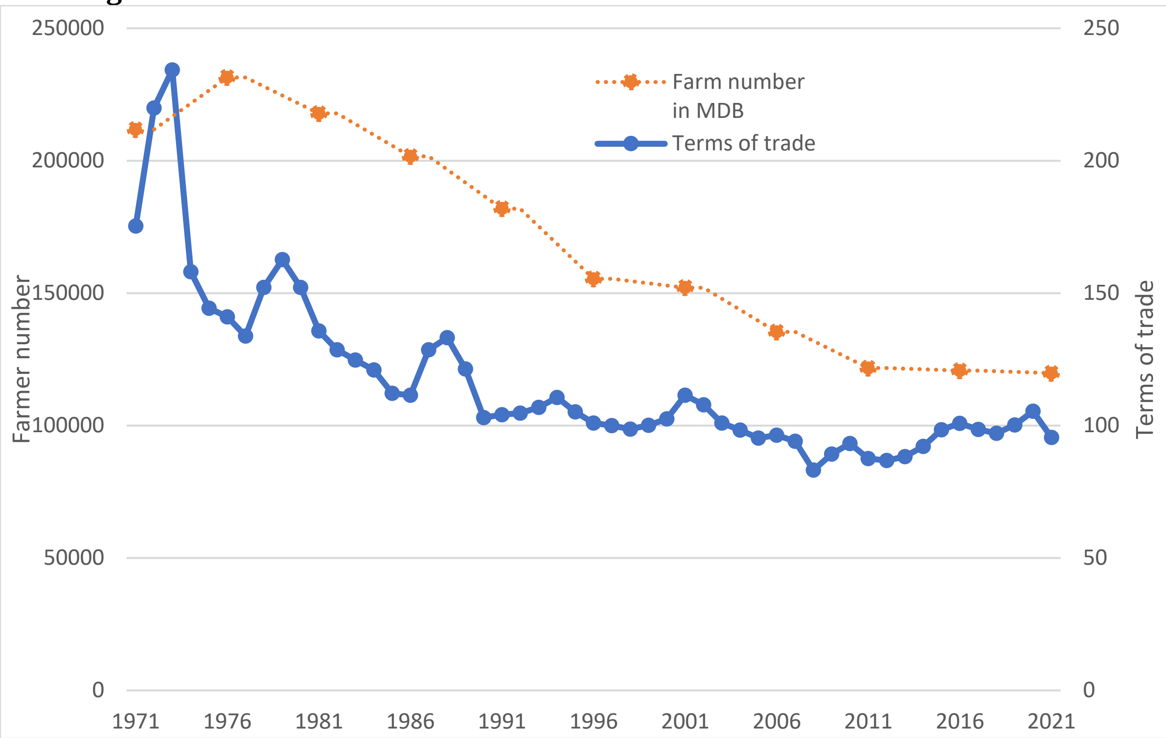
2.3.1.2 What is known about current condition and trend of agricultural economic values?

The number of farmers in the MDB has steadily declined since the mid-1970s, a result of declining term of trade for farmers, and other factors (Figure 2.2). However, despite the reduced number of farms and the structural changes in agriculture, the overall gross value of agricultural commodities produced in the MDB has not been affected; it followed a fairly steady upwards trajectory for the last 15 or so years, both in MDB regions and MDB states[1] overall (see Figure 2.3 and Figure 2.4). Also note the increasing value of horticultural commodities produced over time.

The gross value of irrigated agriculture production (GVIAP) increased over the years between 2005 to 2015 (Figure 2.5). The trend of GVIAP after 2015 is not that obvious as it becomes more variable yet stays at a high level.

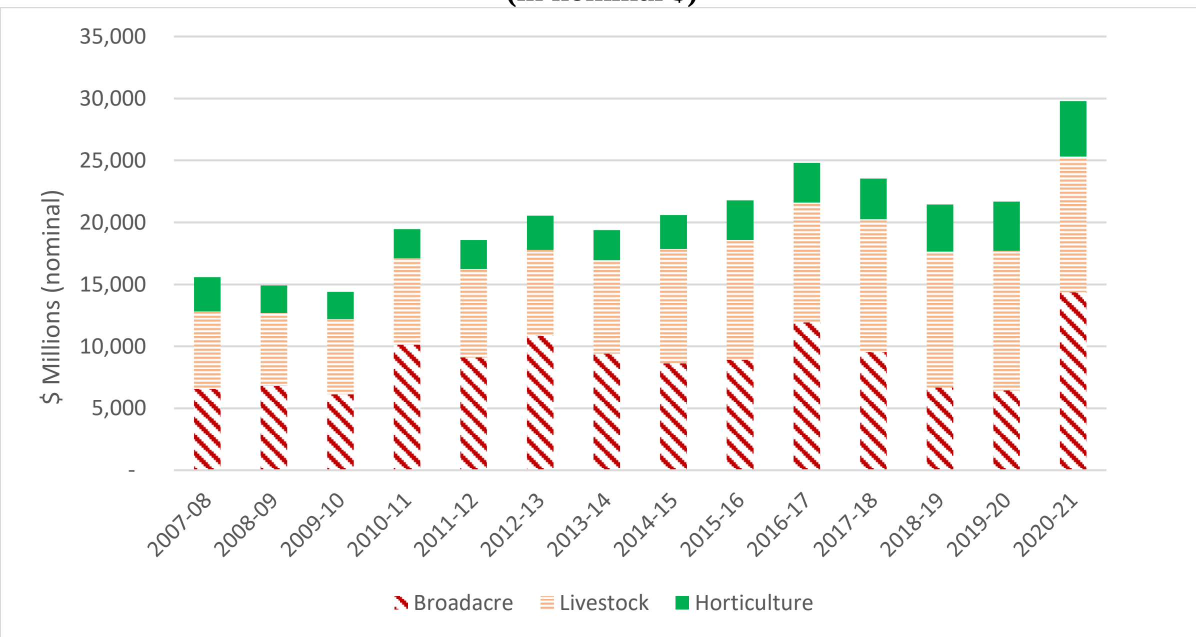
[1] NSW, VIC, QLD, SA, and ACT

Figure 2.2 Farmer numbers in the MDB and terms of trade from 1971 - 2021



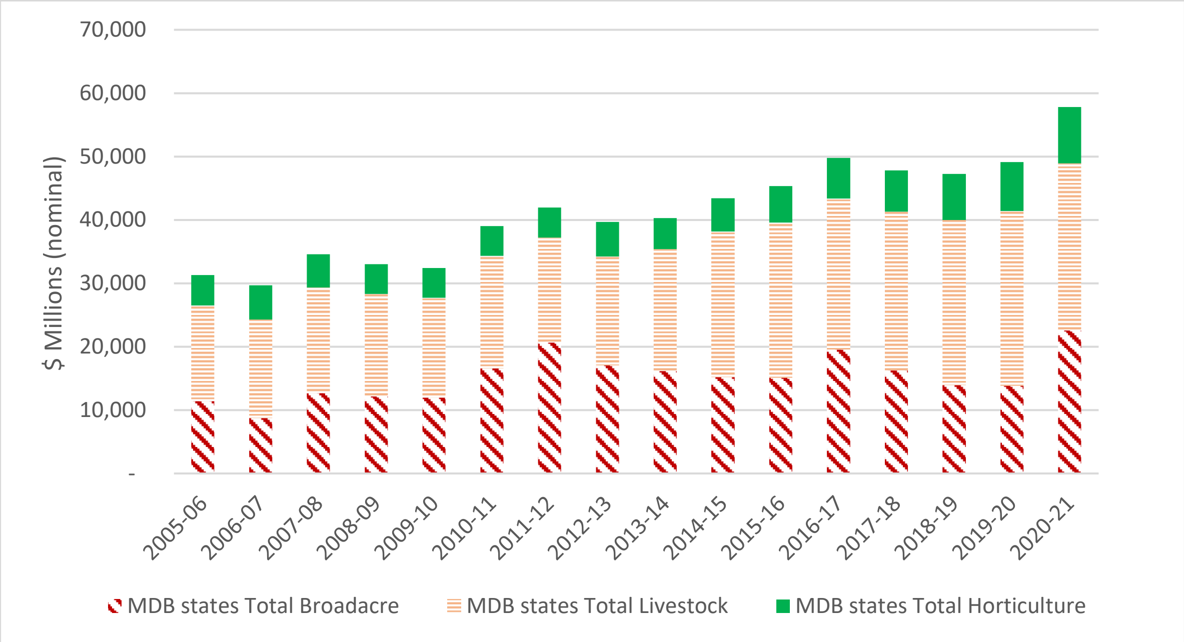
Source: Updated from Wheeler et al. (2020b)

Figure 2.3 Gross value of agricultural commodities produced in the MDB since 2007/08 (in nominal $)



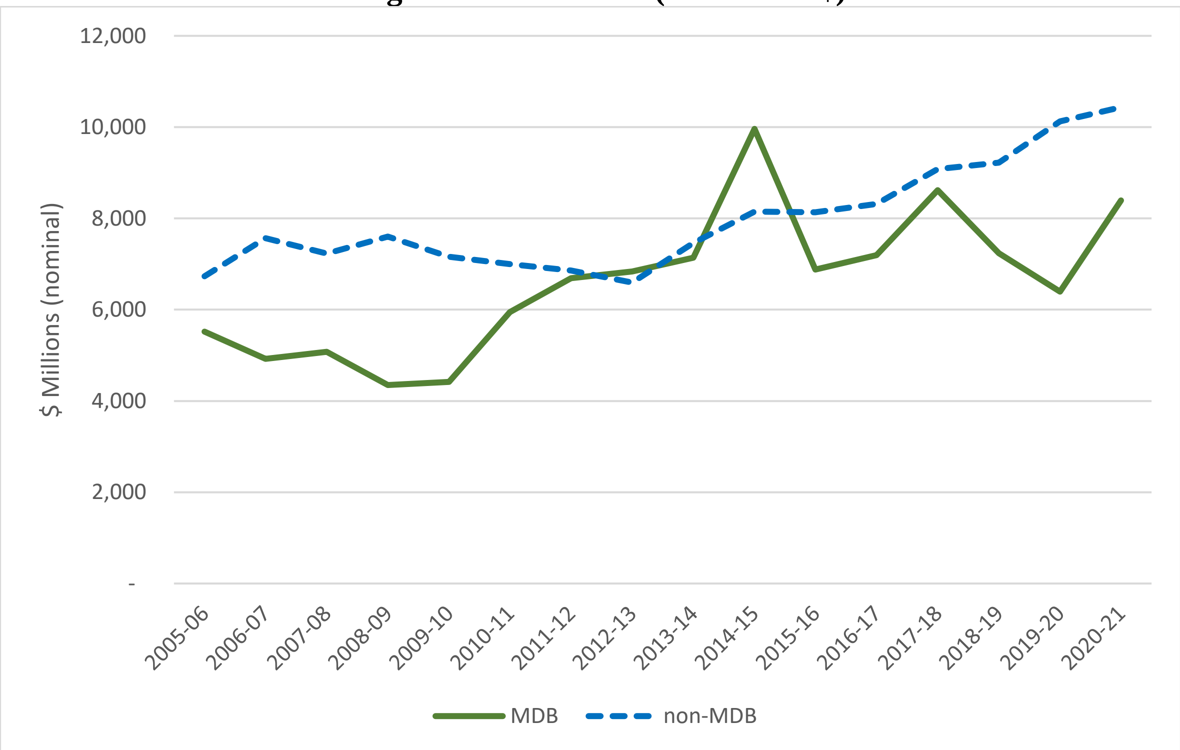
Note: MDB regions determined by NRM codes illustrated in Appendix A
Source: compiled from data in ABS (multiple years)

Figure 2.4 Gross value of agricultural commodities produced in MDB states since 2005/06 (in nominal $)



Source: compiled from data in ABS (multiple years)

Figure 2.5 Gross value of irrigated agricultural production in MDB and non-MDB regions since 2005/06 (in nominal $)

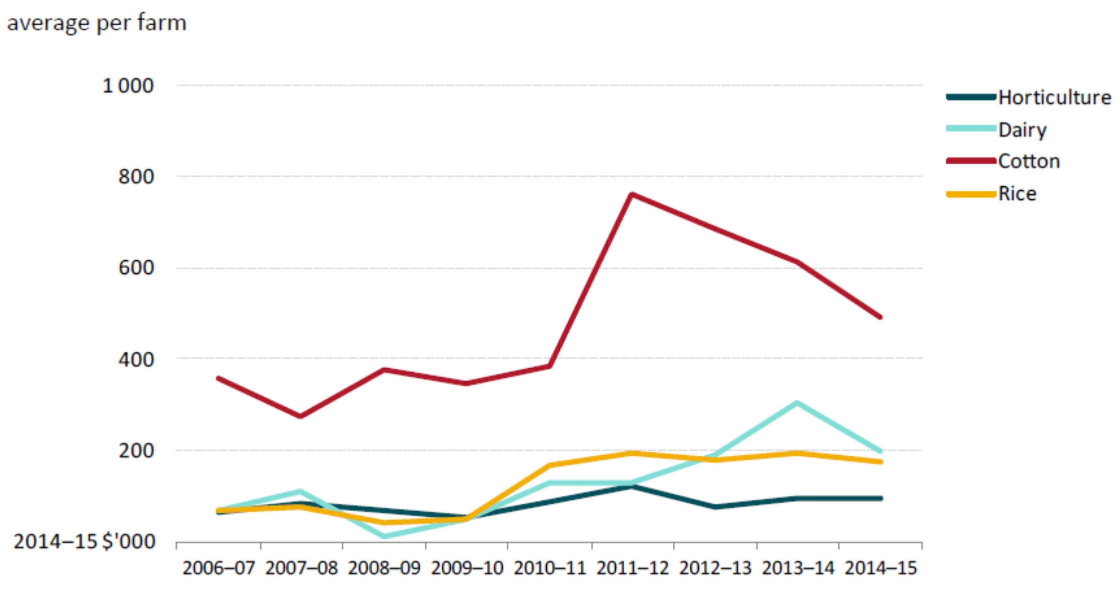


Note: GVIAP (non MDB) not reported for 2005/06 - 2012/13. We calculated GVIAP (non-MDB) = GVIAP (Australia) – GVIAP (MDB) for these years.
Sources: compiled from data in ABS. ABS (multiple years). Gross Value of Irrigated Agricultural Production, Australia. ABS, Canberra. ABS. (2022d). Water Account 2020-21, Australia. ABS, Canberra.

Despite the overall increase in the gross value of agricultural commodities produced, the area of annual irrigated agriculture in the MDB and volume of water extracted can vary substantially in response to several challenges, yet the number of agricultural businesses declined over the longer term (see Table 2.2). However, the problem with such ABS statistics on agricultural and irrigation businesses, is the definition of what constitutes an agricultural business (it changes over time). For example, in 2015-16, the ABS increased the EVAO (estimated value of agricultural operations) from $5,000 to $40,000, and this had the effect of 'eliminating' a substantial number of small farms from the recorded data. Hence, there seems to be greater decline in farm businesses than there actually was. This is why estimates of actual 'farmer' numbers (as in Figure 2.2) from census results are more reliable as a timeseries measure than estimates of businesses from the annual agriculture surveys. This also has a potential (albeit very small) impact on hectares irrigated and water applied.

Due to the influence of changing commodity prices, costs of farm inputs, climate conditions, and varying irrigation water availability, average farm cash income[2] in MDB varies substantially from 2006-07 to 2014-15 (see Figure 2.6). Generally, dairy farms have gotten increased average farm cash incomes from 2008-09 to 2013-14 despite slight fluctuation. After peaking in 2013-14, average farm cash incomes decreased. The trend of average farm cash incomes of cotton farms is very similar, as it was at its peak in 2011-12 and fell in subsequent years. The reduction probably resulted from falling cotton prices and dry conditions. In contrast with dairy and cotton, the farm cash incomes of dairy and horticulture varied slightly, while there was no significant changes after 2012-13.

Figure 2.6 Irrigated farm cash income by industry, 2006-07 to 2014-15



Note: 2014–15 data are provisional estimates.
Source: Murray–Darling Basin Irrigation Survey

Source: Ashton & Oliver (2015, p.9)

[2] Farm cash income is the surplus farm-based income available after paying for cash operating costs. It is calculated by using total cash receipts (revenues received by the farm business during the financial year) less total cash costs (payments made by the farm business for materials and services and for permanent and casual hired labour, excluding owner, manager, partner and family labour).

Aither (2021) report a value of the economic value (profit) of floodplain harvesting for agriculture across five valleys (NSW Border Rivers, Gwydir, Namoi, Mcquiarie, Barwon-Darling) for a base case scenario of the next ten years of $524-1023 million. They suggest implementing policy restrictions on harvesting would decrease on farm profit by 14%. Input-output analysis was used by the DPIE to consequently suggest over $7 million 'loss' as a result.

There has also been substantial agricultural structural change in the MDB over the last two or so decades. Generally, water has moved to higher value (or more efficient) uses and provided incentives to increase irrigators' water-use efficiency (e.g., Bjornlund & McKay, 1995; Young et al., 2000). In particular, there has been a decrease in irrigated pasture and broadacre activities, with their water moving to the expanding horticultural industry (Bjornlund & McKay, 1996, NWC, 2012). Although irrigated grape areas have declined since the early 2000s, other horticultural crops, especially almonds and other nuts, have expanded so much as to more than offset the loss of grape area (ABARES, 2016).

However, the changes of agricultural production vary across different MDB regions. For example, there has been a net increase of permanent horticultural plantings in the Lower Murray-Darling of 35,575 ha between 2003 to 2021, from 95,905 ha to 131,480 ha, with almonds now the dominant permanent crop, from 7,330 ha to 45,145 ha (SunRISE, 2022).

2.3.1.3 What is known about the major risks, threats and recoverability of agricultural economic values?

The biggest risk to agriculture values in the Basin is climate change. Many agricultural regions are expected to be adversely affected by climate change: through higher temperatures, less rainfall, and increased incidence of extreme events, such as floods and droughts (IPCC, 2019) and the MDB is not an exception (Qureshi and Whitten, 2014).

2.3.1.3.1 Predicted Basin climate change

Under current climate change predictions, the MDB is expected to become hotter and drier, and experience more frequent extreme weather events such as drought and flooding (Chiew et al., 2011; CSIRO, 2012a; Timbal et al., 2015), significantly impacting water availability (Zhang et al., 2020), water quality and therefore agricultural production (Quiggin et al., 2010; Baldwin, 2021). Earlier projections suggested surface water availability reductions of a median 11% by 2030, under a median climate change scenario. In the presence of climate change induced uncertainty, there is expected to be decreasing volumetric rainfall trends, but also increased variability, leading to severe intensity rainfall events causing flood inundation problems, with the probabilities and risks of extreme events changing (such as extreme rainfall falling within a short space of time) (CSIRO, 2012a).

Table 2.2 MDB farm irrigation water extractions, hectares irrigated, and GVIAP and GVAP in ABS and ABARES data

	ABS data							*ABARES data*[2]		
Year	*Agricultural businesses (no.)*	*Irrigation businesses*	*Area irrigated (ha)*	*Volume applied – including farm irrigation surface, groundwater & floodplain (ML)*	*Extraction rate (ML/irrigated ha)*	*Gross-value Agricultural Production (GVAP) ($ nominal)*[1]	*Gross-value Irrigated Agricultural Production (GVIAP) ($ nominal)*	*Area irrigated (ha)*	*Volume applied – including farm irrigation surface, groundwater & floodplain (ML)*	*Extraction rate (ML/irrigated ha)*
2005-06	61,504	18,674	1,664,000	7,397,678	4.45	14,990,900,000	5,522,000,000	1,560,629	7,286,585	4.67
2006-07	59,864	17,063	1,101,000	4,458,279	4.05	12,739,200,000	4,921,900,000	1,104,124	4,444,954	4.03
2007-08	56,585	15,875	957,753	3,141,659	3.28	15,575,883,630	5,078,900,000	957,300	3,168,242	3.31
2008-09	54,096	15,476	929,074	3,492,409	3.76	14,915,081,563	4,349,100,000	893,213	3,537,813	3.96
2009-10	53,681	15,486	975,660	3,564,480	3.65	14,400,600,000	4,416,620,000	905,758	3,480,959	3.84
2010-11	54,023	15,794	1,194,253	4,518,369	3.78	19,461,800,000	5,944,280,000	1,237,005	4,653,358	3.76
2011-12	53,946	14,684	1,411,612	5,875,449	4.16	18,566,680,000	6,691,450,000	1,424,018	5,929,310	4.16
2012-13	51,203	13,361	1,597,454	8,283,439	5.19	20,542,600,000	6,836,680,000	1,627,977	8,337,703	5.12
2013-14	50,929	14,496	1,559,565	7,736,385	4.96	19,384,218,681	7,135,390,000	1,554,143	7,606,616	4.89
2014-15	49,096	14,587	1,366,738	5,868,785	4.29	20,586,182,070	9,962,104,667	1,364,153	5,930,577	4.35
2015-16	35,465	9,216	1,238,106	4,938,381	3.99	21,772,424,204	6,881,857,920	1,253,008	5,094,660	4.07
2016-17	36,083	9,196	1,347,592	6,355,072	4.72	24,795,414,745	7,195,157,137	1,359,998	6,488,109	4.77
2017-18	35,203	9,496	1,460,054	6,797,678	4.66	23,537,032,948	8,615,437,698	1,496,490	6,999,051	4.68
2018-19	36,590	8,853	1,085,891	4,421,983	4.07	21,457,311,324	7,240,000,000	1,094,250	4,476,222	4.09
2019-20	35,698	7,308	700,997	2,703,741	3.86	21,686,782,368	6,393,000,000	741,622	2,841,425	3.83
2020-21	35,386	8,389	1,170,284	4,843,788	4.14	29,801,840,642	8,395,000,000			

Notes: [1] MDB regions determined by NRM codes illustrated in Table A.1. 2005/06 – 2006/07 values from ABS (2010)
[2] ABARES data in their spreadsheet starts 2006, and we have assumed this is the financial year 2005-06
Sources: compiled from ABS (2010, 2022d, multiple years), Wheeler et al. (2020a), and ABARES (2021a).

Recent studies predict for a 2.2°C global average warming scenario, mean annual runoff in the MDB will reduce by 18-19% for the period 2046-2075, with an estimated reduction of 18% and 19% for the MDB, and Northern Victoria respectively (Zhang et al., 2020). This stems largely from projected drier winters with higher declines in winter rainfall the further south in the Basin. Basin-wide daily mean temperature has increased by 1.0°C for the period 1910-2017, with daily minimum and maximum temperature increasing by 1.3°C and 0.8°C respectively, and the Basin is projected to continue to warm by 0.6-1.5°C by 2030 and 0.9-2.5°C by 2050 (Whetton & Chiew, 2021). Additionally, 3-year hydrological droughts will occur once every 10-15 years under future median climate change projections, and once every 5 years under the dry scenario, as compared to once every 20 years for the historical climate sequence of 1976 – 2005 (Prosser et al., 2021). These figures are of similar magnitude to the ~3,000GL of water returned to the environment under the Basin Plan (Whetton & Chiew, 2021). However, presented predictions may yet underestimate water availability reductions, as climate change projection models commonly lack understanding of long-memory processes, such as multi-annual groundwater level decline, and their impacts on water availability (Fowler et al., 2022).

2.3.1.3.2 Basin-wide climate change impacts on agriculture

Australian agricultural productivity as a whole is projected to decline by 17% by 2050 as a result of climate change (Commonwealth of Australia 2008). However, learnings from the Millennium Drought show that farmers have a variety of tools at their disposal to lessen the impacts of water scarcity and climate change (Wheeler et al., 2013a; Kirby et al., 2014; Kirby et al., 2015; Seidl et al., 2021). For example, water trading in particular could support high agricultural production values by reallocating water from low to high-value crops (Loch et al., 2013; Qureshi et al., 2018). However, while the relative high value of horticultural crops and associated water allocation purchase power will provide some resilience to drought and drier climates, potential plant and crop damage can still occur even for horticulture as a result of water shortages in drought (ABARES, 2020). Thus, irrigated agriculture may be able to tolerate a median climate change scenario as agricultural profits would only decrease by a small amount (Jiang & Grafton, 2012), but an extremely dry (and variable) climate in the future would substantially reduce profits. Furthermore, adaptation to higher intensity floods is more difficult, as water trading does little to mitigate these.

Climate change is also expected to impact the prevalence of crop disease and pests. Noxious weeds, crop disease and pests thriving in warm conditions can extend their range with rising temperatures, towards currently cooler climates. They may also occur earlier in the season than previously (FAO, 2015), putting pressure on MDB agricultural production systems.

Heat stress is another negative impact of climate change on agriculture. Livestock experiencing heat stress has been found to experience significantly higher mortality rates and reduced milk yields (FAO, 2015). Plants are also affected by heat stress: hotter night temperatures have been found to negatively impact rice yield and quality, and extreme daytime temperatures have a strong negative effect on crop yields. Reduced winter/night time chill has negative impacts for many fruit and nut trees, such as almonds or citrus, who rely on a chill event for flowering or fruit set, leading to less and lower quality yields (FAO, 2015).

It has been suggested that the farm profitability cycle observed in the past (3 years profit, one year loss, 4 years breakeven) will change in the future, with increasing risk needing to be managed (Loch et al., 2013).

2.3.1.3.3 Water recovery impacts on irrigated agriculture

The impacts on irrigated farms and their rural communities of returning less than a third of all held consumptive water entitlements in the Basin to environmental use is a highly contested and politicised issue. There has been a considerable amount of economic modelling done to evaluate the impact of water recovery on both farms and communities, with some vastly differing results, depending on the methodology employed. Chapter Five provides a more comprehensive literature review, broken down by type of method, of all these studies, their quality and their results. At one end of the scale, there are a number of consultancy studies (using simple assumptions and discredited methods such as input-output modelling) that suggest that water recovery, and in particular the buyback program, will have significant negative impacts. RMCG (2016) suggested water recovery programs through buyback would double water price in Victoria, and lead to a reduction in the annual farm-gate value of dairy production by $200 million, with mixed farming and cropping sector losing a value of $25 million, coupled with a total loss of $580 million per year and the loss of 1,000 jobs across the region. This report assumed a direct linear relationship between water use and milk production (ignoring surplus water, on-farm resource movements, other adaptation measures). Similarly, Frontier Economics and TC&A (2022) state that if it had not been for buyback (adjusted for trade), then water diversion would have been 46% higher in the GMID, and suggested the average annual costs in lost production would be greater than $400 million per year.

Other studies, using real data and more sophisticated modelling to account for the many different linkages within the economy, find much less impact of water recovery, with small impacts on employment and GDP (Dixon et al., 2009, 2011; Wittwer, 2010; Wittwer, 2011a; 2011b). ABARES modelling found slightly higher negative impacts on rural communities (ABARES, 2010a; 2010b; 2010c), which were still considerably lower than the impacts modelled by input-output studies.

Other applied economic studies using surveys and real data have also found that there is not a direct proportional relationship between reductions in farm water use and farm irrigated hectare production, because of factors such as farmer adaptation, surplus water use, water substitution, water trade and farm restructuring following buyback (e.g., Wheeler et al., 2014, 2014a; Wheeler & Cheesman 2013). It has been shown that actual reductions in Basin wide farm revenue are much less than the reductions of water during drought (Kirby et al., 2014; Connor et al., 2014). These studies find that for every 1% reduction in water, farm production revenue decreased by as little as 0.1% to around 0.6% (Kirby et al., 2014; Adamson et al., 2011; Wittwer & Griffith, 2011).

Hence, there is a significant difference between much work commissioned by irrigator groups, governments and the work done by academics and other research groups. Rural media and communities have a strong belief that buyback has been negative, and often associate it with all things that have gone wrong in their community. It is therefore very important to understand just what study can be relied upon for water policy going forward. To do so, we need to understand the internal and external validity of each study, its methodology and its application. Given the importance of understanding this, this Outlook report undertakes additional analysis to more fully explore and understand the various literature on water recovery in the Basin (Chapters 4 and 5).

To date, there has been no overall, high quality, longitudinal assessment on actual outcomes of water recovery over time on agricultural economic values to be able to draw definitive conclusions from. However, evidence suggests that other factors (such as commodity and input prices, climate variability, technology change) play a much more significant role in influencing economic values than water recovery.

Recoverability aspects

Notwithstanding the substantial challenges agriculturalists in the MDB face, they have a range of adaptation and management options at their disposal when facing water scarcity issues (or to reduce water supply risk) on the farm. Adaptation comes in two main forms: a) incremental adaptation; and b) transformational adaptation. In the irrigation context of the MDB, transformational change may include: a) a complete shift to dry-land operations, and selling all water entitlements; b) large-scale buying of irrigated land and/or water entitlements in a variety of different areas to hedge against declining water allocations and climate risk; c) selling the farm and relocating to an area with more reliable rainfall; and d) leaving farming to take up job opportunities elsewhere.

Incremental adaptation is more related to the adoption of actions that do not require major decisions and or information (Wheeler et al., 2014a). Wheeler et al. (2014a) provides commentary and an overview of the many incremental adaptation measures irrigators can adopt. They include information; trade; agronomy; farm structure; land; infrastructure and environment measures. It has been found that irrigators adopt more strategies (especially water-related strategies) than dryland farmers (Dinh et al., 2017).

Governments can also positively contribute to community and irrigator ability to adapt to water supply issues through policy. To address agricultural and environmental issues linked to limited and, in the future increasingly, variable water supply, water resources management in the MDB is characterised by a myriad of agreements and other initiatives over a long history of water governance (Cummins & Watson, 2012; Quiggin, 2012).

2.3.1.4 What is known or estimated about the future condition of agricultural economic values, especially under climate change?

Overall trends also manifest directly for different crop types or regions in the MDB. Indeed, a recent study found that increases in average growth season temperature negatively affect the price of wine grapes, as quality diminishes with rising temperatures (Puga et al., 2022). In general, the massive expansion of horticultural tree crops in the southern MDB is already predicted to push sectoral water demand to be equal to (ABARES, 2020) or larger than regional water availability in extreme dry periods, benchmarked on historical data like the Millennium Drought. This effect will only become more prevalent under progressing unfavourable climate change.

Using stage-contingent modelling, Adamson et al. (2017) show that with increasing drought frequency and overall decline in water availability, the ability of perennial producers to adapt to water scarcity decreases, and risk management strategies become more expensive. This is especially true for homogenous production systems dominated by perennial crops. Hence, the expansion of permanent horticulture in the Basin can increase the vulnerability of Basin agriculture to drought and climate change and exacerbate decline of future farm profits (Adamson et al., 2017).

Climate change can also negatively impact Basin annual cropping and livestock industries. ABARES (2020) predict a reduction of MDB dairy and rice water use of 55% and 32% for a future dry climate scenario. As the study's climate scenarios are based on historic data, it concedes that model predictions may underestimate reductions in water use and GVIAP in case future climate is substantially drier than historical climate. Lewis et al. (2022) modelled crop mix and crop area responses to climate change by 2060 for the Murrumbidgee Irrigation Area, finding that if current crop mix was maintained, reduced water availability would lead to a reduction of current (2020's base case) planted area by 70%. It would further reduce net revenue from irrigation by 28%, with the most substantial area reductions for canola, pasture, lucerne and cotton. The model also suggests crop mix change from canola and cotton towards higher value vegetable crops can maintain irrigation net revenue for the 2060 scenario. Confirming the negative impacts of climate change on farm profits, Hughes et al. (2022) modelled the farm profits of cropping and livestock farms for predicted temperature and rainfall under different climate change scenarios by 2050. They predicted a reduction of average farm profits of 2-32 % under the median, and 11-50% under the high emissions climate change scenario. For the MDB regions of NSW Riverina, VIC Central North, and SA Murray Lands and Yorke Peninsula, the model predicted respective reductions in average farm profits of -16.2%, -28.1%, and -1.9% for a high emission scenario.

Aither (2020) used a scenario approach to estimate how the consumptive water supply in the sMDB in any given future year will be required by permanent irrigated horticulture and the 'headroom' above that (namely the amount available to other industries). However, the model assumes that there is no reduction in permanent horticultural plantings that may occur because of water availability. Their conclusions were that existing permanent horticulture in the connected Murray region is growing, and will grow from their estimated 1,230 GL per annum to 1,400 GL at full maturity.

2.3.2 Community economic values

2.3.2.1 What is known about the benefits and impact of community economic values?

Data on the community values such as employment and Gross regional product (GRP) can be accessed in ABS population census and other information. GRP is the measure of wealth generated by a region and its overall economic performance. Since 2011, Aither (2022) cites that GRP in Basin communities have trended upwards. However, from 2016 to 2021, about half LGAs in the Basin experienced a decline, while the other half grew their GRP. As the increase of GRP in some LGAs outweighed the decline in other LGAs, the overall GRP in the Basin still increased despite the decrease in GRP in half of LGAs. Local jobs also expanded before 2019.

Other social and economic condition measures such as employment, overall community welling, infrastructure, and services vary substantially across MDB. Sefton et al. (2020) compared Basin communities with the average of all other regions outside major cities in Australia. The results indicated that 64% of MDB people live with economic, employment and standard of living conditions in line with the regional Australia average, while nearly a third in the MDB live below, and 6% live above that average. Compared with economic, employment and standard of living conditions of all regional Australian communities, they found that the conditions of overall community wellbeing and infrastructure and services in the MDB are better. Specifically, 42% of MDB people have higher overall community wellbeing compared to the regional Australian average and 41% of people in MDB have obtained better infrastructure and services than the average. Yet they identified that social and economic conditions in some regions and towns in MDB are poor and trending noticeably downward.

2.3.2.2 What is known about current condition and trend of community economic values?

University of Canberra produces 'wellbeing' estimates for MDB communities from survey data for most years from 2013-2020. Such estimates tend to be variable, and hence there is difficulties in estimating any trend in the data. Schirmer and Mylek (2020) categorised community wellbeing into six dimensions: overall resident ratings of community wellbeing; population size; ageing and health; economy, employment, and standard of living; community and social connection; physical amenity; and access to services and infrastructure. They found that three dimensions out of six in Basin communities are poorer than those outside the Basin: population size, ageing and health; economy, employment and standard of living; and access to services and infrastructure. Only one aspect of community wellbeing in MDB is better than those outside: community and social connection. Their findings also suggested that while inner regional communities in the Basin have similar social and economic conditions overall to inner regional communities outside the Basin, outer regional and remote parts of the Basin are experiencing poorer social and economic conditions compared to outer regional and remote areas in other parts of Australia. Considering the heterogeneity among different areas in MDB, they suggested a strong need to focus on addressing the factors driving poorer social and economic wellbeing in outer regional and remote in MDB communities.

In terms of house prices, the general trend has been upwards over the past decade. In trying to understand the link between environmental assets and house prices, Tapsuwan et al. (2012) applied a hedonic property price model of rural land in a natural resource management region. In this study, environmental assets are described in terms of their recreational attractiveness, based on park facilities and recreational activities. Results show that for a property that is 1 km away from the River Murray, decreasing the distance to 500 m from the river increases the property price by $245,000. This value is magnified by $27,000 if the house is in an area where there is high river recreational attractiveness and drops by $14,000 if river recreational attractiveness is low.

2.3.2.3 What is known about the major risks, threats and recoverability of community economic values?

The socio-economic outcomes in the MDB are shaped by a variety of factors, such as the amount of water available, historical development, and water recovery decisions, drought, floods, and further water recovery, public social investment, education, roads, wifi etc as well as COVID-19, are significant future risks and challenges for the MDB community. Schirmer and Mylek (2020) specified that the drought experienced in many Basin communities has resulted in changes in unemployment rates, financial distress, and labour force participation. Sefton et al. (2020) suggested (without providing direct evidence) that water recovery increased community vulnerability through increased competition for scarce water. They also described that COVID-19 pandemic is another risk to the communities in the MDB as it is expected to have a mixture of short and potentially longer-term consequences. There are possible impacts on the health and social capital of the Basin communities, markets and the demand for food and fibre, supply chains of food, and other services and businesses.

Given the desire to try to offset any negative impacts of water recovery on rural and regional communities in the MDB, there are a variety of different regional development funds and programs. Sefton et al. (2020) report that the Commonwealth government undertakes substantial support and economic stimulus activities in MDB communities, such as the Building Better Regions Fund ($842 million, closed December 2019), the Community Development Grants Programme ($980 million, 2013-2016), the National Stronger Regions Fund ($1 billion, 2015-2020), the Bridges Renewal Program ($640 million, 2015-2023), drought and farm support schemes (> $100 million, yearly), and concessional loans to farmers. These are complemented by Basin state regional development schemes, such as the NSW Regional Growth Fund ($3 billion, by 2021), the Victorian Regional Jobs and Infrastructure Fund (> $500 million to date, ongoing), the Queensland Jobs and Regional Growth Fund ($175 million, ongoing), and the SA Regional Growth Fund ($150 million, 2018-2018). As such, Basin communities receive disproportionately more government development and stimulus funding per capita, than the average Australian (Sefton et al., 2020). The Productivity Commission (2018) found that the $189 million provided to Basin communities by 30 September 2018, for means of structural adjustment to reduced water availability have not been effective in helping communities to adjust to the Basin Plan: assistance was not targeted to the most vulnerable Basin communities, and some funded projects failed to provide community assistance. These structural adjustment programs include the MBD Regional Diversification und ($100 million, 2013-2019), the Strengthening Basin Communities Fund ($64 million, 2009-2011), and the SA River Murray Sustainability Program ($25 million for regional development, $120 million for industry assistance).

Wittwer and Young's (2020) work using TERM-H2O to model two scenarios: 1) obtaining the remaining water recovery target through infrastructure only; and 2) removing the same amount of water entitlements and spending on regional services instead between 2020 and 2024; found that scenario one had a net present value (NPV) welfare loss of $1.1 billion (but increased jobs up to 1,000 in the short-term, and 100 in the medium-term), while scenario two found that each dollar spent on education, health and community services created four times as many jobs as spending on infrastructure, and had a NPV welfare loss of $0.125 billion (nine times less than spending on infrastructure). Note, no welfare benefits of increased environmental water were allowed for. This emphasised that community economic values depend a lot more on essential social and economic services than water recovery in general.

In summary, substantial funds are channelled into Basin communities for structural adjustment and economic stimulus. There is a very large risk that such funds are not well spent, and do not have the desired impacts.

2.3.2.4 What is known or estimated about the future condition of community economic values, especially under climate change?

Future community values will be dependent somewhat on agricultural outcomes, and will also be highly specific to regions and various rural towns. The known potential impacts on jobs, GDP have been modelled in various ways, and were reported in the section on agricultural economic values and are not elaborated further here.

2.3.3 Recreational, fishing and tourism economic values

2.3.3.1 What is known about the benefits and impact of recreational, fishing and tourism economic values?

Recreation includes skiing, fishing, water-based and river-side activities. Recreational sites in regional and rural areas provide important opportunities for tourism and hospitality which can mean a local waterway, or a waterbody can become the lifeblood of a community. Those sites have the ability to attract and retain people in the area through better amenity, social and recreational opportunities and providing an income for local businesses (e.g., DELWP, 2019; DELWP, 2020).

Surveys and data collected from *Tourism Research Australia* (TRA) report various categories of overnight trips, day trips, tourism consumption, tourism gross value added, tourism gross regional product (see below for definition industries), and tourism satisfaction surveys. Some data are at the tourism region level. TRA provides 18 different categories of information.

The Murray is provided as a separate region (and is broken down into Murray East and Murray River, Lakes and Coorong). All estimates suggest that tourism and recreation grew progressively in the MDB over the years (Aither, 2022).

The ABS has information on tourist accommodation in Australia, from the Survey of Tourist Accommodation, which is a census of all in-scope accommodation establishments. Data are reported quarterly. ASGS SA2 area statistics are available for 2015-16 and can be used to calculate MDB figures. SA2s are also aggregated to tourism regions as defined by relevant state and territory tourism organisations. Historical data are available back until the 1970s. However, accommodation statistics do not differentiate tourism and business travel.

ABS Microdata (BLADE, The Business Longitudinal Analysis Data Environment) also provides business net income for tourist accommodation providers, recreational activity providers, and travel agencies from 2001-02 to 2020-21, on the mesh block level. Yet, business receipts do not differentiate between tourism and business travel.

2.3.3.2 What is known about current condition and trend of recreational, fishing and tourism economic values?

Dyack et al. (2007) provided an exploratory economic analysis of the recreational values that visitors hold for their visits to the Coorong in SA and the Barmah Forest in Victoria based on two surveys of recreational visitors from 2006. In this study, recreational values were estimated using three different methods (values are expressed in terms of consumer surplus): the travel cost method (TCM) and the contingent valuation method (CVM) to estimate the consumer surplus of recreation trips, and the contingent behaviour (CB) approach to estimate the responsiveness of visits (and the change in value) to changes in access. Travel costs and time are considered as investments in the TCM which are then used to evaluate the value of the recreational experience (i.e. revealed preference technique). On the other hand, surveys were used directly asking about the value of recreational experiences or tourists' response to potential changes in condition for the CVM and CB approach (i.e., stated preference techniques). The TCM results showed that the average non-market value of visiting was $134 per adult per day for Barmah Forest and $218 per adult per day for the Coorong. Considering the different time spent, results showed a total non-market recreational value per adult per trip of $529 for Barmah Forest and $503 for the Coorong. Results using CVM for the

Coorong were half of the TCM results potentially reflecting the different context of the questions.

Rolfe and Dyack (2011) estimated recreational values of the Coorong, SA, based on count data models extended with contingent behaviour (CB) questions. Results show recreational values at $111 per adult visitor per day, or $242 per trip, based on 120,000 visitors; and a consumer surplus of $30.5 million per year. Furthermore, random coefficient negative binomial models showed that a marginal value of at least $17.20 per person per trip is attached to each 1 per cent change in site access. The study concludes that recreational values are sensitive to different conditions of the site, which should be accounted for when evaluating potential management options.

Heagney et al. (2019) estimated economic value of tourism and recreation across protected areas in NSW using data from a stratified random phone-survey of more than 62,000 individuals. This study applied random effects ordered logit models which estimated the value of tourism and recreation services at $3.3 billion per annum. The comparative values from this study may indicate that the recreational services provided by protected areas can be a similar order of magnitude to extractive uses. It is recommended that land use decisions should consider these values to optimise societal benefits from land use allocation.

A different study assessed the economic value of ***recreational boating*** across different MDB sites based on values for gross output and gross value-added (MJA, 2019). Based on broad Tourism Research Australia data the gross output of recreational boating was estimated at $350 million and gross value-added at $300 million per year. The study further suggested that the contribution of recreational boating is not significantly affected by changes to water availability, e.g. from increased environmental flows (except during very low flow events).

MDBC (2006) further highlighted the importance of the availability of water for ***boating*** as an important ecosystem service and estimated that about 6,800 watercrafts were in regular use on the lakes and Coorong in 2001 (120,000 user days per year). The gross economic value of recreational boating was estimated at $14 million per year with around 140 employed people (Helicon, 2004).

The value of ***duck hunting*** in the Coorong region was estimated at $42-$62 per hunter per shoot (using a travel cost survey), and provided just over $1 million per year in net present value (2000) for wetlands in south-east SA for about 500-1500 hunters (Whitten & Bennett, 2002). On the other hand, based on choice modelling, the conservation costs to society of duck hunting were calculated to be over three times greater than the benefits to hunters (Bennett & Whitten, 2003).

Crossman et al. (2014) identified further important activities at the Coorong and Lakes, specifically ***passive activities***, such as enjoyment of the scenery, nature-based recreation and relaxation and learning. ***Four-wheel driving*** is also a popular activity at relevant beach sites, however, Paton (2009) discussed potential trade-offs of sand compaction, erosion due to destruction of surface salt crusts, and mortality of hooded plovers that nest in the dunes.

Australia’s mainland alpine and ski industry are located in the headwaters of the MDB in NSW and Victoria and (partially) include some MDB communities. NSW has seven, and Victoria six alpine ski resorts, attracting tourists for winter sports and summer mountaineering activities (ARCC, 2022; DPE, 2022). However, winter visitations are about double than summer season visits (ARCC, 2021). Victorian resort visitor data is available from 1985 onwards through annual seasonal visitor reports by the Alpine Resorts Co-ordinating Council, with 1,185,000 winter season visitors in 1985 and 741,000 in 2021, with annual average winter visitors 1,231,000 from 1985-2021 (ARCC, 2022). Ski resort visits contributed between $790 million in 2016 and $1,061 million in 2019 to the Victorian gross state product and employed 7,892 and 9,866 people respectively[3] (ARCC, 2021). In NSW, ski resorts are located in the Snowy Mountains Special Activation Precinct, with an estimated 3,200 people employed all year, an additional winter seasonal workforce of 3,265, and $520 million in total gross value added to the NSW state economy in 2017/18. Estimated winter visitor numbers for 2014/15 -2017/18 were 373,000, contributing an estimated $329 million in regional expenditure (CIE, 2021).

Recreational and commercial fishing are some of the most important ecosystem services for human well-being based on income generation (Colloff et al., 2015). The NSW Department of Primary Industries (2018) publishes the *NSW Recreational Fishing Survey* which includes questions relevant for a TCM study. The *NSW Aquatic Ecosystem Research Database* contains site-specific data and information from fish-related projects. The *NSW Recreational Fisher Licence Database* includes contact and license information for fishers who purchased a licence from licence agents or via electronic methods (NSW Department of Primary Industries, 2018). NSW Department of Primary Industries (2018) also developed a valley-scale assessment methodology for valuing recreational fishing in the MDB.

McIlgorm and Pepperell (2013) estimated the economic contribution of recreational fishing in NSW using a telephone survey that collected information on related activities and expenditures of recreational fishers in 2012. Expenditure and economic impact of recreational fishing was provided for NSW overall and for four regions, including the “Inland” region relating to the NSW part of the MDB. The economic output for recreational fishing in “Inland NSW” was $353.81 million with an associated employment of 1,539 equivalent full-time jobs. The economic output analysis was based on a Simulating Impacts on Regional Economies (SIRE) input-output model of the respective regions. The economic impact of the “Inland” region in absolute and relative terms was smaller when compared with the coastal NSW regions, as saltwater-fishing was related to greater expenditures.

Deloitte Access Economics (2012) estimated that the overall value of the fishing industry (recreational and commercial) increases by $28 million per annum (2.7% increase) and the consumer surplus of recreational fishing increases by $9.1 million per annum for in the MDB after the Basin Plan is fully implemented, and the ecological response function has fully occurred by 2020. Estimates were based on other expenditure studies. However, at that stage the link between the Basin Plan and the fishing industries was unclear and under-researched; thus, estimates were surrounded by a significant level of uncertainty.

West et al. (2015) used a regionally stratified random telephone survey of 9,400 NSW/ACT households to explore recreational fishing participation, associated catch and expended effort, boat ownership levels and recreational fishers’ attitudes to various topics. The survey found that an estimated 849,249 NSW/ACT residents participated in recreational fishing from June 2012-May 2013, representing a participation rate of 11.9 %. Fishers spent an average 4.3 days per year fishing, with 21% of fishing activity occurring in freshwater of which more than 50% took place in rivers (including the Murray and Darling). Recreational fishers had higher levels of boat ownership (38%) than the general NSW/ACT population (11%), leading to an estimated market value of the recreational fishing fleet of ~$1.534 billion in May 2014. Fishing provides important recreational and community benefits to stakeholders, as attitudinal survey participants indicated *being outdoors*, *enjoyment of fishing*, *spending time with the family*, and *spending time with friends* as the most important motivations to engage in fishing activities. In particular, ~20% of residents in the south-west fishing region (which

[3] The contributions in 2020 were $109 million for the gross state product and employment of 960 people, likely due to the impact of Covid and associated ski resort closures.

includes the River Murray and Murrumbidgee areas) participated in recreational fishing, as compared to 11% for the general population. This region also accounted for the majority of fishing days for the inland fishing zones, followed by the Darling/North West region.

Using publicly available data, stakeholder interviews and modelling, MJA (2020) concluded that recreational fishing in the MDB provides an estimated baseline economic contribution of \$100 million gross output and \$90 million gross value-added per year.

Hardaker et al. (2020) provided an analysis on the impact costs of carp and expected benefits and costs associated with carp control in the MDB. Based on a survey, choice modelling (CM) was conducted to estimate community willingness to pay (WTP) for the potential environmental outcomes of reduced Carp numbers to provide data on the non-market benefits and costs for the overall CBA. Non-market costs were estimated based on a per-household WTP for changes in particular environmental outcomes (native fish, native waterbirds, and area of healthy wetlands) over 10 years' time following Carp suppression resulting in the following range of possible total WTP: \$24,372 - \$2.08 billion for fish, \$39,187 - \$313.5 million for wetlands, and \$5,422 - \$601.8 million for water birds (specific values depend on the extent of environmental recovery forecast).

BDO EconSearch (2021) provided a comprehensive analysis of various economic indicators for the commercial fisheries of the lakes and the Coorong region. Estimations use various databases, such as licence holder surveys (every 3 years in each fishery), SARDI catch and effort and GVP data, PIRSA cost of management and quota transfer data and other primary and secondary data sources. Despite some declines in the real value of the Lakes and Coorong Fishery between 2000/01 and 2019/20 the overall trend is increasing over the 20-year period. The average income per licence holder fell to \$206,000 in 2009/10 and recovered to \$536,000 in 2019/20.

2.3.3.3 What is known about the major risks, threats and recoverability of recreational, fishing and tourism economic values?

Risks to recreational, fishing and tourism economic values include flooding, water quality issues, climate change, drought, poor quality infrastructure, and environmental degradation (Hatton Macdonald et al., 2011a). Economic values are dependent on the viability of the fish or water source - hence if stock is declining or river quality unfit for recreation, then tourism and recreation will correspondingly decline.

For example, studies have found that maximum snow depths in the Australian alpine regions have declined, and the snow season finishing earlier due to increasing temperatures, with these trends expected to continue under climate change (Ruddell et al., 1990; Bhend et al., 2012; CSIRO & BOM, 2015). Projections show that average temperature across the Australian Alps could increase by 4-5°C by 2070-2099, lowest winter temperatures rise by 2.5-7°C, with annual snowfall declining by 60-80% leading to only the highest peaks experiencing any snow fall. As climate change progresses, ski resorts increasingly rely on snow-making technologies to sustain themselves, with associated costs expected to rise as natural snow cover declines and water and electricity costs increase, limiting the economic viability of snow-making beyond the 2030s (Harris et al., 2016). As such, the ski industry in Australia faces existential decline by the end of century, significantly reducing its economic and employment contributions to Australia and the MDB communities.

2.3.3.4 What is known or estimated about the future condition of recreational, fishing and tourism economic values, especially under climate change?

As above, future recreational, fishing and tourism values will be dependent upon the availability of fish, river quality (and quantity), infrastructure and surrounding environment. Colloff et al. (2015) highlight that under projected climate changes, e.g., declining river flows, increasing salinity levels, major shifts in the nature and extent of supply of ecosystem services are likely. It is important to identify those ecosystem services that can continue to be supplied or require better management under climate change to ensure supply and the successful adaptation to climate change for those communities that depend upon those services for their livelihoods.

2.3.4 Mining and energy economic values

2.3.4.1 What is known about the benefits and impact of mining and energy economic values?

The MDB is home to a number of power generation and mining operations, supporting local economies and communities. The following section outlines both the benefits and costs associated with these industries. Data on mining and energy generation in the MDB is very scarce: it is limited to spatial maps of mineral deposits and power generators by state, ABS data on annual overall sectorial employment, revenues, and water use for Australia and the MDB states, state government data on annual mineral extraction and associated royalties, and some limited private industry reports which utilise company surveys to explore employment and value added. There is an absence of research reports dedicated to the economics of mining and energy generation in the MDB; the peer-reviewed literature is limited to qualitative case study explorations of community and environmental impacts of mining and hydroelectricity generation in MDB regions, without attempting to estimate associated monetary values. Overall data availability is disjointed, and at scales which do not readily translate to MDB boundaries. Hence, using state or region level data which encompasses (some of) MDB regions is often the closest approximation to MDB basin scales.

Senior et al. (2021), using data from existing mines and Australia's National Classification System for Identified Mineral Resources, show the location and size of know mineral deposits across Australia, indicating that the MDB is home to significant mineral resource deposits (defined in kilo tonnes deposit size) and extraction operations, which include black coal, brown coal, cobalt, copper, gold, iron ore, lead, zinc, silver, mineral sands, platinum, rare earths, and tin. Most of these minerals are concentrated in Queensland, NSW and to a lesser extent Victoria, which encompasses gold and substantial (unexplored) brown coal reserves. Available data on mining (costs and benefits) in the MDB is very poor, since mining statistics often exclude monetary values, and are collected at the national, state or regional scale, not the NRM or SLA scale which would allow for a relation to MDB areas. Furthermore, data is fragmented, difficult to source and access, and requires collation from multiple public and industry sources that were not designed to be combined.

Benefits from mining are largely economic and financial, i.e. (export) earnings from mining, direct expenditure in the community, and employment in mining operations.

In 2009, a Senate inquiry explored the impacts of the current and projected mining operations on environmental values in the MDB, and particularly on the Namoi Valley (NSW) and Darling Downs (QLD) catchment. This also included the impacts on agricultural productivity (The Senate, 2009). The inquiry focused on the impacts of coal mining and coal seam gas extraction as the matters of largest public interest. The report states that NSW had seven major coal mining operations in the MDB, in the western coal fields near Mudgee, and in the Gunnedah Basin, whereas QLD mines coal in the Surat Basin of southern Queensland. Based on submission data, coal makes a substantial contribution to state economies: The value of the NSW coal production was estimated at $9.8 billion in 2007/08; with royalties from mining in the NSW MDB around $174 million in 2008/09. Queensland exported $153.36 million of coal in 2007/08, and received around 20% of its Gross State Product from the resources sector. Queensland saw a substantial expansion in its coal seam gas production to approximately 600 wells in 2007/08, with the Surat Basin (combined with the Bowen Basin) contributing 80% of Queensland's natural gas production (The Senate, 2009).

Similar to mining, energy production takes place in MDB regions. However, relevant data is not at the Basin scale, but rather at the state or regional level. There are two broad sources of energy production in the basin: fossil resources, such as coal and coal seam gas, and renewable sources, such as wind, solar, and hydro-electric generation. Benefits from energy generation are largely monetary and through employment, albeit the impact of renewable energies on green-house gas emission and decarbonising the economy are positive but hard to quantify. Employment and revenue data for the energy industry is only available on the Australian scale and does not distinguish between fossil and renewable energy: in 2020/21, electricity supply employed 48,000 people, paid $6,481 million in wages and salaries, and had EBITDA of $26,095 million (ABS, 2022a), using 46,418,778 ML of water (ABS, 2022b).

Energy production from fossil resources is intimately linked with mining in the MDB, and therefore only relevant for NSW and QLD. NSW generated 69.7% of its 2021 electricity production from coal (DCCEEW, 2022a), and coal and coal seam gas play an important role for Queensland's electricity generation, with 65.1% and 14.2 % of electricity in 2021 generated from coal and coal seam gas respectively (DCCEEW, 2022a). In 2020/21, electricity and gas supply industries used 68,546 ML and 63,343 ML of water in NSW and Queensland respectively (ABS, 2022b). Some of Queensland's largest power stations are encompassed by the MDB, such as the coal-fired power plants of Millmeran and Kogan creek (852 MW and 744 MW), and the gas-powered plants of Darling Downs and Condamine (643 MW and 144 MW) (Business Queensland, 2020). In contrast, NSW's coal and gas-fired power stations are outside Basin boundaries (AEMO, 2022b), and therefore do not contribute to Basin communities.

2.3.4.2 What is known about current condition and trend of mining and energy economic values?

The ABS reports annual employment, wages and salaries, and sales and services income from the mining sector on the state level in its Australian Industry series, from 1998/1999 – 2020/21. This data also includes income and expenses, operating profits, earnings before interest, tax, depreciation and amortisation (EBITDA) and industry value added for different types of mining[4], such as coal, or oil and gas, but only at the national scale (ABS, 2022a). A general trend since 2011-12 is that employment in mining has been falling across NSW, QLD and VIC, while sales and services income, wages and salaries paid have been stable or increased. Thus, it is challenging to relate this to the MDB scale, beyond the fact that of the employment (NSW: 22,906; VIC: 5,615; QLD: 35,930), wages (NSW: $3,361 million; VIC:

[4] Coal mining, oil and gas extraction, metal ore mining, non-metallic mineral mining and quarrying, and exploration and other mining support services.

$890 million; QLD: $5,384 million), and sales and services income (NSW: $29,162 million; VIC: $7,294 million; QLD: $55,678 million) reported for 2020/21 (ABS, 2022a), a substantial fraction could be attributed to the MDB.

Mining is a major industry in the QLD part of the nMDB, with major extraction of coal, coal seam gas and oil from the Bowen Basin and Surat Basin fairways around Chinchilla, Roma, and Surat, providing 90% of Queensland's coal seam gas production. There is also oil and gas extraction around Roma from mixed oil and gas fields (DNRME, 2019). Coal extraction in the southern coal statistical region, which includes some nMDB areas, is reported for the period 2015/16 – 2021/22, and was a total of 41,235,814 tonnes for thermal and coking coal combined (DNRME, 2022a). 6-monthly coal seam gas, oil, and gas extraction data is available from the Queensland Department of Natural Resources, Mines and Energy, from December 2004 – December 2020. In the 6 months between June–December 2020, 16,191 million m^3 of coal seam gas and 9,000 barrels of oil were extracted from the Surat Basin, using 21,456 ML of water. For the Bowen Basin over the same time period 4,335 million m^3 of coal seam gas and 25,000 barrels of oil were extracted using 5,082 ML of water (DNRME, 2022b). These extraction numbers have increased continuously over the observation period, especially for coal seam gas, but the economic value of extraction is not publicly reported. In 2020/21 the Queensland government received $2,038 million in coal and petroleum/natural gas royalties (Queensland Treasury, 2022), and mining contributed $27,386 million of the $342,931 million QLD gross state product in 2020/21 (Queensland Government Statistician's Office, 2021).

In NSW, mining for petroleum and gas is concentrated in the Gunnedah Basin which overlaps with the Namoi catchment. Coal is also extracted, but the more substantial coal mining takes place in the Upper Hunter Catchment and around Newcastle, which is outside of the MDB (Department of Regional NSW, 2021a). According to ABS (2022b), coal mining in NSW used a total of 99,488 ML in 2020/21, a slight decrease from previous years' figures which oscillated around the 115,000 ML mark since 2014/15. The NSW government publishes spatial maps to display coal, gas, petroleum and mineral extraction licences, areas, and mines in the state (Department of Regional NSW, 2021a), and also provides information on mining royalties from 2004/05 (Department of Regional NSW, 2022). Royalties from mining have followed an upward trajectory from $396.37 million in 2004/05 to $1,687.44 million in 2019/20, with 90% coming from coal (Department of Regional NSW, 2022). Mining provides significant employment in NSW, with 28,800 jobs directly, and 115,000 jobs indirectly related to mining, as of December 2020. There were 4.6 petajoules of natural gas and 200 mega tonnes of coal extracted in 2019/20, with the majority of coal going into export (Department of Regional NSW, 2021b). Additional to government reporting, the NSW Minerals Council publishes its NSW Mining Industry Expenditure Impact Survey series annually since 2014/15, with the latest report available for 2020/21 (Lawrence Consulting, 2022a). These reports assess the regional economic impacts of mining, but the regions used do not align with MDB boundaries. Using regions that encompass MDB areas as a proxy[5], mining employed 7,340 people, and spent $2.36 billion, supporting 2,219 local supplies in 2020/21. Using input-output modelling of 28 case study mining companies, mining's gross value added is assessed, contributing an estimated $5.3 billion to NSW MDB regions' total gross value added (Lawrence Consulting, 2022a).

Water use for mining activities in Victoria was 17,909 ML in 2020/21, a decrease from figures in previous years since 2014/15, which was in the 20,000 – 25,000 ML range (ABS, 2022b). The Victorian government publishes interactive spatial maps for all extractive

[5] In this case: Central West, Far West, Murray, Murrumbidgee, North Western, and Northern.

industries (DJPR, 2021a), and annual statistical reports on extraction by its mining industry, the Earth Resources Regulation Annual Statistical Report series, available from 2000/01 – 2020/21, with gold as the major mining activity in the MDB regions of northern Victoria (DJPR, 2022). Expenditure by gold mining has steadily increased over recent years, from $280 million in 2015/16 to $401 million in 2020/21, reflective in increasing production, from 200,872 ounces to 722,239 ounces, a production value of $284.7 million and $1,781.5 million respectively (DJPR, 2021b). Gold mining created significant royalty income for the Victorian government to the tune of $48 million in 2020/21, with only coal royalties larger at $82.8 million. Yet, as compared to Queensland and NSW, mining is much less important for the state economy in Victoria, as can be seen by the comparatively modest royalties and production values (DJPR, 2021b). A Victorian mining industry commissioned study explored the contributions of mining to the Victorian economy in a similar format as Lawrence Consulting (2022a). This was again done on a regional scale using economic and expenditure survey data of five mining companies, with regions not aligning with MDB boundaries. Therefore, assessing the mining contributions to VIC MDB uses regions encompassing the MDB as a proxy[6]. In 2020/21, mining employed 810 people, included $169.2 million in direct spending, and supported 602 local suppliers. Input-output modelling assessed the VIC MDB regions' mining gross value added as $316.9 million (Lawrence Consulting, 2022b).

2.3.4.3 What is known about the major risks, threats and recoverability of mining and energy economic values?

Mining in the MDB is controversial, especially in agricultural regions and for coal seam gas or open cast mining operations. Obviously issues of greenhouse gas emissions are critical, but other concerns are centred around coal seam gas and water and coal issues, largely from agriculturalists and rural community representatives. In particular, coal seam gas extraction was opposed on grounds of negative impacts on agriculture, stemming from soil and water quality degradation, and groundwater aquifer deformation (The Senate, 2009).

These concerns are also reported in a number of studies exploring the impacts of coal and coal seam gas mining through drawing on qualitative case studies from MDB regions, namely the Bowen, Surat, and Gunnedah Basin, the Darling Downs, and the Western MDB (Comino et al., 2013; de Rijke, 2013; Mercer et al., 2014). Franks et al. (2010) in studying the Hunter Valley, the Bowen and Gunnedah Basin, found that while some communities may enjoy benefits from mining, cumulative impacts of resource extraction can interact with impacts from non-mining activities to exacerbate stresses for social and environmental systems, including for non-mining communities. Examples include environmental pollution (greenhouse gases, dust, noise, vibration, salinity, water quality), ground subsidence, accommodation shortage and rising costs, skill shortages and staff retention issues, and pressure on community services due to the high population of fly-in, fly-out workers. In their study of the Western MDB, Moran et al. (2013), drawing on regional ABS census and CSIRO National Land and Water Resources Audit data, observed that mining can impact the social environment of communities and displace or compete with existing agriculture: mine employees have little time to participate in community activities, as evidenced by lower volunteering rates of mine workers as compared to pastoralists and the general populous. There was also loss of pastoralist area through open cut mining and associated infrastructure (e.g., roads), and competition for local water resources as it is needed for mining and grazing. This is supported by Comino et al. (2013) who found water extraction from coal seam gas

[6] In this case Mallee, Central Victoria/Loddon Murray, and Hume/ Northeast Victoria.

mining in Queensland is often uncapped, in contrast to agricultural and other water users, creating disadvantages for established water users in mining districts.

Renewable energy production has been of great importance for NSW and VIC MDB communities through the Snowy-Hydro Electricity Scheme in the headwaters of the River Murray. With earliest construction in 1955, the Snowy-Hydro Scheme now consists of nine power stations with a generation capacity of 4,100 MW and is since 2018 fully owned by the Commonwealth (Snowy Hydro, 2022a). In 2020/21, Snowy Hydro made $577 million EBITDA and employed 1,743 people, 60% of which live in VIC and 30% in NSW. (Snowyhydro, 2021). The scheme is currently extended to include 2,000 MW of pumped hydropower, Snowy 2.0, at an estimated cost of up to $4,500 million (Snowy Hydro, 2022b). However, this extension has drawn substantial criticism for cost blow-outs, failure to deliver on the promise of more sustainable and cheaper energy, and for not adequately managing the negative impacts of construction on alpine environments (Normyle & Pittock, 2020; Woodley, 2022).

Further to hydroelectricity, other renewable energy sources, such as wind and solar, have become increasingly important for MDB communities. In 2020/21, NSW increased its wind and solar power generation to 4,805.9 GWh and 8,260.9 GWh, respectively. For Victoria, it was 4,631.9 GWh for solar, and for QLD it was 8,556.7 GWh for solar. Much of this additional capacity across all the three states was installed in MDB regions (AEMO, 2022b, 2022a, 2022c; DCCEEW, 2022b).[7]

The social/environmental costs of mining and energy generation are linked to pollution stemming from fossil fuels, climate change, environmental impacts in general, and water consumption. Many of these costs are largely environmental and unquantified. In particular, negative health impacts from coal and gas-fired electricity generation are widely documented (López et al., 2005; AlRafea et al., 2016; Finkelman et al., 2021; Nelson, 2019), but not for MDB regions. Infrastructure corridors for power lines and access roads, necessary for fossil and renewable energy generation, can negatively impact wildlife habitat (Andrews, 1990). Even renewable energies such as wind power or hydropower can have negative environmental impacts, such as noise pollution, bird, and bat strikes (Wang & Wang, 2015), and runoff and aquatic habitat changes, erosion, and introduction of exotic species and pests (Lawrence, 2001; Normyle & Pittock, 2020).

2.3.4.4 What is known or estimated about the future condition of mining and energy economic values, especially under climate change?

The social/environmental costs of mining and energy generation are linked to pollution stemming from fossil fuels, climate change, environmental impacts in general, and water consumption. Many of these costs are largely environmental and unquantified. Greater quantification and social pressure will lead to growing financial costs for non-renewable mining and energy activities, reducing their future viability.

[7] Renewable power generation in SA does not take place in MDB regions; wind power generation in VIC and QLD takes place outside MDB regions

2.3.5 Water market economic values

2.3.5.1 What is known about the benefits and impact of water market economic values?

Informal water ‘swapping’ markets started in the MDB in the 1940s, with greater establishment of other informal and early formal MDB markets for temporary water from the 1960s onwards (Wheeler et al., 2014b). Further reform and the unbundling of land and water rights in the 2000s led to the development of formal water markets in the Basin (Wheeler, 2014). There are both regulated and unregulated water licenses in Australia. Regulated water has different levels of reliability (namely high, general, and low security) by area. Unregulated systems have no formal reliability, and they are usually determined by restrictions on extraction. To date, most water trade has been in regulated water leases in the southern MDB. For further information on MDB water markets, Wheeler (2022) provides a detailed literature review.

Water markets transact different property rights to water, codified in state and federal legislation. Various types of water property rights exist in the MDB: 1) ***water access rights*** (i.e., right to take/hold water from a water resource); 2) ***water delivery rights*** (i.e., right to have water delivered); and 3) ***irrigation rights***. Water allocations are seasonally announced as a percentage of their access entitlement depending on the water availability in the specific water resource to prevent water over-allocation (NWC, 2011). These rights allow for three broad types of water trading: i) short-term or temporary transfers of water (known as water allocation trade); ii) medium-term leasing of water allocations to secure access to water for a period of time specified in a contract (known as water leasing); and iii) permanent transfers of water entitlements – the on-going property right to either a proportion or fixed quantity of the available water at a given source (known as water entitlement trading) (Wheeler & Garrick, 2020).

Each state introduced individual legislative and administrative processes (water trading regulations) depending on the individual historical developments in water resources management and the characteristics of the water resources and water demand. For example, each state adopted their own terms to describe water access entitlements, water allocations and water delivery rights. As a result of states’ individual historical water legislation processes, there are now over 150 different water entitlement types in existence in the MDB (MDBA, 2019).

2.3.5.2 What is known about current condition and trend of water market economic values?

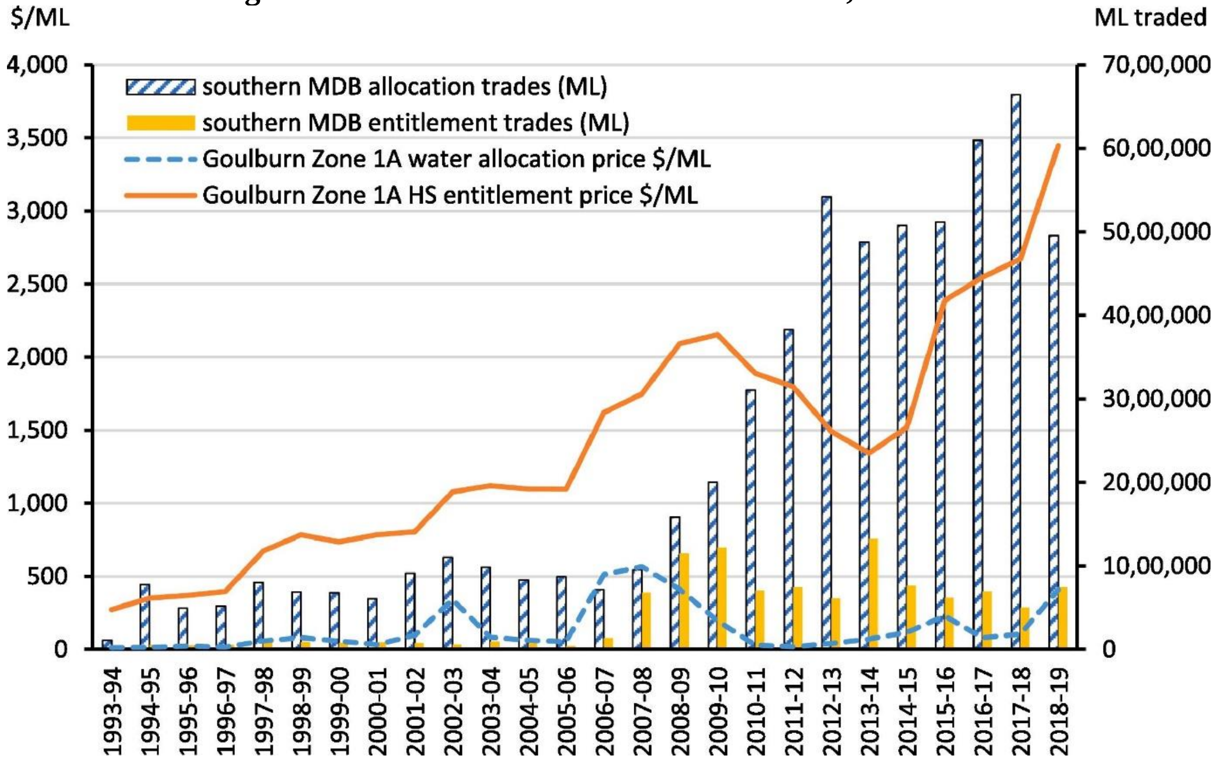
The markets for allocation and entitlements are fundamentally different. Demand for water allocation is highly seasonal and strongly influenced by short-term climate drivers. Consequently, water allocation prices (generally) display an inverse, sometimes lagged, relationship with water available in MDB storages as illustrated in **Error! Reference source not found.**. This figure highlights the positive trend in water entitlement prices over time, while allocation prices are a lot more variable. Trade volumes in both markets have also increased.

In contrast, water entitlement markets are more influenced by strategic and long-term factors such as water supply risk management considerations and agricultural industry change (Seidl et al., 2020b). Overall, water entitlement prices have substantially increased across the southern MDB over the last 20 years albeit with fairly constant trade volumes, whereas water allocation trade volumes have been increasing, but allocation prices have been more variable.

Table 2.3 illustrates a summary of the economic values associated with water allocation trade in the Basin during 2020-21, which have grown substantially over the last decade (BoM, 2022a). A record volume of allocations was traded in 2020–21, increasing by 27% from the previous year, although the number of transactions was similar.

Figure 2.7 illustrates the growth in water market trade in the sMDB over time, both in water allocations and water entitlements. Water allocation volumes have consistently grown, while water entitlement volumes have been more variable.

Figure 2.7 Southern MDB water market trade, 1993-2019



Source: Seidl et al. (2020b)

Regarding volumes of water entitlements traded, 2,547 gigalitres (GL) were traded nationally in 2020–21, a 30% increase compared to the previous year (Table 2.4 Water entitlement trade summary 2020-21 Once again, this record high volume of entitlement trades was primarily driven by increased trade in the southern MDB (BoM, 2022a).

Table 2.3 Water allocation trade summary 2020-21

Region	Resource Type	Transactions	Trades with market rate price reported[1] (%)	Volume (GL)	Estimated Turnover[2] ($M)
Southern MDB	Surface Water	29,890	57	7,267	469
Northern MDB	Surface Water	1,035	34	350	30
Groundwater MDB	Groundwater	718	53	172	17
Rest of Australia	Surface Water	2,023	9	186	3
	Groundwater	286	19	16	1
Australia – total	Surface and Groundwater	33,952	53	7,991	520

[1] Allocation trade market rate price involved transactions with a reported price above $5/ML and below $10,000/ML.
[2] Price data have been cleansed to remove zero prices and outliers that are unlikely to be valid (see BoM, 2022a for details)
Source: BoM (2022a)

Table 2.4 Water entitlement trade summary 2020-21

Region	Resource Type	Transactions	Trades with market rate price reported[1] (%)	Volume (GL)	Estimated Turnover[2] ($M)
Southern MDB	Surface Water	3,836	51	1,662	3,930
Northern MDB	Surface Water	448	41	174	450
Groundwater MDB	Groundwater	827	36	170	260
Rest of Australia	Surface Water	2,867	22	316	670
	Groundwater	1,619	10	225	290
Australia–- total	Surface and Groundwater	9,597	34	2.547	5,600

[1] Entitlement trade market rate price involved transactions with a reported price above $50/ML and below $20 000/ML.
[2] Price data have been cleansed to remove zero prices and outliers that are unlikely to be valid (see BoM, 2022a for details).
Source: BoM (2022a)

In terms of monetary value of water traded, water markets in Australia had an estimated turnover of AUD $6 billion in 2020-21, down from a record high of AUD $7 billion in 2019–20. With increased rainfall leading to improved water availability in 2020–21, there were record high volumes of water allocations traded (7,991GL) and therefore prices paid decreased significantly from the previous year (BoM, 2022b; 2022c). Trading water is also associated with increased returns over time, though returns can be variable (Wheeler et al., 2016).

2.3.5.3 *What is known about the major risks, threats and recoverability of water market economic values?*

The main risks to water market values are associated with water scarcity in the Basin (hence see the agricultural area for more discussion on this) (Qureshi et al., 2012). Given that volumes traded, and prices paid are directly related to water scarcity, this therefore has direct implications, with both benefits and costs (depending on a buyer or seller of water point of view). For example, water prices are higher in a drought, hence increases the value traded, but this can have positive impacts from a seller point of view, or negative impacts from a buyer (Wheeler et al., 2020c).

Water markets are controversial in public, and especially rural opinion (Wheeler, 2022). Existing (and perceived) flaws in the water market system, its data quality and transparency, and allegations of misconduct lead to several water market studies and reviews, the latest by the Australian Competition and Consumer Commission (ACCC, 2021). Hence, one potential risk is loss of public confidence and participation. To address issues of market failure, the ACCC made 29 recommendations for water market reform, consisting of 70 distinct action items in the overarching categories of market integrity and conduct, trade processing and market information, market architecture, and governance of Basin water markets (ACCC, 2021). The Commonwealth government has recently published recommendations on how to implement the ACCC water market reform recommendations (Quinlivan, 2022), with the responsible Water Minister supporting the implementation of all 23 recommendations (DCCEEW, 2022c).

Water recovery impacts on water markets have been modelled via a number of different methods. The most modelled economic outcome from water recovery using econometrics has been water market prices. Aither (2016a) collects the annual median water allocation price from the ABS from 1998 to 2014, and measures the potential impact of Commonwealth water purchases on historical water allocation prices. It also finds that Commonwealth water purchase increases the annual median water allocation price by using the water allocation price model, which is based on a regression analysis over 17 years of observed data (n=17). Specifically, the difference between modelled annual median prices with and without Commonwealth purchases is $24/ML, increasing from $88-$112/ML. Yet it highlights that Commonwealth water purchase is a less important driver (i.e., a quarter of the increase in temporary water prices) of allocation prices compared with total water availability and prevailing climatic conditions. Aither (2016b) analyses demand change data over the past ten years (2005 to 2015), and looks forward to the demand change that occurs over the next five years (2015 to 2020). It finds that the water allocation price could be 13-36% higher in a moderate allocation season with Commonwealth environmental water purchases. It also projects that there are likely to be further significant changes in the sMDB allocation market between 2015-16 and 2020-21. Specifically, over the next five years, allocation prices are estimated to increase from $207 to $231/ML ($24/ML or 12%) in low allocation seasons, $118 to $131/ML($13/ML or 10%) in moderate allocation seasons, and $37 to $41/ML ($4/ML or 9%) in high allocation seasons.

However, Zuo et al. (2019) apply the VARX-BEKK-GARCH time-series regression to model the water market dynamics of monthly permanent and temporary water market trade from 1997 to 2017 in the Goulburn-Murray Irrigation District. It finds that water buyback has a small price and volume impact and highlights that estimates in some other studies about the impacts of government water recovery on water markets (e.g., RMCG, 2016) are overestimated. Specifically, it finds that the effect of government recovery is not significant on temporary water prices or permanent market prices and volumes, and 1% increase in water recovery resulted in a 0.14% reduction in temporary water volume-traded.

ABARES (2020a) used annual data from 2006 to 2019 to estimate the impact of water recovery on water allocation prices in the sMDB and find effects on market prices have increased over the period. Specifically, the estimated average effect of all water recovery across a mix of 'dry', 'typical' and 'wet' years on price was $72/ML.

2.3.5.4 What is known or estimated about the future condition of water market economic values, especially under climate change?

Given that water market economic values (both in prices and volumes) are intrinsically tied to water availability – climate change will have significant impacts on water market economic values.

2.4 Indirect economic values in the MDB

2.4.1 Ecosystems services values

2.4.1.1 What is known about the benefits and impact of ecosystems economic values?

Green & Moggridge (2021) provide information about inland water issues in Australia's state of the environment 2021. The MDB has high ecological values, with water supply, diverse species, and other ecosystems (Pittock & Finlayson, 2011). Ecosystems values include a wide range of water and other habitat values. There are nearly 5.7 million hectares of wetlands in MDB (Kingsford et al., 2004), among which 16 were listed as internationally important under the Ramsar Convention on Wetlands (DEWHA, 2009). The afforestation and farm forestry in the MDB also contributed to increased carbon sequestration, offsetting increased carbon emissions (AGO, 2007).

MDBA manages the River Murray water quality monitoring program to monitor water quality on an ongoing basis. Under the program, water samples are collected at regular intervals from 28 sites along the River Murray and across its tributaries in New South Wales, Victoria and South Australia (MDBA, 2022). A variety of characteristics are examined from the samples of these sites, including electrical conductivity (indicator of salinity), pH (indicator of acid or alkali), temperature, turbidity, total phosphorus and total nitrogen, soluble organic carbon, silica, sulphate and bi-carbonate and chlorophyll and phaeophytin (indicators of algal health). Data from sites of the River Murray water quality monitoring program and from other locations are available via WaterNSW (WaterNSW, 2022), and the Victorian Department of Environment, and Land, Water and Planning (Department of Environment, Land, Water and Planning, 2022). These data are real-time data, with the Victorian Department of Environment, and Land, Water and Planning publishing real-time (less than 1 hour old) data from 2000 up to now, whereas data from a WaterNSW monitoring site is typically measured every 15 minutes from 2010 up to now. Besides surface water and ground water quality, these datasets also collect a wide range of other data, including surface water level and flow, groundwater levels, storage level and volumes, and biological conditions. Biodiversity such as waterbird and fish population is also sampled at different sites in studied river systems (Growns, 2007). For example, Growns (2007) examined the abundances of individual fish species in relation to hydrological change in six regulated rivers in the MDB. Fish were sampled by boat via electrofishing in 53 sites on 6 river systems in 1999/2000, 2000/2001 and 2001/2002 in either spring, summer and/or autumn. The carbon stock can be calculated by Full Carbon Accounting Model (FullCAM) which is the software used to construct Australia's national greenhouse gas emissions account for the land sector (Maraseni & Cockfield, 2011; Paul et al., 2013). The biodiversity and carbon value can also be evaluated by a benefits-transfer approach. The benefit transfer method transfers available information from stated or revealed preference studies already completed in other location/contexts to the context under investigation. There are a range of publicly available benefit transfer websites. Some of these include: USGS Benefit transfer toolkit; the Environmental Valuation Reference Inventory (EVRI); or the specific i-Tree tools website.

2.4.1.2 What is known about current condition and trend of ecosystem economic values?

Bark et al. (2016) applied avoided cost methods to estimate the value of ES improvements linked to water quality improvements under the Basin Plan. It is estimated that the value of the improved water quality is about $2 million totally ($1.1 million of reduced salinity, and $0.9 million of reduced cyanobacterial bloom risk) with the implementation of the Basin Plan.

In 2020-21, 0.62 million tonnes of salt was exported from the MDB into the southern ocean, higher than it had been in years, but less than the Basin Plan objective of 2 million tonnes annually (Aither, 2022). Climate, weather, and land use change are the main drivers of salinity in the MDB (Holland et al., 2015). Climate and weather are the natural, biophysical drivers of salinity, while the amount of rainfall can largely explain changes in the amount of groundwater recharge (Petheram et al., 2002).

According to Holland et al. (2015), salinity can be measured as the Electrical Conductivity of streams or groundwater, while soil salinity is measured by the electrical conductivity of its saturated extract. Groundwater quality assessments in the MDB in 1988 showed that almost half (48%) of the MDB's groundwater was saline (Williams et al., 1994). Moreover, more than 5 million tonnes of salt were assessed in 1999 to mobilise over the MDB (MDBC, 1999). Jolly et al. (2001) estimated the trends in stream salinity between 1985 and 1994 in MDB and calculated the salt output/input ratio, which showed that salt was mobilised in the upper catchment areas and stored in mid-catchment irrigation districts to be remobilised later. The MDB salinity strategies successfully achieved the Basin Plan goal by 2010, while the Basin Plan provides the potential for groundwater pumping for salt interception to be reduced over time (Walker & Prosser, 2021). However, the historical land use changes stimulate groundwater processes and potentially lead to future salinity increases. Despite this, the likely salinity benefits from the Basin Plan could possibly offset these potential increases (MDBA, 2015). Also, climate change with higher temperatures, less rainfall, and longer droughts (BOM & CSIRO, 2018) would probably reduce salt impacts. However, these impacts would be lessened by flooding and recharge from higher-intensity rainfall or reduced dilution effects from reduced runoff. Moreover, future changes in land use would be another big driver which will affect recharge and lead to salinity (Walker & Prosser, 2021). Adamson et al. (2007) applied linear and non-linear programming models to show the potential value of improved water use. They found that the value of the loss in yield due to salinity is approximately $100 million, but the social loss in the sequential solution, relative to the global optimum, is significantly greater than the value of the direct loss in yield due to salinity, which is estimated to be $400 million.

McLean et al. (2007) conducted the Economic Assessment of Salinity Impact on Lower Murray Horticulture by measuring the value of lost production in the 4 major sections of the

River when and if Morgan salinity reaches 1000 EC during the growing season. The assessment involved range of industries/crops included in each of 4 River sections starting Mildura to Lake Alexandrina. The results indicate that with 100% Leaching Efficiency (LE)[8], the total losses along the four sections of the river were found to be between $50 - $60 million with 1000 EC (1dS/m) at Morgan. At 70% LE, the total losses along the four sections of the river would be between $80 - $90 million with 1000EC at Morgan.

Nutrients such as nitrogen (N) and phosphorus (P) are natural elements of water bodies. However, excessive supply of N or P can stimulate excessive and nuisance levels of algae, cyanobacteria ('blue-green algae'), or macrophytes (Walker & Prosser, 2021). These organisms can grow to the detriment of other organisms, disturb aquatic food webs, and smother riverbed habitats (ANZECC, 2000). Rapid growth of algae, referred to as an 'algal bloom' has been of concern in the Basin. The death of a large biomass of algae kills fish and has other impacts. This occurred in the lower Darling River in January 2019 (AAS, 2019).

The MDB also suffers from another water quality issue, cyanobacterial bloom. Toxins released by blue-green algal blooms are detrimental or even fatal to stock and humans. In 1991, a toxic cyanobacterial bloom occurred in the Darling river which extended for 1000 km (Pittock & Finlayson, 2011). Previous studies showed that sediment and attached nutrients significantly harm water quality in the MDB (Clifton et al., 2007). Wetlands along the River Murray have changed from clear water, benthic macrophyte-dominated systems in historical times, to turbid phytoplankton-dominated systems (Walker & Prosser, 2021; Gell et al., 2006; Ogden, 2000; Reid et al., 2007). Also, Thoms and Delong (2018) found the Darling River also experienced increased erosion and presumably increased suspended sediment loads.

2.4.1.2.1 Biodiversity

Biodiversity is another key issue that needs attention in the MDB (Booth, 2012). The MDB was assessed to have poor conditions regarding fish species. Davies et al. (2010) found that fishing species in thirteen river valleys in MDB were rated as very poor, seven were poor, two were moderate, and only one was good. Arthington and Pusey (2003) pointed out that the MDB's aquatic biodiversity has been severely affected by barriers, cutting access to rivers and floodplains, and flood levees, with more than 3600 weirs in the MDB that block longitudinal connectivity.

Due to various reasons (e.g., flow regulation disrupting the natural water-regime triggers for fish spawning, thermal pollution, and barriers to movement), native fish make up only 20% of the total catch in the regulated rivers in the MDB (Gehrke et al., 1995; Gehrke & Harris, 2001; Growns, 2008). Large water extraction is one of the primary reasons for fish reductions (Gehrke et al., 1995; Grafton et al., 2022). Bowling and Baker (1996) and Gehrke et al. (1995) found that extensive water extractions along the Darling River contributed to the 1991 blue-green algal bloom, which led to declines in the abundance and diversity of native fish. Jackson and Head (2020) investigated the possible effects of water extractions following the 2019 Menindee fish kills and showed the importance of habitat connectivity for fish spawning and fish movement along the Darling River. Pittock and Finlayson (2011) also suggested that restoring connectivity is vital for freely moving aquatic species to more favourable places. Moreover, water extractions are also found to be associated with waterbird abundance. Grafton et al., (2022) showed that reduced water extractions upstream could increase downstream streamflow at Wilcannia (Barka River) and therefore enhance the

[8] Leaching efficiency is the efficiency with which the leaching fraction carries the salt when it moves through the soil profile.

resilience of waterbird abundance. Also, an in-stream water reallocation from irrigation could facilitate the recovery of waterbird abundance. Moreover, besides waterbirds and fish, reduced extraction and increases in streamflow may also be beneficial for other flora and fauna such as wetland plants and trees, mammals, frogs and reptiles (Krauss et al., 2018).

Williams and Cary (2002) found a range of benefits of forestry, agroforestry revegetation and native vegetation on biodiversity protection, dryland salinity and other water quality problems. The value of biodiversity is found in Rolfe et al. (2000) at AUD $0.5 million for non-threatened species and $3.4 million for endangered species.

Healthy ecosystems and green spaces also provide a range of benefits. Varcoe et al. (2016) assessed the economic value of public goods provided by Victorian parks (many of which are within MDB boundaries). The study draws on environmental accounting and environmental valuation to provide a snapshot of parks' ecosystems, and to evaluate the quantity of associated ecosystem services. Victorian parks account for 38% of all native vegetation and 60% of wetlands of international significance in Victoria. They also provide the ecosystem services of provisioning (e.g., clean water, honey); regulating (e.g., water purification, air filtration, pollination); and cultural services (e.g., recreation, amenity, heritage connection). Parks contribute more than $1 billion in Gross Value Added from tourism, and 14,000 jobs. Park-based apiaries produce $3.4-4.6 million worth of honey and $123-167 million worth of pollination services per year. Water filtration through non-metropolitan parks' ecosystems is valued at $50 million per year. Other benefits include recreational value to visitors of $600-1,000 million, and additional $80-200 million in avoided health costs through physical activity. The heritage value of Victorian parks was estimated $6-23 million per annum.

2.4.1.2.2 Carbon sequestration

Maraseni et al. (2011) find that the total net present value from absorption of greenhouse gases in plantations ranges from $490/ha to $862/ha. Regan et al. (2020) evaluated the economics of land-use change via active afforestation for local carbon abatement in SA. They estimated that a carbon price of AUD$50/tCO2.would be required to incentivise land-use change to carbon farming.

Schrobback et al. (2011) simulated the effects of payments for carbon sequestration in the south-eastern catchments for the MDB. The simulation results show that at least $100 per tonne of CO2 will be required for land users to create a price incentive for large-scale forest plantations to be established in the south-eastern catchments of the basin. However, even with high carbon permit prices, the contribution of forestry establishments in the south-eastern catchments of the Basin to emission mitigation is still found to be modest. It also highlighted that the price of $100/tonne is unlikely to be realized in the near future under current policy settings. Moreover, the increasing carbon permit price is also found to significantly reduce the environmental flows, water use and water quality.

Settre et al. (2019) used a dynamic hydro-economic simulation of river flows, floodplain inundation, forest carbon dynamics, carbon credit value, and water opportunity cost in the Murrumbidgee. The study results indicate possible synergies in joint provision of carbon sequestration and environmental flow benefits through a carbon-water trading strategy. This involves funds for environmental water purchases generated through sale of carbon credits from improved floodplain conditions. Results identify limited trading opportunities at the carbon price (AUD$13/tCO2), resulting in an economically viable re-allocation of 2.31 GL/year (0.1% of water currently diverted for irrigation) to the environment with frequent years of zero re-allocation. At prices above AUD$20/tCO2, there may be additional trading opportunities and as much as 5% of current irrigation diversion was predicted to be reallocated at AUD$100/tCO2.

2.4.1.2.3 Dredging costs

One method of estimating the economic value of having an open Murray Mouth to the sea is the cost of dredging (a form of damage and preventative economic valuation approach). There has been no official research work done on the cost of dredging the Murray Mouth – so this section reports information mainly from newspaper articles (which were not officially reviewed as part of our literature review). Dredging has been required most years[9] from 2003/04 – 2021/22 to keep the Murray Mouth open and free from silting up, a crucial criterion of the Basin Plan. The significant costs are shared equally between the Basin states and the Commonwealth under the River Murray Operations budget. The dredging is contracted out by SA Water, which is subsequently reimbursed (Cardno, 2019). Given the complicated financial and funding structure of the River Murray Operations budget and the dredging contracting arrangements, annual cost of dredging cannot be easily discerned for all years. However, dredging costs were said to be $7 million annually from 2003-2005 (The Age 2005), estimated at a total of more than $40 million for 2003- 2011 (Kelton, 2010), between $3 -$6 million in 2014/15 (Strathearn & Simmons, 2020), $6.4 million in 2015/16 (Strathearn, 2016), $5.65 million in 2016/17 and $7 million in 2017/18 (Aither 2017a), and $6.1 million in 2018/19 (Strathearn & Simmons 2020). While Aither (2017a) states that dredging costs can be as low as $50,000 in years of high flows, water flow over the barrages in the last two decades was insufficient for significant sediment export through the mouth, and required high levels of dredging, as demonstrated by reported dredging costs mostly around $6 million per year. This situation is unlikely to change in the future. Indeed, in September 2022, SA Water awarded a new four-year contract for dredging the Murray Mouth until November 2026, for a total contract expense of $31,864,200.50, or $7,966,050 annually (SA Tenders & Contracts, 2022). Together with the expenditure increase to nearly $8 million per year, the new contract also includes a significant increase in the average dredging rate to 8,000 m^3/day, as compared to an average dredging rate of 5,000-6,000 m^3/day previously (Strathearn & Simmons, 2020). This increase is facilitated by replacing the current two smaller dredges with one large dredge and replacing current 290mm diameter pipelines with a size that can accommodate larger dredging equipment (SA Tenders & Contracts, 2022).

2.4.1.3 What is known about the major risks, threats and recoverability of ecosystems economic values?

Clearing native vegetation for agriculture is extensive and has significantly changed land use in the MDB over the past 150 years. Consequently, various water quality issues arose and worsened significantly in MDB with the widespread clearing of native vegetation. Land use change which contributes to salinity has been driven by the clearance of native vegetation and introduction of farming systems that use less water (Haron & Dragovich, 2010; Zhang et al., 2001). The expansion of irrigation caused increased irrigation-related salinity, rising water tables and inadequate flows of water to sensitive ecosystems. Salinity affects water quality through substantial increases in salt content. It impacts the quantity of food products by reducing plant growth and affecting the quality of food products to varying extent (Nuttall et al., 2003; Munns, 2005; Holland et al., 2015).

[9] Dredging operations were not required from 2011/12 – 2013/14 due to high river flows in these years.

Further, other risks such as large water extraction and changed land use by expanded agricultural land, combined with climate change have affected water quantity, quality, and the threat to aquatic species, thereby resulting in ecological value loss in MDB (Gehrke et al., 1995; Timbal & Jones, 2008; Pittock & Finlayson, 2011; Grafton et al., 2022). Hence, various programs have been implemented such as interventions to reduce salinity levels, cyanobacterial blooms, restore native fish populations and provide environmental flows.

Many studies on native vegetation and water quality find that plantation, agroforestry, and restoration plantings lead to higher water quality – either through reducing sediment, total nitrogen and phosphorus, reducing salinity, or increasing clarity of water and oxygen level. Several studies find that controlling the water table using trees through revegetation is an efficient way to control and manage dryland salinity (Stirzaker et al., 1999; Dunin, 2002; Hajkowicz & Young, 2002).

Acid sulphate soils are the soils or sediments that contain sulphide minerals or sulphide minerals that have subsequently oxidised (Fanning, 2002; Baldwin, 2021). It was believed that acid sulphate soils did not occur in inland Australia, until around 2011. Acid sulphate soils have since been detected throughout Australia, including the MDB (Baldwin, 2011a, 2011b). According to Fitzpatrick et al., (2012), changes to the hydrology in regulated sections of the MDB system, and the chemistry of rivers and wetlands, have caused significant accumulation of sulfidic material in subaqueous and margin soils. In wetlands of the lower River Murray, pH levels of less than two have been recorded (McCarthy et al., 2006; Baldwin, 2011b). According to Baldwin (2021), risks associated with acid sulphate soils include: (i) mobilisation of metals, metalloids, and non-metals, (ii) decrease in oxygen in the water (iii) production of toxic gases, (iv) direct exposure to acidic minerals (v) mobilisation of acidic minerals by wind, and (vi) damage to infrastructure. These risks can potentially lead to environmental damage, deterioration of water quality, and harm to human and livestock health.

Blackwater events are characterised by high concentrations of dissolved organic carbon (DOC), sufficient to give the water column a dark 'tea' colour associated with reduced levels of dissolved oxygen in the water column (Howitt et al., 2007; Whitworth et al., 2012). Blackwater events generally occur in low-gradient river systems with forested floodplains or extensive wetlands. When flooding occurs, it allows carbon movement from the floodplain to the river channel. Microorganisms can immediately use up about one-third of the carbon leached from the leaf litter, using oxygen from the water as they consume the dissolved carbon (Baldwin, 1999; Howitt et al., 2007). Therefore, blackwater plumes often have very low levels of dissolved oxygen, which leads to the death of fish and other aquatic animals that rely on dissolved oxygen for respiration. Baldwin (2021) pointed out that water temperature plays a vital role in determining whether a blackwater event will lead to the deaths of native fish and other aquatic animals. Specifically, increasing water temperature contributes to the reduction of the amount of oxygen that can be dissolved in the water (Whitworth et al., 2014).

In south-eastern Australia, a series of spring and summer flood events occurred in 2010–2011 after a decade long drought, resulting in a large-scale hypoxic blackwater event in the sMDB (Whitworth et al., 2012). It affected over 2,000 km of river channels and persisted for six months. This hypoxic blackwater event was found to be driven by unseasonal inundation of forested and agricultural floodplains (Howitt et al., 2007; Whitworth et al., 2012). Moreover, Whitworth et al. (2012) also showed that hypolimnetic discharge from weirs and post-drought flushing of the upper catchment could also contribute to blackwater events. They also showed that patterns of carbon release varied substantially with catchment land use and carbon reactivity. The altered seasonality of flooding increased the risk of hypoxic blackwater generation.

Deforestation is the second largest contributor to carbon dioxide emissions, while on the flip side, afforestation has the potential to sequester carbon (Booth, 2012). A range of efforts and initiatives have been made previously to encourage afforestation and farm forestry in the MDB and other areas of Australia (Booth, 2012).

2.4.1.4 What is known or estimated about the future condition of ecosystems economic values, especially under climate change?

The predicted future climate, such as increased climate variability including more extreme droughts and floods, in sMDB and other Mediterranean climate zones would potentially increase the risk of blackwater events and cause water quality issues in the future (IPCC, 2007; Suppiah et al., 2007). Biodiversity, salinity issues etc will also suffer though climate change impacts such as drought and decreased rainfall, with increased evapotranspiration as a result of increased air temperatures leading to the loss of large quantities of water from the MDB's surface, soils, and aquifers (Leblanc et al., 2009).

2.5 Non-use economic values in the MDB

2.5.1 Option, bequest, and existence economic values

2.5.1.1 What is known about the benefits and impact of non-use economic values?

A number of key valuation studies have examined the notion of a healthy MDB under a range of different contexts. Some of these studies have already been referenced in this document previously. Meanwhile other studies have employed the choice modelling technique to evaluate the benefits of wetlands and rivers – some notable examples here being Bennett et al. (2004; 2007; 2008), Gillespie Economics et al. (2008), Morrison and Bennett (2004), van Burean and Bennett (2000) and Whitten and Bennett (2001). These studies identified a wide variety of tangible values under a healthy MDB system, including those obtained from fishing, abundant bird life, vegetation, and the provision of hunting grounds.

Hatton-MacDonald et al. (2009) reviewed available studies of the environmental and recreational values associated with the MDB, floodplains and wetlands, finding that Australians were willing to pay substantial amounts to improve the quality of the River Murray and Coorong – indeed much more than had been found under previous studies. Hassall and Associates and Gillespie Economics (2003) undertook an investigation under *The Living Murray* initiative and estimated the economic value of non-agricultural river dependent industries in the southern MDB alone at $1.62 billion annually. This included the value of camping and caravan parks, recreational fishing, and boating activities. Hatton-MacDonald et al. (2009) suggested that amenity values and the value of lifestyle properties were increasing.

Brouwer (2009) analysed choice modelling studies applicable to Australian rivers and wetlands over the previous decade and concluded that Australians valued the use benefits of water resources (such as swimming and fishing) significantly higher than non-use benefits (such as ecosystem conservation).

In NSW, Mazur and Bennett (2009) generated environmental values, by catchment, to be used by Catchment Management Authorities (CMAs) when evaluating alternative natural resource improvement programs. Environmental asset values estimated included the area of native vegetation in good condition, number of native species present, length of healthy waterways, and agricultural employment. Bennett et al. (2004) estimated community willingness to pay to maintain viable rural communities in the Murrumbidgee (southern NSW). The evidence showed that a significant proportion of households were willing to pay to see rural population levels maintained. However, an important caveat to this result was that support was not maintained in the absence of any environmental stewardship obligations.

A further study of non-use values obtained from the River Red Gum and East Gippsland Forests in the state of Victoria (Bennett et al., 2007) was commissioned by the Environment Assessment Council. The environmental attributes valued in the River Red Gum Forests region were: area of healthy River Red Gum forest; Murray Cod and other threatened native fish populations; breeding pairs of endangered Regent and Superb Parrots; and the provision of recreation facilities for campers along the Murray River. While, for the East Gippsland Forests, attributes valued were: breeding pairs of endangered Masked, Powerful and Sooty Owls; number of threatened Long-footed Potoroos; area of significant rainforest sites protected; and area of old growth forest protected.

CSIRO (2012b) identified and quantified the ecological and ecosystem services benefits that arise from recovering 2,800GL/year of water for the environment in the MDB. Using economic valuation techniques, it estimated the habitat ecosystem services which arise from floodplain vegetation, water-bird breeding, native fish and the Coorong, Lower Lakes, and Murray Mouth – is worth between $3 to $8 billion under the 2,800GL scenario. Under the 2,800GL scenario, the additional volume of carbon held within river red gum and black box floodplain vegetation is worth $120 million to $1 billion. The increased supply of the aesthetic appreciation ecosystem service under the 2,800 scenario is worth more than $330 million. The avoided damage and treatment costs associated with the supply of fresh water are worth $30 million and the tourism benefits are worth up to $160 million annually. Hatton-MacDonald et al. (2011b) published the willingness to pay estimates for improvements in environmental quality for the River Murray and the Coorong. Bark et al. (2016) provided another summary of this work in a published journal article, as well as a spatial distribution of the costs and ecosystem service benefits under the Basin Plan.

Deloitte Access Economics (2012) employed a travel cost method to estimate the benefits of the Basin Plan for the fishing industries in the MDB. The results suggest that the overall value of the fishing industry is estimated to increase by $28 million per annum due to 2,750GL Basin Plan, which also leads to an increase in consumer surplus of $9.1 million per annum for recreational fishing and an increase in producer surplus of $254,000 per annum for commercial fishing. GHD (2012) provided an estimated of the benefits of the plan for floodplain primary producers. A healthy MDB has also been found to have an impact on nearby property values. Tapsuwan et al. (2015) conducted a hedonic property price analysis of house sales between 2000-2011, to estimate the marginal value of instream flows and proximity to an iconic MDB freshwater ecosystem, the Barmah-Millewa Forest. A non-linear relationship was identified between instream flow and sales price, suggesting purchasing preferences for flow that is neither high (i.e., flood flows) nor low (i.e., drought flows).

It is worthwhile reflecting on a recent stated preference study conducted in Australia – that although is not directly related to native vegetation ecosystem services, is a reflection on how much Australians are willing to pay to conserve native species. Zander et al. (2022) applied the contingent valuation and choice experiment to estimate the value 12 threatened species, using survey data of 2,400 respondents from the general Australian public. They found that the median annual willingness to pay across all 12 species was $60/year, ranging from $39.6/year to $128.81/year. A previous contingent valuation study conducted by Zander et al. (2014) explored funding support for threatened bird conservation in Australia, revealing that two thirds of respondents (n=645) were willing to pay towards a hypothetical fund for bird conservation. On average, each respondent was willing to pay $11/year – or the equivalent of a conservative estimate of $14 million/year for threatened Australian birds, when extrapolated to the entire Australian population.

A study by MJA (2017) estimated the environmental values of Queensland MDB regions, such as the Condamine-Balonne and the Border Rivers regions, with a total population of 220,535 and 100,675 jobs in 2011. To this end, the report uses government and census data on regional population and economic activity, and transfer pricing of values for ecosystem services assessed in previous scientific studies, rather than undertaking valuation techniques directly. Thus, the value of regional tourism, dependent on healthy water ways and aquatic ecosystems, is estimated at $952 million per year, whereas recreational fishing, and boating and water-based recreational activity benefits are valued at $104 million and $128 million per year respectively. Willingness to pay to protect and maintain wetlands in the region was estimated around $1.9 billion in order to maintain biodiversity and ecosystems. An overview of estimated economic contributions of different sectors in the region indicates that environmental values and ecosystem services contribute 15% ($1,184 million from tourism, recreation, and fishing) to the overall economic activity ($7,415 million[10]) in the region.

2.5.2 Cultural economic values

The aquatic ecosystem in the MDB also has important cultural values for both Indigenous and non-Indigenous people (Robinson et al., 2015). Indigenous peoples of the MDB hold distinct water perceptions related to cultural identity, community, and connection to place (Bischoff-Mattson et al., 2018). Water contains deep cultural significance, regarding the understanding of traditional heritage, human–environment relations, and the rights and custodial responsibilities of traditional ownership (Morgan et al., 2004). Considering the significant cultural value of water resources, Johnston et al. (2011, 2012) revealed that integrating social and cultural values into environmental management and planning has become a key focus. However, while the cultural value is important, there is a lack of recognised and widely accepted methods to evaluate cultural values (The Getty Conservation Institute, 2002; Venn & Quiggin, 2007).

For example, The Yorta Yorta consists of 16 family groups (95 percent of the Indigenous population) in the Barmah-Millewa region, which contains forests and wetlands that hold special significance for local environmental and cultural values (Weir, 2009; Yorta Yorta Nation Aboriginal Corporation, 2014). Yorta Yorta water values also include cultural values like many other Indigenous communities in the MDB, which include cultural identity and connection to their country, community, and wellbeing as a result of waterways, and watersheds being healthy (Weir, 2009; Jackson et al., 2010; Robinson et al., 2015).

The concept of cultural water was developed during the Millennium Drought (COAG, 2004a, 2004b): the function of cultural water was to provide culturally significant river flows and as a means of getting more water 'for country' (Weir, 2016). Moreover, cultural flows were

[10] Note that the report states overall economic activity as $7,415 million, which is less than the sum of displayed subcomponents (MJA, 2017).

introduced and defined by the Murray and Lower Darling Rivers Indigenous Nations (MLDRIN) in 2007. Specifically, it was defined as water entitlements that are legally and beneficially owned by the Indigenous Nations of a sufficient and adequate quantity and quality to improve the spiritual, cultural, natural, environmental, social, and economic conditions of those Nations' (MLDRIN, 2007). Since then, different steps have been taken towards clarifying and operationalizing a cultural water policy relative to environmental water management (Bischoff-Mattson et al., 2018).

In terms of cultural values held by the general public, Jackson et al. (2019) found in a survey of MDB residents that 70% of respondents supported reallocating 5% of total irrigation entitlements to Indigenous communities, with no preference for how that water should be used. An earlier study by Bark et al. (2015) applied a cultural ecosystem services framework to a location of major indigenous cultural significance within the MDB – the Brewarrina Aboriginal fish traps – and examined the potential implications for water planning. The data obtained from qualitative interviews with indigenous custodians demonstrated a wide range of cultural values and associated benefits with respect to the fish traps themselves, and to their connectivity with another key water site, an upstream lagoon.

In Noble et al. (2018) the values held by various stakeholders, such as Aboriginal Traditional Owners, landowners, tourism businesses, scientific researchers, non-governmental organizations, and government agencies, regarding the Murray crayfish (Euastacus armatus) were examined. Using qualitative methods (i.e., individual semi-structured interviews with expert informants) this study found that the Murray crayfish is connected to culturally significant values with relevance to fishing and other non-extractive activities. Relevant stakeholders agreed on four important themes to achieve a more equitable and effective conservation and management: increased public education, co-management with non-government stakeholders, federal government co-ordination, and spatial protection of critical areas. Stakeholder-led or bottom-up approaches are key for a strong conservation and management design. Better stakeholder engagement and co-management may improve capacity and confidence of managers to implement strategies that support the social and ecological resilience of aquatic ecosystems.

Various information on indigenous corporations, employment and water ownership exists in the Basin (Aither, 2022), suggesting that the gap is still large between indigenous and non-indigenous.

2.5.2.1 What is known about current condition and trend of non-use economic values?

Non-use valuation studies tend to be one-off studies, and usually are not conducted on the same attributes or populations, so it is difficult to establish trends. But in general, people's willingness to pay for environmental (or cultural) attributes often increases over time as the environment is valued more and income/wealth increases.

2.5.2.2 What is known about the major risks, threats and recoverability of non-use economic values?

The same risks and threats face non-use values as implicit in the use values described above.

2.5.2.3 What is known or estimated about the future condition of non-use economic values, especially under climate change?

Future non-use values will be dependent upon the condition of use values described above. They will also be dependent upon the general preferences of Australians, and as such, may increase with expected wealth and increasing scarcity.

2.6 Summary of the economic values in the MDB

Table 2.5 provides a summary of all the economic values in the MDB and a qualitative overview of future risks.

These economic values in the MDB are variable and driven by various factors. Based on the review, the main drivers of MDB direct economic values include climate factors, external influences (terms of trade on agricultural profitability), public service investment, community factors, technology, trade and, to some extent, water recovery programs.

However, particularly the case with the impact of water recovery on economic agricultural and community values, there is a vast difference within the literature as to the actual impact of water recovery on economic values. Regarding recovery programs, there is a concern that buyback policy has been detrimental to fundamental economic values. And in particular, any future water policy that involves returning water from consumptive to environmental use would also be detrimental.

To understand this issue further, we must investigate the quality of evidence that has sought to evaluate the impacts of water recovery and/or reduced water diversions on economic outcomes. We have to understand the quality of previous studies in order to establish confidence in their findings.

To the best of our knowledge, there is no comprehensive systematic review of different drivers and their impacts on MDB economic values and quality assessment of the qualities. As such, this report addresses these gaps by reviewing economic studies involving the MDB and assessing the various quality of these studies – to better understand the effects of key drivers on economic values, along with the latest findings from high-quality research.

Therefore, the following chapters focus on providing a systematic literature review regarding the drivers of economic values in the MDB, together with their impacts, and the quality assessment.

Table 2.5 Summary of direct, indirect and non-use economic values in the MDB

Categories	*Measures*	*Values*	*Trends*	*Major risks, future conditions*
Agricultural economic values	GVAP/GVIAP (data source: ABS)	About $29,000 million in 2020-21 for GVAP, and $8,500 million in 2020-21	GVAP increasing from 2007 to 2021, and GVIAP increasing in general from 2005-2021	The climate will be hotter and drier, with rainfall more variable. Aust. agricultural productivity as a whole is projected to decline significantly by 2050 due to climate change. Farmers terms of trade important, along with other economic factors
	Profit (data sources: ABARES, ABS, ATO)	Rate of returns highly variable, 3% average 2013-2015	Increasing slightly over time, but highly variable	Climate change will influence profits and reduce net revenue from irrigation significantly without adaptation. Farmers terms of trade important, along with other factors
Community economic values	GRP/GDP/GSP (data source: ABS)	$232 bill of GRP in MDB, 2020-21	Increasing since 2010	Drought, social services, climate change, terms of trade
	Employment, unemployment rate (source: ABS)	About 1.63 million in 2020 to 2021	Increasing from 2010 onwards	Drought, social services, climate change, terms of trade
Recreational, fishing & tourism economic values	Recreation & Tourism (sources: Tourism Research Australia, ABS, BLADE, consultancy & academic research)	Variety of different benefits found for skiing, boating, tourism	Increasing from 2001 onwards	High dependency on flow regime of ecosystem services makes the region highly vulnerable to climate change
	Economic values of fishing (sources: NSW Recreational Fishing Survey, consultancy reports, academia)	Gross output of recreational fishing in the Basin worth $108 million in 2018	Increasing during from 2014 onwards	Projected climate changes, declining river flows, and increasing salinity levels will be main threats and thus better management is required
Mining & Energy economic values	Mining & energy (source: state gov data; ABS, private industry reports)	Substantial revenue, wages, and jobs	Uncertain	Climate change emissions, soil and water degradation issues, groundwater contamination
Water market Economic values	Water prices and trade volumes (data source: BoM; state water registers, academic and consultancy research)	$5+ billion turnover in the MDB in 2020-21	Increased over time	Water markets can be controversial, and therefore one potential risk is loss of public confidence and participation. Other risks to water market values are associated with water scarcity and climate change
Indirect value: ecosystem service values	Water quality (e.g., salinity, sediments) (source: water quality monitoring program - MDBA), biodiversity (irregular), carbon sequestration, consultancy and academic research	The MDB has high ecological values, with water supply, diverse species and other ecosystems providing diverse values	Salinity quality improved. Other ecosystems services decreased and experienced ecological value loss	Large water extraction and changed land use by expanded agricultural land, combined with climate change have affected water quantity, quality, and threat to aquatic species, thereby resulting in ecosystems value loss
Non-use economic values & cultural values	Option, bequest and existence & cultural values (estimated by methods such as choice modelling, travel cost method) (consultancy and academic literature)	Values vary by different study areas	Unknown	Climate change, over-water allocation, decreased water quality and quantity are main threats

3 Systematic Literature Review of Studies Investigating Water Reduction Impacts, and Development of Quality Assessment Methods

Following the economic values chapter, this chapter provides a systematic literature review of previous studies investigating the impacts of water reduction in the MDB (journal articles, working papers, reports, and theses), following guidelines by Khan et al. (2003). The procedures included framing the research question, identifying relevant publications, assessing study quality, summarising evidence, and interpreting findings. The research question is framed as investigating MDB studies by examining the conditions of the Basin, drivers (e.g., climate change, drought, Basin plan, water trade) of water-related values. Given the diversity of findings in the economics literature regarding the impacts of reduced water, we developed a methodology in the chapter to allow us to rank the quality of the evidence in each study.

3.1 Review criteria and data collection

We employed Web of Science, EconLit, and ScienceDirect as the database searching tool for journal articles and Google Scholar for grey literature. We conducted a two-stage process to identify the relevant studies. Firstly, we undertook a search of literature within Web of Science, EconLit, and ScienceDirect databases. We started the search in 2007, due to Commonwealth water recovery starting at this date, and the need to focus on the most up-to-date literature. BDA Group (2010) provide a review of all the water economic literature prior to 2010 – hence we direct readers to this reference for the earlier literature. Keywords for the search were combined and are shown in Table 3.1.

Table 3.1 Searching strings used

Database	Searching String
Web of Science	TS=("Murray-Darling Basin" AND ("water markets" OR "water recovery" OR "water policy" OR “water use”) AND ("economic impacts" OR "farm profitability" OR "impact" OR "economic value"))
Science Direct	Title, abstract or author-specified keywords: "Murray-Darling Basin" AND ("water markets" OR "water recovery" OR "water policy" OR “water use”) AND ("economic impacts" OR "farm profitability" OR "impact" OR "economic value" OR “buyback”)
EconLit	TX "Murray-Darling Basin" AND ("water markets" OR "water recovery" OR "water policy" OR “water use”) AND ("economic impacts" OR "farm profitability" OR "impact" OR "economic value")

Source: authors’ own design.

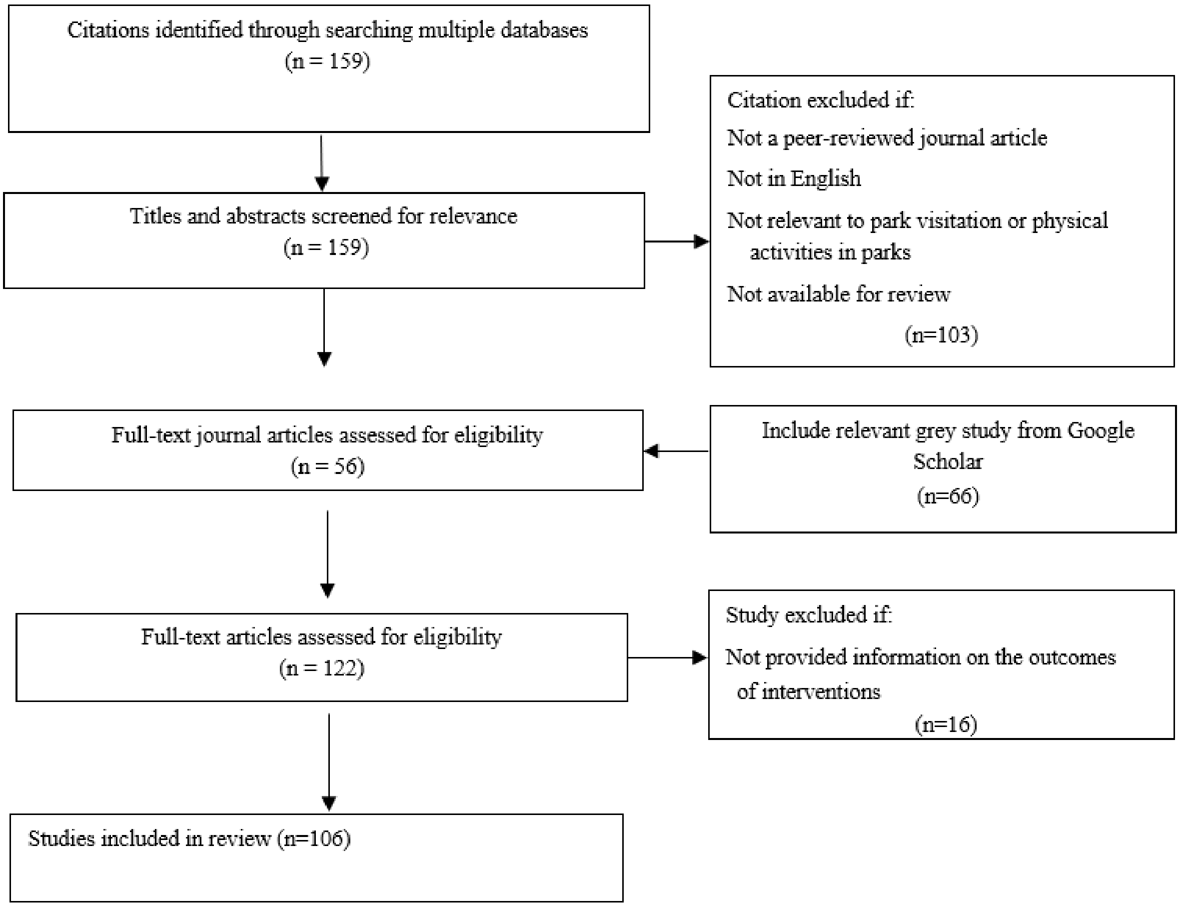
This stage of the search returned 159 journal studies. Reviewers then screened the titles and abstracts of these studies using the following criteria:

- In English
- A peer-reviewed and published journal article
- Full text available for review
- Relevant to the topic

Subsequently, 56 studies were identified after screening. We also conducted an extensive search afterwards to try and identify as much of the grey literature as possible. Another 66 relevant grey studies were found using Google Scholar, including reports, book chapters and working papers, which were also added to the review wherever available[11]. Although we have tried to review all existing studies, there were consulting reports that were not publicly available, and there are likely other relevant consulting studies that we missed.

Of the 122 studies in total, 16 studies were literature reviews or qualitative studies and therefore excluded. This resulted in 106 relevant studies focusing on the economic values in the MDB. A flow chart of the literature selection process is provided in Figure 3.1.

Figure 3.1 PRISMA Flow chart of the literature selection process



Source: Authors' own design.

[11] The grey literature and journal articles were all most collected prior to September 2022.

3.2 Analysis

To conduct the analysis, we collated information in a database, which included the following types of data:

(1) Metadata: including title, author, journal, published year, study period, state, studied region, dataset, and agriculture industry.

(2) MDB data: The drivers of water-related values in MDB data, which include water recovery programs (on-farm irrigation infrastructure, off-farm irrigation infrastructure, buyback), temperature, rainfall change, water trade, allocation %/storage, carryover %/volume, water entitlements owned, and water use/availability/flow. The impacts of these drivers on industry characteristics (e.g., farm number/farm size, industry production, the value of production), trade modelling (e.g., volumes, prices, participation), water use, environmental values and low characteristics, land use change and community broad socio-demographics (e.g., GDP, GRP, consumption, employment, population, tourism).

(3) Data collected for the strength of evidence: adapting the method used in Harrison et al. (2014) and England et al. (2020) and expert opinions, the strength of evidence was calculated based on a variety of factors. The index was structured so that an increase indicates higher quality. We had a rating system that was a) applied just to theoretical studies, and b) additional factors considered by empirical studies.

3.2.1 Rating system for theoretical studies

(a) General rating factors for both theoretical studies and empirical applied studies are described below, with a higher score indicating a higher quality assessment.

- **Blind/independent peer-reviewed** (3 = in journal, 1 = paid reviewer; 0 = no): If a study is published in a peer-reviewed journal, on average it has gone through six to eight reviews. The reviewers of the study include both: a) two external expert reviewers – (although four external expert reviews are not unusual) and b) reviews by the associate editor and editor of the journal. If not rejected in the first review round, the minimum amount of times reviewers will look at a study is two – but again – it is possible that authors will be asked to revise their work up to five times. High-quality journal outlets have a high rejection rate. In agricultural economics, only 6-18% of manuscripts submitted are actually published. For example, the *Australian Journal of Agricultural and Resource Economics* – the most published journal of the reviewed academic studies – had an acceptance rate of 15% in 2019 (Finger et al., 2022) (and its rejection rate has increased since). Most studies submitted to high-quality journals are therefore either desk rejected, or rejected after review. In addition, often submitting to top journals requires the provision of actual data and the code for reviewers to check results. Therefore, writing, reviewing and publishing usually take years. Hence, having a value of '3' as a quality index for publishing in a journal is very conservative;
- **Non-industry funded** (1 = yes, 0 = no): If a study has been funded by an industry body with a vested interest in the study's findings, there is the possibility that outcomes are overly positive, non-critical, and do not acknowledge methodological weaknesses. Indeed, this is a recognised issue in the medical field, where studies on the efficacy of drugs are often funded by pharmaceutical companies. A metareview by Jørgensen et al. (2006) found that industry-funded studies of drugs are less transparent and state few methodological limitations. This finding is supported by Bero (2013)

who finds that the number of studies reporting favourable results is 24% higher among industry-sponsored studies, and these studies showed less evidence of harmful drug effects. As such, non-industry-funded studies have a higher likelihood of unbiased results, and are therefore regarded as of higher quality, and again, an index value of ‘1’ is regarded as very conservative;

- **Observational studies with modelling controlling all confounders** (4 = yes, 0 = no): considers whether the studies controlled for as many possible controls, e.g., commodity prices, climate, policy, irrigated agricultural variables, spatial and locational variables OR for theoretical/hydro-economic models –- CGE models used (4 = yes, 0 = no): CGE is considered in this space of water recovery to be one of the most comprehensive and high-quality models given their dynamic nature (see Chapter 2 for an overview), though it is noted that different CGE models can have differing reliabilities;
    - **Observational studies with modelling –- but not controlling all confounders** (3=yes, 0=no) OR for theoretical studies, dynamic partial equilibrium models or hydro-economic models used 3 = yes, 0 = no): considers whether only some controls were included in the regression testing;
- **Descriptive statistics –- with significance testing** (2 = yes, 0 = no) OR theoretical models of input-output modelling or pure theory with no real data (2 = yes, 0 = no);[12]
- **Sources of information/data sources provided** to enable data collection (1 = yes, 0 = no);
- **All assumptions and functional forms reported** for analysis to allow replication (1 = yes, 0 = no).

Therefore, the maximum score for theoretical studies is 10, and the overall score of each study is the sum of the scores obtained from all questions. The overall grade quality of a study is therefore calculated as the following:

$$Overall\ grade\ quality\ of\ theoretical\ study = \frac{Overall\ sum}{Maximum\ score\ (10)} * 100\%$$

3.2.2 Rating system for empirical applied studies

(b) Rating factors specifically for empirical applied studies:

- Includes all above questions;
- **Sample Size** (0 = less than 100[13] (though the sample size can depend on population), 1=otherwise): the sample size has to be sufficient to draw statistically significant conclusions, and determined in reference to a population and sampling methodology;
- **All available data used** – within reason (1 = yes, 0 = no): includes both timeseries and cross-sectional information;
- **Random Sampling**. If surveys are conducted, are they representative (e.g., have high response rates, similar to the general population etc.) (1 = yes, 0 = no);
- **A variety of functional forms** tested for sensitivity analysis (1 = yes, 0 = no);

[12] As a reminder, many have highlighted the significant limitations of I-O analysis and suggest extreme care should be taken in any interpretation of its results (ABS, 2021; NSW Treasury (2017). A score of ‘2’ for input-output analysis is on the generous side for this form of modelling.

[13] A minimum observation to predictor ratio of 10, with a minimum sample size of 100 or 50, is recommended in multivariate analyses (Marascuilo & Levin, 1983; Tabachnick & Fidell, 2001).

- **Basic regression tests** done (e.g., heteroscedasticity, autocorrelation, test statistics reported on regression) (1 = yes, 0 = no);
- **Causality identified** – where regression type enables estimating a causal relationship and endogeneity is controlled for (1 = yes, 0 = no).

Where a particular rating factor was not applicable to a study (for example, if the empirical study was not survey-based, then the rating system of 'random sampling' was not applicable), then this value was removed from the total maximum score. The total maximum score for empirical applied studies was 16 – or 15 if no survey data was used. The overall score of each study is the sum of the scores obtained from all questions. The overall grade quality of a study is therefore calculated as the following:

$$Overall\ grade\ quality\ of\ emprical\ study = \frac{Overall\ sum}{Maximum\ score\ (15\ or\ 16)} * 100\%$$

3.2.3 Overall ranking system

For both the theoretical studies, and the empirical applied studies, we ranked the studies in the following way:

- an overall grade of 0 to 40% was classified as ***low quality;***
- an overall grade of 41% to 70% was classified as ***middle quality***; and
- an overall grade of 71% to 100% was classified as ***high quality***.

4 Result Summary of the Systematic Literature Review

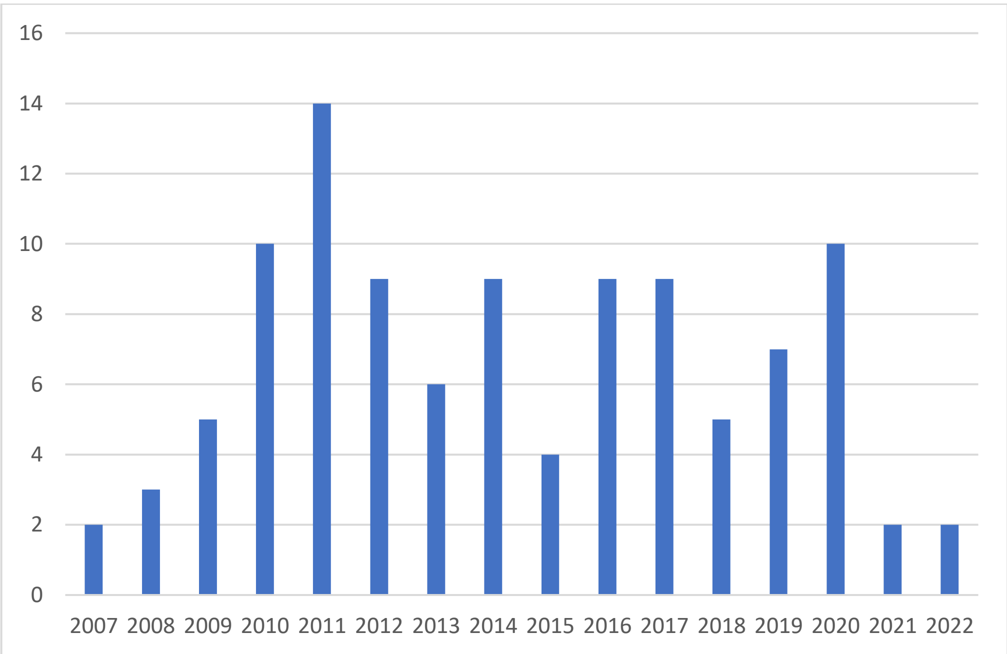
4.1 An overview of selected publications

It is important to note that our studies try to cover as much information available, within the limits acknowledged previously. The studies chosen within our systematic literature review focus primarily on quantification of some aspect of water recovery (or scarcity) on various economic outcomes.

4.1.1 Timing

The quantitative analysis of the economics of water issues in the MDB has increased over time. Unsurprisingly given that we started our search from 2007 onwards when water recovery first started, the first study in our database was written in 2007, and from 2007-2011 there was an increase in relevant studies. The publication number is relatively stable during 2012 to 2020. However, the number identified declined from 2021 onwards. Figure 4.1 illustrates the trend over time.

Figure 4.1 Annual publications over time



Source: Authors' own multi-database literature search.

4.1.2 States

The selected publications encompass a total of 4 regions, namely NSW, SA, Victoria, and Queensland (Table 4.1). The majority of studies focus on the entire MDB, and then the southern MDB, followed by Victoria, NSW, and SA.

Table 4.1 Geographical coverage of publications

Study region	Number of studies
All MDB	52
Southern MDB	32
Northern MDB	2
NSW	7
Victoria	8
SA	5

Source: Authors' own multi-database literature search.

4.1.3 Study types

For ease of comparison, we broke down the studies into two broad categories:

1. **Theoretical**: optimisation and mathematical modelling models; economy wide models (CGE) to partial equilibrium to mathematical programming to input output, benefit cost analysis and scenario analysis;
2. **Applied**: timeseries econometrics, panel econometrics, cross-sectional analysis econometrics and descriptive statistics.

Note, there is some overlap between both methods, especially where some theoretical models use empirical data.

Table 4.2 reports the methods of selected studies (n=106). There are 54 studies that used theoretical models broadly, while 52 used applied methods. Among the theoretical studies, most studies (37%) applied mathematical programming. 24% used general equilibrium models, 22% of studies used the partial equilibrium hydro-economic or hydrologic models, and 17% applied other theoretical methods such as cost-benefit analysis, scenario analysis and input-output modelling. Among the applied studies, about half (44%) used panel or time-series data, the other 29% used cross-sectional or pooled cross-sectional data, and 27% used descriptive statistics only.

Overall, the theoretical studies have higher quality than applied studies – which is driven mainly by the lower-ranked studies in the descriptive section of the applied studies. The percentages of high-quality studies are quite close, 39% and 41% among theoretical and applied studies, respectively. However, middle-quality studies account for a higher percentage among studies using theoretical models (41%) than among studies that use applied models (19%), while low-quality studies account for a lower percentage among theoretical studies (20%) than among studies that use applied models (40%).

In subgroups of study methods, high-quality studies account for the highest percentage of the studies using panel or time series analysis, which is 57%. The next is studies using mathematical programming, of which high-quality studies account for 55%, studies using cross-sectional or pooled cross-sectional econometrics, of which high-quality studies account for 53%, studies applying partial equilibrium hydro-economic or hydrologic models, of which high-quality studies account for 42% and general equilibrium studies account for 38%. In contrast, studies using descriptive statistics, and other methods (e.g., cost-benefit analysis, input-output) generally have relatively lower qualities. Specifically, these models have no high-quality studies, while lower-quality studies account for more than three-fourths (78% in theoretical and 86% in applied).

Table 4.2 Study methods and numbers of reviewed studies by quality ranking (n=106)

Groups of Study method	Subgroup of study methods	Subgroup numbers and qualities				Group numbers and qualities			
		Number of studies	High-quality studies	Middle-quality studies	Low-quality studies	Number of studies	High-quality studies	Middle-quality studies	Low-quality studies
Theoretical	General equilibrium model	13	5 (38%)	8 (62%)	0 (0%)	54	21 (39%)	22 (41%)	11 (20%)
	Partial equilibrium hydro-economic / Hydrologic models	12	5 (42%)	4 (33%)	3 (25%)				
	Mathematical programming	20	11 (55%)	8 (40%)	1 (5%)				
	Other (BCA, input output, scenario etc)	9	0 (0%)	2 (22%)	7 (78%)				
Applied	Panel/time-series econometrics	23	13 (57%)	4 (17%)	6 (26%)	52	21 (41%)	10 (19%)	21 (40%)
	Cross- sectional/pooled cross-sectional econometrics	15	8 (53%)	4 (27%)	3 (20%)				
	Descriptive Statistics	14	0 (0%)	2 (14%)	12 (86%)				
Total studies		106	42 (40%)	32 (30%)	32 (30%)	106	42 (40%)	32 (30%)	32 (30%)

Note: % reported in parentheses; only the main methods used in each study are reported, although multiple methods may be used in a study.
Source: Authors' own multi-database literature search.

4.2 Drivers of water-related values in the MDB and various economic values studied

4.2.1 Drivers of change in the MDB (independent variables in models)

The purpose of this section is to understand the extent of the literature that has looked at various water explanatory variables on economic outcomes in the MDB. Explanatory water-related variables were grouped into four categories:

1. ***Water reform*** (e.g., water entitlements recovered through buybacks and infrastructure investment (GL) – both together and separately, where modelled as such);
2. ***Climate*** (temperature, rainfall, drought);
3. ***Water trade and ownership*** (permanent and temporary trade; entitlement ownership); and
4. ***Water availability*** (water consumption, water supply, water allocations, water storage, carryover.

It is important to note that we did not review the full water trade literature – it was only possible to try to identify studies that had modelled water sales to the government (rather than just water sales in the normal market).

As shown in Table 4.3, most studies are relevant to water recovery programs (44%), while about half (51%) of which specified the way of water recovery, either through on-farm/off-farm irrigation infrastructure or through buyback. About 22% of reviewed studies examined water trade and its impact, and 20% examined climate change and its influence.

Table 4.3 Numbers of studies in the MDB by independent factor/driver of economic outcomes

Group of independent factors of change/drivers	Subgroup of independent factors of change/drivers	Number of studies	Number and % of groups[2]
Water recovery programs	Water recovery programs (unspecified)[1]	34	70 (44%)
	Water recovery programs through on-farm/off-farm irrigation infrastructure	21	
	Water recovery programs through Buyback	15	
Climate	Temperature	6	31 (20%)
	Rainfall change	25	
Water trade variables/ownership	Water trade (e.g., volumes, price)	26	34 (22%)
	Water entitlements owned	8	
Water availability	Water use/supply /flow	8	23 (15%)
	Allocation %/Storage	13	
	Carryover %/vol	2	

Note: [1] Water recovery programs (unspecified) include water recovery program studies, which do not specify the way for water recovery.
[2] % reported in parentheses. Some studies are double counted as multiple drivers are examined in these studies.
Source: Authors' own multi-database literature search and synthesis.

4.2.2 Economic outcomes studied in the MDB (dependent variables)

Our definition of MDB economic outcomes of importance came up with five main categories. Some of these outcomes will be regarded as more important than others, depending on people's preferences (Table 4.4). They include:

1. ***Industry Characteristics*** (e.g., farm numbers, farm size, industry area, industry production, productivity, GVIAP, GVAP);
2. ***Water trade modelling*** (e.g., volumes, price, participation);
3. ***Water use, environmental values, and flow characteristics*** (e.g., volumes, price, participation);
4. ***Land use irrigated/ change*** (e.g., irrigated or dryland area change, annual or perennial agriculture land use change, land change among crops which need different amount of irrigation water); and
5. ***Community broad socio-demographics*** (e.g., GDP, GRP, GSP, consumption, employment, population, tourism, recreational, mental health).

Among all the studies, the influences of drivers on industry characteristics (e.g., farm numbers, farm size, industry area, industry production, productivity, GVIAP, GVAP) account for the highest percentage, which is 34%. A majority of these studies (46 out of 73) examine the influence on value of industry production and profits. Besides industry characteristics, the influences on community broad socio-demographics (e.g., GDP, GRP, Consumption, employment, population, tourism, recreational, mental health) are also broadly investigated, which account for 23%. Most of these studies examine the influences of various drivers on consumption/employment/population (25 out of 49), and GDP/GRP (20 out of 49). Moreover, the influences on water use, environmental values and flow characteristics account for 20%, influences on trade modelling (e.g., volumes, price, participation) account for 15%, and influences on land use change (e.g., irrigated or dryland area change, annual or perennial agriculture land use change, land change among crops which need different amount of irrigation water) account for 8%.

Table 4.4 Numbers of studies in the MDB per influence of dependent factors/drivers

Groups of influences	Subgroups of influence	Number of studies	Number and % of studies
Industry Characteristics	Farm numbers/ farm size	6	74 (34%)
	Industry area	9	
	Industry production/productivity	12	
	Value of industry production/profits (GVIAP, GVAP)	47	
Water trade modelling (e.g., volumes, price, participation)	Water volumes	8	34 (15%)
	Water prices	16	
	Water market participation	2	
	Water use efficiency	8	
Water use, environmental values, and flow characteristics	Water flow	4	42 (20%)
	Water use	24	
	Water availability	5	
	Environmental values	9	
Land use irrigated/change	Irrigated or dryland area change and other changes (e.g., annual or perennial agriculture land use change, land change among crops which need different amount of irrigation water)	16	16 (8%)
Community broad socio-demographics	GRP/GDP	20	49 (23%)
	Consumption/employment/population	25	
	Tourism/recreational	2	
	Others (Indigenous business, business diversity, mental health)	2	
Total[1]		213	

Notes: [1] Some studies are double counted as multiple influences are examined in these studies.
Source: Authors' own multi-database literature search and synthesis.

5 Literature review of the influences of water recovery programs on economic outcomes

Among all the drivers, ‘water recovery programs’ is the factor most widely studied (accounting for 44% shown in Table 5.3). It is also found to be significantly associated with diverse economic values within the MDB. However, the impacts of programs such as water entitlement buyback and on-farm/off-farm infrastructure investment are the most disputable, warranting further investigation.

Therefore, the following chapter provides a summary of the literature that in particular has tried to investigate the impact of water entitlement buyback (*Restoring the Balance Program*) or on-farm and off-farm infrastructure investment (*Sustainable Rural Water Use and Infrastructure Program*) on various economic outcomes in the Basin. Our literature review is provided by type of quantitative study utilised. We start with theoretical studies, which mainly focus on modelling at an area or regional level, to give insights for either the whole Basin, or parts of the Basin. We then move to econometric studies, of which some focus on average or regional statistics in modelling regional or Basin-level economic impacts, while others employ individual-unit data (e.g., individual survey or records) to model impact at the individual level (considering regional factors). We finish by describing descriptive studies, that provide both regional and individual outcomes – using less rigorous quantitative methods. Table 5.1 illustrates the breakdown of studies by type.

The second half of the chapter provides a brief summary of some of the other economic literature (not exhaustive), which has attempted to establish a relationship between water diversion/reductions and socio-economic outcomes.

Table 5.1 Overview of quantitative modelling by type and quality ranking

Study method	Subgroup of study method	Number of studies	High-quality studies	Middle-quality studies	Low-quality studies
Theoretical	General equilibrium model	11	4 (36%)	7 (64%)	0 (0%)
	Partial equilibrium hydro-economic model/Hydrologic model	9	3 (33%)	3 (33%)	3 (33%)
	Mathematical programming	9	6 (67%)	2 (22%)	1 (11%)
	Other (BCA, input - output, scenario analysis etc)	8	0 (0%)	1 (12%)	7 (88%)
Applied	Panel/time-series econometrics	10	2 (20%)	3 (30%)	5 (50%)
	Cross- sectional/pooled cross-sectional econometrics	7	2 (29%)	2 (29%)	3 (43%)
	Descriptive statistics	11	0 (0%)	1 (9%)	10 (91%)

Note: Only the main methods used in each study are reported.

5.1 Theoretical studies on the influences of water recovery programs

We break down theoretical studies into computable general equilibrium models; Partial equilibrium hydro-economic/hydrologic models; mathematical programming; input-output; benefit cost and scenario modelling.

5.1.1 Computable general equilibrium models – water recovery studies

Comparatively, more water recovery program studies (37 in total) use theoretical models for analysis (see Table 5.1). Among these, 11 studies use general equilibrium models, which are known as either computable general equilibrium model (CGE) or dynamic general equilibrium models.

5.1.1.1 Buyback program evaluation – Community impacts

Several CGE studies have investigated the different impacts of buybacks (Dixon et al., 2009; Wittwer, 2010; Wittwer, 2011a; Wittwer, 2011b; Dixon et al., 2011; Wittwer and Young 2020). The predominant CGE model used is TERM-H2O, where TERM = The Enormous Regional Model, and is a ‘bottom-up’ CGE model of Australia which treats each region as a separate economy. It is able to handle a great number of regions or sectors, e.g., 172 sectors in 206 statistical sub-divisions, allowing the model to accurately represent water catchment, major urban or tourism regions, as well as farm water use. Dixon et al. (2009) modelled the Australian Government’s buyback scheme using TERM and showed that the impact of the buyback on regional economics was quite small. Specifically, the impact of buyback on regional GDP was estimated to be -0.33% across the southern MDB. The employment impacts were also small, while the largest percentage loss at the regional level by 2018 was around 0.2%. Their model also showed that cuts in water allocations arising from drought have much greater impacts on regional economics within the sMDB than buyback. Wittwer (2010) and Wittwer (2011a) highlighted that buyback would not be as detrimental to agriculture in the MDB as drought, and found that the impact of buyback on regional communities would be relatively modest through modelling Sustainable Diversion Limits (SDLs) with different targets. Wittwer (2010) showed that if farmers are compensated at the market price for water removed from production under SDLs, the impact on real GDP across the MDB will be negative but small (less than -0.3% from the forecast), and the reduction of employment is at most -0.07%.

In addition, Wittwer (2011b) argued that communities conflated the impacts of buybacks and drought while there had already been job losses in the basin due to drought. Using the CGE model, it showed that drought is more severe in the regional impact than buybacks – even without considering the positive impact of buyback proceeds on regional communities. The dynamic CGE modelling results showed that around 6,000 jobs were lost due to drought in MDB from 2006-07 to 2008-09, but only 500 jobs were lost across the basin due to buybacks. It also showed that buybacks only lead to 0.3% of GDP loss, but drought lead to a 5.7% GDP loss in MDB. Dixon et al. (2011) analysed the effects of government buyback water from irrigators in the sMDB and found that water prices would rise substantially if 1,500GL of irrigation water (about 23% of supply in a normal year) were diverted to environmental uses. However, rather than causing a sharp reduction in farm activity in the sMDB, they found that the increase in the price of water would cause a reallocation of farm resources between activities. For example, in all sMDB regions, some irrigable land would change to dry-land uses and dry-land farming would expand relative to irrigation farming. Also, with more expensive water, irrigation-intensive regions would move from crops that use a large amount of irrigation water per hectare of irrigable land to crops that use lower amounts. Their model also suggests that buyback would increase economic activity in the sMDB (e.g., positive effect on household consumption), while there is little effect on aggregate sMDB farm output and national macro effects on GDP. Specifically, their model indicates that the long-run reduction in GDP caused by a 1,500GL scheme is 0.006%.

Moreover, ABARES (2010c) used a combination of methods, and their ABARES Water Trade Model, to estimate the economic effects of buyback on irrigated agriculture, and then used the ABARES AusRegion model (a CGE model), to estimate the flow-on effects to regional economies. The model results suggest a substantial increase in entitlement prices (13% and 18% in the nMDB and sMDB, respectively) compared with the case in the absence of buyback. Moreover, it suggests the negative effect of buyback on GVIAP (-2.4%), water use (-5.1%) and irrigated land use (-1.6%). The AusRegion results suggest that the broader economic effect of the buyback is almost indistinguishable at the national level (less than 0.01% of GDP, and 0.1% or less of GRP).

ABARES (2010a) focused on estimating the economic impacts of the full reduction in water availability relative to long-run historical levels imposed by the SDLs, using a two-stage modelling approach, ABARE–BRS's Water Trade Model (WTM) and the AusRegion model. It suggests that the 3,500GL Basin Plan reduces the gross value of irrigated agricultural production (GVIAP) by -16.5%, a profit by -8.2%, and reduced water use by -29.1% with no interregional trade allowed. The reduction decreases when interregional trade is assumed. Also, they highlight that the overall effects at a broad regional level are likely to be small relative to the total size of these regional economies (GRP reduced by 1.3% and GDP reduced by 0.13%). ABARES (2010b) also applied the same two-stage modelling approach and modelled the effect of the Basin Plan (3,500GL SDL option) only (Scenario 1) and the combined effect of the Basin Plan with other mitigating policies (e.g., *Water for the Future,* and additional water purchases) (Scenario 2). The model further shows that Australian Government actions under the *Water for the Future* initiative and other water purchases could mitigate the negative effect of the Basin Plan on GVIAP (an increase from -15% to -10%), profit (an increase from -8% to -5%) and Basin-wide GRP (an increase from -1.3% to -0.7%). The decrease in water use also declines with water purchases (from -29.7% to -20.5%). However, the estimated employment effects of changes in employment are estimated to be much smaller than changes in GRP under the two scenarios. ABARES (2011a) modelled several Basin Plan policy scenarios while assuming different sizes of the SDLs using the same models as ABARES (2010a). Consistent with previous ABAREs analysis, it shows the reductions in GVIAP driven by reductions in irrigation diversion. The effects were estimated to be relatively small for the Basin economy as a whole, although the impacts could be significant for smaller local-level areas that are highly reliant on expenditure from irrigated agriculture.

ABARES (2011b) also employed the ABARES Water Trade Model, to estimate the economic effects of reduced water availability on irrigated agriculture, and the ABARES AusRegion model, to estimate the flow-on effects to regional economies. However, this study introduced a number of refinements to the methodology and assumptions by considering water entitlement purchase and infrastructure investment by governments. The model shows that SDLs with either buyback or infrastructure reduce water use, land use, GVIAP, and profit and influence basin macroeconomics by reducing GRP, household consumption and employment.

ABARES (2016) used a simulated model of the water market to model a 'without environmental recovery' scenario, where environmental purchases were ignored. The model was run annually, for the main regions in the trading zone of the sMDB, from 2000-01 to 2014-15. The variables in the model only included: allocation, rainfall, with the dependent variable water allocation price. ABARES (2018) presented an econometric partial equilibrium model of water trade and irrigation combining econometric estimation of water demand with bio-economic optimisation models. Variables of commodity prices, water prices, rainfall and time were included in yearly models from 2002-03 to 2016-17 in NRM areas in the sMDB (9 regions), by industry. It was found that in general, total area of irrigation contracts as water prices increase, with higher value activities less sensitive to changes in price in comparison with lower value activities like pasture.

ABARES (2020b) built upon the model in ABARES (2018) and modelled a series of forward-looking scenarios for the sMDB water market, namely 1): *current irrigation development* (horticultural plantings), current water recovery under the Basin Plan, current trade rules and commodity prices; 2): *future market*: Full maturity of recently established almond plantings, and future water recovery to meet Basin Plan requirements (3,200GL target) via on-farm infrastructure upgrades; and 3): *future market (dry):* as in the future market scenario, but with an 11% reduction in water supply and a 3% reduction in rainfall), examining future water prices, trade flows and irrigation outcomes. Key findings included:

- *Higher water prices*: a significant increase in average water allocation market prices is estimated across the sMDB, with a 28% (50% increase in allocation prices in the future market scenario (future market (dry) scenario).
- *Inter-regional trade limits impact*: growth in water demand in the lower Murray due to maturing Almonds trees (particularly in NSW and SA Murray), leads to greater pressure for inter-regional water trade, more frequently binding trade limits and large differences in prices between regions.
- *Growing demand from horticultural plantings in dry years*: Water supply (including both surface water and other sources such as groundwater) is predicted as sufficient to meet estimated demand from horticultural plantings (fruits, nuts and grapevines) in all scenarios, but some supply shortfalls will persist. Horticultural plantings are estimated to use around 1,276GL on average each year in the 'future scenarios.'
- *Reductions in water use and GVIAP in traditional irrigation sectors and regions*: water use in the dairy and rice sectors is predicted to decrease by 14-15% in the future market scenario and up to 55% and 32%, respectively, in dry year, with less decrease in GVIAP expected, with the decrease in other sectors partially offset by an increase in farm productivity and input substitution. Overall, the total GVIAP across all sectors is modelled to increase by 0.8% (4.1%) in the future market scenario (future market (dry) scenario) (ABARES 2020; v-vi).

5.1.1.2 Irrigation infrastructure – Community impacts

Several studies investigated the economic impacts of investment in irrigation infrastructure (e.g., Banerjee, 2015; MJA, 2017). Banerjee (2015) applied a CGE model to evaluate the economic impacts of investment in irrigation infrastructure in the Murrumbidgee. The regional production, household consumption, real GRP, capital stock, average real wages, income and employment were all found to increase. There was also a small negative impact on the national level, due to the transfer of the resources to the basin. Specifically, aggregate capital stock, exports, real investment, household consumption and real GDP were all negatively affected, except the imports and the average real wage. By comparing two different scenarios, in one of which 100% of water savings returned to the environment, and in another one 50% returned to the environment and 50% to irrigators, Banerjee (2015) found that the portion of water recovered and used for production generates further gains for the region. However, the magnitude was less than the economic activity stimulated by infrastructure investment. MJA (2017) used two approaches to estimate the impact of water recovery in the MIA: a static approach (disaggregating water recovery expenditures and identifying those which contribute to value added) and a 'dynamic' approach involving the use of a general equilibrium dynamic and multi-period model, which solves for both price and quantity – conducted using the VU TERM CGE (computable general equilibrium) model by Victoria University. They found the economic effect of the purchase (buy-back) program on the MIA was very small, if not neutral, because water was purchased at the prevailing market price. Positive effects on MIA employment (an increase of 168 full-time jobs) and real GDP (an increase of $178 million) in the local economy and also showed net gains in on-farm productivity due to on-farm technology investments.

Wittwer and Young (2020) is the latest economic modelling of water recovery, undertaken for the Independent Socio-Economic MDB Panel (Sefton et al., 2020). Wittwer and Young (2020) used an updated version of TERM-H2O to model two scenarios: 1) obtaining the remaining water recovery target through infrastructure only; and 2) removing the same amount of water entitlements and spending on regional services, instead between 2020 and 2024. It was found that scenario one had a net present value (NPV) welfare loss of $1.1 billion (but increased jobs up to 1,000 in the short term, and 100 in the medium term), while scenario two found that each dollar spent on education, health and community services created four times as many jobs as spending on infrastructure, and had an NPV welfare loss of $0.125 billion (nine times less than spending on infrastructure). Note, no welfare benefits to increased environmental water were allowed.

5.1.1.3 Buyback vs infrastructure comparisons – Community impacts

A few of the earlier studies compared the effects of buyback with infrastructure using CGE models (Wittwer, 2011a; Wittwer & Dixon, 2013). Wittwer & Dixon (2013) found that voluntary and fully compensated buybacks were much less costly than infrastructure upgrades as a means of obtaining a target volume of environmental water, even during drought, when highly secure water created by infrastructure upgrades is more valuable. However, their results indicated that infrastructure upgrades are inferior to public spending on health, education and other services in the Basin. For each job created from upgrades, the money spent on services could create between three and four jobs in the Basin. Wittwer (2011a) found that water savings arising from infrastructure upgrades were relatively more expensive than water purchases, as the decline of national GDP in the infrastructure scenario is larger than in the buyback-only scenario.

5.1.2 Partial equilibrium hydro-economic/hydrologic models – water recovery studies

Nine identified studies applied the partial equilibrium hydro-economic or hydrologic model to discover the influences of water recovery programs. Mainuddin et al. (2007) used a hydro–economic model and found the environmental water allocated based on maximizing profit leads to reductions in irrigated areas, but the overall economic profit remains almost unchanged and, as expected, the flows in the Murray River increase. However, they find if the environmental water is deducted proportionately from all sub-catchments irrespective of the economic value of water, the overall profit declines. GHD (2012) also applied hydrology modelling and spatial analysis and suggested that a reduction in diversions of 2,750GL/year would reduce the incremental value of floodplain agriculture in MDB from $313 million to $32 million. Grafton and Jiang (2011) constructed a hydro-economic model in 19 regions across the MDB. Results indicated that substantial reductions in surface water extractions impose only a moderate reduction on net profits in irrigated agriculture (reduce 10% for 3,000GL/year, and about 17% for 4,000GL/year), the effects are nevertheless substantial in specific regions/catchments. Kirby et al. (2014) used an integrated hydro–economics model to model three levels of reallocation (2,400, 2,750 and 3,200GL) under the historical climate, and under a dry, a median and a wet climate change projection. The analysis results indicated that estimated river flows and diversions are more sensitive to the range of climate change projections than to the range of diversion reallocation scenarios considered. Also, the reduction in economic returns to irrigation was less than the reduction in water available for irrigation. Specifically, Kirby et al. (2014, table 1, p. 157) compared actual farming outcomes in the MDB from 2000–2001 to 2007–2008 and found that the real adjusted gross value of irrigated production fell by just 10%, despite a 70% decline in irrigated surface-water use, again highlighting the importance of trading in dealing with water scarcity.

Notwithstanding the negative impact on irrigated agriculture, there is evidence of the positive effect of environmental outcomes and value. Akter et al. (2014) combined a hydro-ecological model with stated preference models to estimate the economic value of environmental water in Macquarie Marshes. Using these methods, this study shows that under a scenario of 135GL of environmental water released per water year, a GL of water used in the environment appears to generate a higher economic value for the environment than its alternative use in the agricultural sector.

Williams and Grafton (2019) estimated the impact of water infrastructure subsidies. This study employed published water balance data from irrigated cropping and showed that water recovery through infrastructure subsidies has resulted in smaller increases in the net stream and river flows than is estimated by the Australian Government – and may even have reduced net stream and river flows. Wang et al. (2018) provided a review of estimates of return flows across the MDB.

Burdack (2014) provided an economic impact analysis of different pricing and non-pricing water management policies on irrigated agriculture in the MDB using a linear Water Integrated Market-Model (i.e., a partial equilibrium modelling framework (WatIM-Model)). Results showed that the impact was most severe on water intensive crops in the case of mandatory non-pricing water management policies. The study further confirms that water is reallocated from less (rice, cotton) to higher water-productive crops (vegetables, fruits, grapes) in cases of higher water prices and lower water availability. Thus, high water-value crops are less affected by different water management policies.

Unlike other studies, ABARES (2020b) used the ABARES Water trade model to estimate potential future water prices, trade flows and irrigation activity, while assuming the future market scenario in which water recovered under the Basin Plan is completed in full and a further 501.6GL of water rights (in LTAAY terms) is recovered across the Basin. The main findings include a significant increase in average water allocation market prices, reductions in water use in some traditional irrigation sectors (e.g., dairy and rice sectors) and regions (e.g., Goulburn-Broken region, Murrumbidgee), and a decrease in GVIAP for traditional irrigation sectors.

5.1.3 Mathematical programming studies – water recovery studies

Some studies use other mathematical programming models to investigate the impacts of water recovery. For example, Ramilan et al. (2011) focused on economic trade-offs from diverting water to the environment from irrigated agriculture under different scenarios in the Broken River catchment of northern Victoria. It showed the existing of opportunity cost of forgone agricultural profit under all climate conditions, which is especially very high during dry season. Qureshi et al. (2007) provided an economic analysis of reallocating water from agriculture to the environment with and without the possibility of interregional water trade, by applying a water allocation optimisation model. This study concluded that irrigation net revenue would be expected to decline if water for environmental flows were acquired through reductions in irrigation water allocations and free trade between regions was not allowed. However, if trade is allowed, the net revenue gains are estimated to outweigh the negative revenue effects of reallocating water for environmental flows.

Mallawaarachchi et al. (2010) applied a water allocation model, which is a regional programming model to compare the economic returns from irrigation for the Baseline scenario that represents the current diversion limit (CDL) and the Basin Plan Cap scenario that incorporates sustainable diversion limits (SDL). The results of the model indicate that under the Basin Plan, the 3,746GL water diverted to the environment would lead to a fall in the gross value of irrigated production (reduced by 16%) and a 16% fall in regional profit. Following on from this study, Adamson et al. (2011) estimated the potential changes to Basin-wide irrigation systems and river health before and after the introduction of sustainable diversion limits. The results indicate that 2,900GL transferred to the environment, either with or without trade, would decrease the gross value of irrigation and economic return. However, trade could mitigate the reduction – similar to the finding made in Mallawaarachchi et al. (2010).

Also, a few studies used mathematical programming models to investigate the impact of investment in infrastructure or buyback. Qureshi et al. (2010) find that when water recovery programs involve water savings being split between irrigators and the environment and high rates of return flows, efforts to generate water for the environment through increases in irrigation efficiency can actually reduce net water available for the environment substantially.

Adamson and Loch (2014) used stage-contingent modelling to review farm capital investment policy in the MDB and explore technical efficiency implications under different states of inflow variability. Building on Adamson et al. (2011), the study employed an updated version of the Risk and Sustainable Management Group MDB Model which maximised possible returns from 21 irrigation activities, one dryland production activity, and water diversion for Adelaide potable supply across three states of climate (wet, normal, drought), by allowing a water planner to allocate all possible resources across the basin. To incorporate the effects of irrigation infrastructure policy, model formulas were modified to reduce the cost of capital, change water use, and associated variable costs, and reduce return flows. Results showed two negative effects of environmental water recovery through on-farm capital investment in irrigation infrastructure. Such investments may encourage the shift from flexible annual to inflexible perennial production systems, which fail to respond to future water scarcity and hence increase exposure to climate risk. The technical efficiency gains from infrastructure upgrades may reduce return flows, which reduces water supply for downstream users and the environment, jeopardising Basin Plan's environmental objectives.

Loch and Adamson (2015) used mathematical programming and showed that increased efficiency through water-efficiency technology can result in rebound effects. Specifically, with greater efficiency, irrigators switched to perennial cropping systems under irrigation infrastructure subsidization incentives, leading to higher consumptive land and water demand. It also paradoxically reduced environmental flow volumes.

Adamson et al. (2017) assessed annual and perennial irrigators' water demand responses and adaptations to drought, normal, and wet future climate conditions, through a state-contingent analysis model. Short-run and long-run choke prices were used to model an individual irrigator's water demand in different states of water scarcity, based on their annual/perennial production system. Results showed that a drier climate in the form of a reduced average water supply is preferable to increased future water supply variability, as producers can adapt by reducing land allocated to production. Perennial producers are less flexible in adapting to changing climate conditions, as they attempt to keep plant capital alive, leading to high short-run choke prices for water. In the longer run, these short-term prices cannot be sustained, and perennial producers will only be able to afford long-run choke prices for water without going bankrupt. Annual producers have greater flexibility in their water demand due to the absence of permanent plant capital, can therefore adapt to future water supply scarcity much more cheaply than perennial producers.

Adamson and Loch (2018) extended the analysis in Adamson and Loch (2014) to evaluate welfare trade-offs of recovering environmental water solely through irrigation infrastructure upgrades instead of water buyback. A state-contingent welfare constrained model was used to investigate how restrictions on water buyback alter water recovery outcomes. Similar to Adamson and Loch (2014), one basin-wide water manager maximised net private returns from irrigation across 21 basin catchments under Basin Plan water quality and recovery constraints and different climatic states of nature. The model shows buyback is more economically efficient to recover water than technically efficient irrigation infrastructure, as operational, maintenance, and water delivery costs are higher for improved irrigation technology, and as water recovery through irrigation infrastructure investments requires higher annuity payments than the water market price. Further, the model predicts large capital losses for any return to drought conditions if irrigation investments result in more perennial production systems, which are more vulnerable to future water scarcity.

5.1.4 Input-output/scenario analysis – water recovery studies

Some studies used input-output methods to examine the impacts of water recovery or water for the future plan programs. RMCG (2016) used a mix of methods, input-output being one, as well as scenario analysis and simple descriptive analysis of the relationship between water recovery and milk production, and suggested water recovery programs through buyback would double water price. This study was commissioned by the GMID water leadership forum to estimate the socio-economic assessment of the Basin plan, particularly for GMID. The assumptions on the reduced milk production were fed into economic impact modelling by EconSearch. RMCG (2016; 36) stated a counterfactual of a drop in 234GL in water available for dairy use in the GMID due to the Basin Plan, and that this translated directly into 440ML of lost milk production (based on average amounts of water needed per dairy cow). They then assumed an 'average' milk price, and claimed this to be a reduction in the annual farm-gate value of dairy production of $200 million, with the mixed farming and cropping sector losing a value of $25 million – input-output is then used to claim a total revenue loss of $580 million per year, together with a loss of 1,000 jobs across the region. This report assumed a direct linear relationship between water use and milk production (ignoring surplus water, on-farm resource movements, other adaptation measures), with Wheeler et al. (2018a) providing a detailed account. Other problems were also raised by the SA MDB Royal Commission (Walker, 2019).

TC&A and Frontier Economics (2017) analysed the impacts of the Basin Plan in Victoria. The authors acknowledged the impact assessment should not be a comparison of outcomes before and after the Plan, rather a comparison of what happened after the Plan with what could reasonably have been expected to have happened if the Basin Plan had not been implemented. The study found that the money the Commonwealth paid irrigators for their entitlements, or for their on-farm efficiency measures, helped these irrigators to adjust to the drought. While water use by horticultural farms was largely unchanged with and without the Plan, the study found that if water recovery had not occurred, water use in the GMID would have been 29-31% higher from 2013/14 to 2015/16, and GMID milk production could be expected to have been about 30% higher than was observed. The foregone production was expected to have had significant flow-on effects in the region where farm inputs are sourced, and processing and manufacturing occur.

Frontier Economics and TC&A (2022) updated the 2017 analysis. Frontier Economics and TC&A (2022) estimate a 'counterfactual case', and state that if it had not been for buyback (adjusted for trade), then water diversion would have been 46% higher in the GMID. However, this is an assumption only and not borne out by the figures reported in Figure 20 of the same report which suggests around a 25-30% decrease in their 'counterfactual'.

Assuming a portfolio mix similar to what the CEWH currently holds, the average annual costs in lost production would be greater than $400 million per year in the sMDB. They then suggest that buying back an additional 372.3GL to meet the 2,750GL requirement would, based on the CEWH's existing portfolio, reduce the consumptive pool of higher reliability entitlements by 209GL. If an additional 760GL in total (372GL for 'Bridging the Gap' plus 388GL for Efficiency Projects) were to be recovered via buyback, in line with the CEWH's existing portfolio, the average annual foregone production would be over $850 million per year. The study recommends the Basin Plan implementation to focus on current or alternative SDLAM projects to offset the full 605GL in a timely manner (rather than by the current 30 June 2024 deadline). There are many problems with the basic assumptions used in this study (discussed later in Chapter 7).

RMCG (2021) applied simple descriptive methods to investigate the impact of additional water purchases on both irrigated agricultural production and water prices within the sMDB. They assume market premium costs of between 14-28%, and also assume a direct linear relationship between water recovery and falls in water use. This study argued that, following the purchase of 120GL and 605GL, annual irrigation production gross margin values are estimated to fall by an average of $54.8 million and $276 million per year, respectively over the following ten years. The future water allocation price and entitlement values were also found to increase. The problems with this study are that it is based on incorrect assumptions (e.g. the assumed gross margins for a variety of crops, no adaptation allowed, and incorrect price premiums for Commonwealth buyback applied). For example, their estimate of market premium costs is way off. They 'estimate' an assumed average buyback price' – but ignore published data on this (e.g., Wheeler & Cheesman, 2013, Table 3; ANOA 2011 Appendix 5, plus data available at webarchive.nla.gov.au/awa/20191115082623/https://www.agriculture.gov.au/water/markets/commonwealth-water-mdb/average-prices), all of which show that the estimated 'premium paid' is grossly overestimated, and very little premium, if any, actually was paid by the Commonwealth in the buyback program.

5.1.5 Benefit Cost Analysis (BCA)

Given the difficulty in performing Benefit Cost Analysis (BCA) to evaluate the impacts of the Basin Plan, it has been rarely used (Wheeler, 2014). CIE (2011) employed BCA to evaluate a range of both the benefits and costs of the proposed SDLs. It highlighted the value of decreased employment, irrigated agricultural production and economic surplus and also evaluated recreational and other benefits (e.g., salinity) due to the introduction of the SDLs. The report suggested that the extent of the benefits and the costs achieved from SDLs depend on the package of reforms, and the interrelationships between environmental water policies, water sharing plans, entitlement buybacks, infrastructure investments, market trading and structural assistance policies. One of the main benefits of water recovery identified was environmental benefits.

Wheeler (2014) provided a summary of the quantified benefits of the Basin Plan, and undertook a heuristic assessment of the returns, to suggest that the benefits may outweigh the costs by up to three times.

5.2 Applied modelling studies on the influences of water recovery programs on economic outcomes

Twenty-seven studies were identified as examining the influence of water recovery programs using the applied methods.

5.2.1 Timeseries regressions – water recovery studies

5.2.1.1 Irrigation area, GDP, GRP, water use studies

Wentworth Group of Concerned Scientists (2017) undertook a review of water reform in the MDB. As part of this, under Appendix 3, was an analysis on the trends in social-economic indicators (i.e., agricultural production, water use and efficiency, commodity prices, population size and density, labour force and population and employment of indigenous communities) and their influences in the MDB. OLS regression was used to model from 2001-2015 (though sample sizes varied from 10-15 observations only). Results vary spatially and showed that drought was a major driver of changes throughout the 2000s – with water reforms, water prices and other factors playing a larger role more recently. Results found the following: a 1% increase in commodity prices increased GVIAP and GVAP by 0.8% and 0.6%, respectively; drought during 2006-09 reduced GVIAP by 23% and GVAP by 18%; a $1 increase in the water allocation price reduced GVIAP by 0.06% and GVAP by 0.05%; a 1GL increase in water recovery reduced GVIAP by 0.04%; and was insignificant on GVAP. For irrigated areas in the MDB, a $1 increase in water allocation reduced area irrigated by 1,300 hectares, and a 1GL in water recovery reduced irrigated area by 800 hectares, while a 1GL increase in water allocations increased irrigated area by 20 hectares. No significant effects were found on the number of agricultural businesses in the MDB, while a 1GL increase in water allocation reduced water allocation price by $0.04. A 1GL increase in water recovery reduced total water extraction by 8GL. Overall, the quality of the analysis was low, mainly due to the low sample size and the lack of controls in many of the regressions.

MDBA (2016c) estimated several regression models of hectares under cotton production in the nMDB. KPMG (2016) employed a pooled cross-sectional regression approach to analyse the aggregated census data at the sectoral level (ten industry sectors in total) across the nMDB. The annual changes in employment from 1999-00 to 2013-14 were examined in 21 communities (five without water recovery) within nMDB. Each of the 21 communities was modelled separately and simulations quantified the impact on the community jobs following a change to the community's surface water availability. The model for each community used a historical baseline extending from 1999 to 2013. The impact of the water recovery scenarios on aggregate employment was found to be negative and relatively small for most communities, but the %-reductions in aggregate employment are larger in the wet years. Wheeler et al. (2018a) identified the following problems with this study: self-selection bias of the chosen 21 communities (all communities should have been chosen); lack of controls (e.g., no prices or other controls), and various other statistical issues.

KPMG (2018) used both cross-sectional and time-series data to quantify the economic impacts of different water recovery scenarios on 12 key sectors in 40 communities within the sMDB. To systematically assess the impact of water recovery scenarios on employment in the sMDB communities, KPMG developed a simulation model that relates for each of the 40 sMDB community's employment in 12 sectors to sector-specific key drivers that are directly impacted by water recovery scenarios, such as measures of irrigated farm production. The results indicate that environmental water recovery under the Basin Plan also has a negative impact on employment in the sMDB. Under the Basin Plan scenario, employment in all 40 communities was projected to contract, with the most affected communities being Robinvale (37%), Cobdogla-Barmera (23%), Waikerie (15%) and Swan Reach (13%). The results of the model indicate that seven of the communities studied are likely to experience relatively small effects (less than 2% reduction in total community employment) under the water recovery scenarios considered in the review, while nine communities experience modest to quite a significant decrease in employment (9% to 21% reduction in employment). The regressions again had multiple statistical issues (e.g., small sample size, use of simple linear interpolation for missing data without clarification), and when modelling fulltime employment, only had hectares of land, and dummies for areas, in the equation. Their results suggested that: a 1% increase in rice hectares leads to an increase in rice equivalent FTEs of 0.44%; a 1% increase in grape hectares leads to an increase in grape equivalent FTEs of 0.24%; a 1% increase in vegetable hectares leads to an increase in vegetable equivalent FTEs of 0.01%; and a 1% increase in fruit/nut hectares leads to an increase in fruit/nut equivalent FTEs of 0.15%.

RMCG (2019) reviewed and proposed policy recommendations regarding the aims of the Basin Plan (concerning water recovery targets), the principles agreed to with regards to the 2004 National Water Initiative, the water allocation policies, water sharing practices, water trading and water use. This study analysed water use within NSW and VIC southern MDB regions based on water use and availability data collected by RMCG over many years (particularly the MDBA's – Transition Period Water Take Report 2017/18). The analysis is based on simple statistics; thus, the overall quality ranking is low. This analysis showed an unexpected step change in water use in those regions. Specifically, between 2009 and 2019, annual average water uses in NSW Murray and Murrumbidgee regions reduced by 1,650GL to 46% of the long-term average. This reduction was caused by a combination of climate, water recovery, carryover policies and decreased water allocations in favour of the environment, high water security holders, and horticulture/dairy users. The study concludes that related policies caused several inequities and failed the NWI principles of "no third-party impacts".

5.2.1.2 Water market prices

The most modelled economic outcome from water recovery using econometrics has been water market prices. Aither (2016a) collected the annual median water allocation price from the ABS between 1998-2014, and measured the potential impact of Commonwealth water purchases on historical water allocation prices. The study found that Commonwealth water purchase increases the annual median water allocation price by using the water allocation price model, which is based on a regression analysis over 17 years of observed data (n=17). Specifically, the difference between modelled annual median prices with and without Commonwealth purchases is $24/ML, increasing from $88-$112/ML. Yet it highlights that Commonwealth water purchases are a less important driver of allocation prices (i.e., a quarter of the increase in temporary water prices) compared with total water availability and prevailing climatic conditions. Aither (2016b) analysed demand change data over the past ten years (2005 to 2015), and considered the demand change that might occur over the next five years (2015 to 2020). This analysis found that the water allocation price could be 13-36% higher in a moderate allocation season with Commonwealth environmental water purchases. It also projected that there are likely to be further significant changes in the sMDB allocation market between 2015-16 and 2020-21. Specifically, over the next five years, allocation prices are estimated to increase from $207 to $231/ML ($24/ML or 12%) in low allocation seasons, $118 to $131/ML($13/ML or 10%) in moderate allocation seasons, and $37 to $41/ML ($4/ML or 9%) in high allocation seasons.

However, Zuo et al. (2019) applied the VARX-BEKK-GARCH time-series regression to model the water market dynamics of monthly permanent and temporary water market trade from 1997 to 2017 in the Goulburn-Murray Irrigation District. This analysis found that water buyback had a small price and volume impact; and highlighted that estimates in some studies about the impacts of government water recovery on water markets (e.g., RMCG, 2016) were overestimated. Specifically, the authors found that the effect of government recovery is insignificant on temporary water prices or permanent market prices and volumes – and a 1% increase in water recovery resulted in a 0.14% reduction in temporary water volume traded. ABARES (2020a) used annual data from 2006 to 2019 to estimate the impact of water recovery on water allocation prices in the sMDB and found effects on price had increased over the period. Specifically, the estimated average effect of all water recovery across a mix of ‘dry’, ‘typical’ and ‘wet’ years on price was $72/ML.

5.2.2 Cross-sectional regression – water recovery studies

5.2.2.1 Irrigated water extractions and farm productivity

Wheeler et al. (2020a) used 2,481 on-farm MDB irrigation surveys across three years (2010-11; 2011-12; and 2015-16) and applied treatment effect estimator methodology to identify a ‘rebound effect’ on water extractions. The study highlighted the rebound effect through the analysis of a unique farm-level survey database, and revealed that irrigation infrastructure subsidies do not generate basin-scale water savings. Specifically, the study found that those who received an irrigation infrastructure grant increased their water extraction volumes (by 21-28%) and rates relative to other irrigators. Receiving an infrastructure grant increased the likelihood of irrigators expanding their enterprises (i.e., increased irrigated area, buying farmland and water entitlements), as well as changing their crop mix – compared to non-grant recipients. Grant recipients were less likely to adhere to contractive strategies, such as decreasing irrigated area and selling farmland, although these differences were not statistically significant. In the 2015-16 survey (n=1000) irrigators were asked about water recovery methods. On average, both grant and non-grant irrigators agreed that water buybacks should be suspended, and that more money needed to be spent on on-farm irrigation infrastructure. They disagreed that more money should be spent on buybacks (Table 5.2).

Table 5.2 Mean agreement score1 between irrigators who received grants and those that did not, with various attitudinal statements in 2015-16

Attitudinal Statements	*Grant (n=471)*	*Non-grant (n=529)*	*p-value*
Water buybacks for the Basin Plan should be suspended	3.89	3.95	0.45
More money should be spent on on-farm irrigation infrastructure by the Commonwealth	3.77	3.82	0.49
More money should be spent on water buybacks by the Commonwealth	2.05	1.95	0.19
Irrigation infrastructure money has been wasteful and inefficient	3.29	3.58	<0.01
I would rather irrigation infrastructure money was spent instead on rural health and education services	2.58	2.77	<0.01

Note: Mean agreement score was simply the average of the Likert scale (where 1=strongly disagree, 2=disagree; 3=neutral; 4=agree; 5=strongly agree) of recipients and non-recipients, respectively.
Source: Wheeler et al. (2020a)

However, 28% of irrigators overall did agree (or were neutral) that more money should be spent on buying water directly back from irrigators. 51% of all irrigators agreed (or 75% agreed or were neutral) that expenditure on irrigation infrastructure had been wasteful and inefficient; while 21% of all irrigators agreed (54% agreed or were neutral) that infrastructure subsidy money should have been spent on rural education and health services instead. Examining this by those who received a grant versus those who did not, irrigators who had not received any grants were more likely to agree the irrigation infrastructure program had been wasteful. Similarly, the groups had different views about whether irrigation infrastructure funds should have been spent on rural health and education services instead (with non-grant recipients more likely to agree) (Wheeler et al., 2020a).

Similarly, Hughes et al. (2020) used the Inverse Propensity Score Weighted regression model to measure the effects of the Australian Government’s on-farm infrastructure programs, particularly the *On-Farm Irrigation Efficiency Program* in the sMDB. The study uses annual survey data from 2009 to 2016 (n=1,889 observations across 833 unique farms) and found that these programs lead to higher farm water demand of an average 293ML per farm per year (an increase of 35%). However, they also found positive effects for participants in terms of, on average, an additional $495/ML and $1,233/ha of irrigation receipts or a 33.2% and 23.8% increase respectively – while also increasing farm profits by $135,200 per farm per year. Moreover, Ernst and Young (2018) also stated that on-farm infrastructure participants experienced positive socio-economic impacts (e.g., enhanced production, increased net financial benefit). The limitation of this study, however, is that data was collected inconsistently between different programs and therefore analysis of the historic cost of efficiency measures has been limited to specific programs where data was available.

5.2.2.2 Farm exit

Zuo et al. (2016) used both revealed data (data from irrigators who sold and offered water to the *Restoring the Balance* program) and stated preference data (data from irrigators who participated in a contingent behaviour exercise in 2011) to explore how irrigators in the sMDB buy and sell water entitlements in response to different prices. A high security water entitlement demand elasticity of -0.57 was estimated, along with a supply elasticity of 0.42. The relative inelastic demand supported the need for multiple tenders over time.

Zuo et al. (2015) examined the various factors that would influence exit package take-up across the different sMDB states. The results found that about one-fifth of farmers in the sMDB would require a price premium of around $1,600/ML over the current water entitlement market price (representing 174%, 81% and 89% over the water entitlement market price at the time in NSW, Victoria and SA, respectively) to take up an exit package. Price elasticity estimates of exit package take-up in all states were elastic at most price levels.

5.2.2.3 Influences on irrigator behaviour in water markets – selling water entitlements

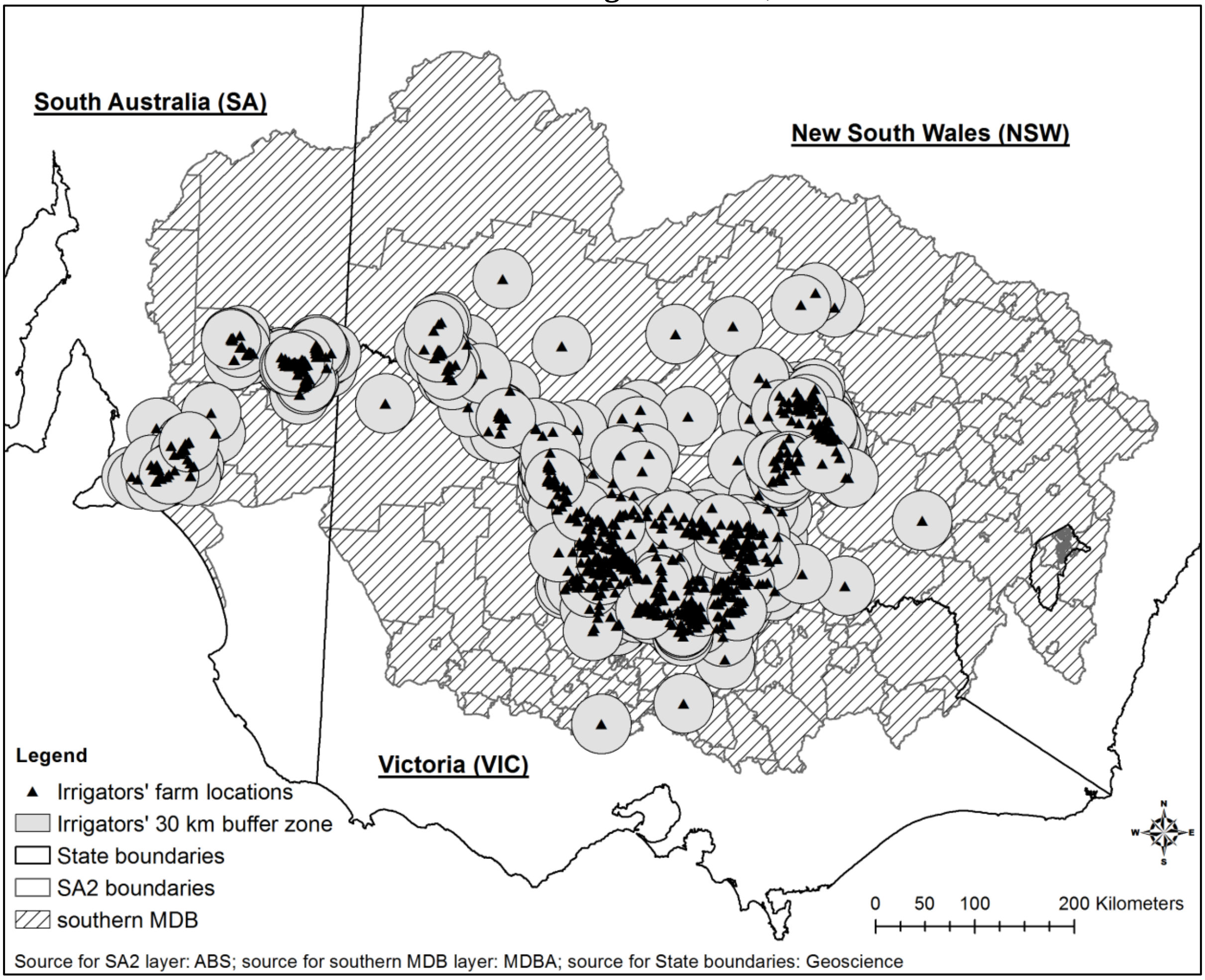
Wheeler et al. (2012) analysed sMDB irrigator intentions (n=1570) and actual decisions to sell water entitlements to the government using probit and ordered probit regression – and found that reasons for sale included debt, death, divorce and strategic reasons (e.g., following farm investment plans, water surpluses). Specifically, the following variables predicted permanent water sales most successfully: age, education, traditional attitude, number of children, information source, past water allocation sales, whole farm plan, water entitlement holdings, land use (% of annual and permanent crops), operating surplus, debt, allocation level, and the location (state).

Haensch et al. (2019) extended Wheeler et al. (2012) with additional data and spatial variable modelling n=1,462) in the sMDB, and found decisions to sell permanent water to the government were also influenced by neighbours' selling decisions (i.e., neighbourhood effect) and other locational factors (e.g., distance to city). The study also found that factors of rural community decline were not associated with higher permanent water sales. Figure 5.1 illustrates the spatial location of irrigators who had sold water to the government in 2012.

Wheeler et al. (2013b) explored alternative water recovery options, such as allocation trade and water leases by the CEWH. Not only was it revealed that irrigator willingness to participate in selling temporary water to the CEWH was much higher, but also that the CEWH was found to be able to recover more environmental water/deliver more environmental outcomes by employing a mix of entitlement and allocation purchases.

As discussed in Wheeler (2022), there are two broad potential impacts that can arise from selling water entitlements: a positive impact (reduction in debt, farm restructure and reinvestment to make it more productive or efficient) and a negative impact (less water for production and/or higher costs in buying water allocations or feed). Wheeler et al. (2014c) found no significant impact on current year profitability from selling water entitlements (although a negative effect from buying water entitlements was found); while Wheeler et al. (2014a) found only weak to no significant evidence of a delayed impact from selling water entitlements within farms that remained farming. Path dependency in farming strategy does appear to exist, in that once a farmer implements a strategy, they are more likely to continue doing so over the next five years – which has been found to be the case for selling permanent water (e.g., Seidl et al., 2021; Wheeler et al., 2013a). This finding indicates that, at some point, selling more permanent water entitlements may be a negative financial strategy for an individual farm if the desire is to continue irrigation, and more research is needed in this area using longitudinal panel datasets.

Figure 5.1 Irrigator locations and spatial units in the sMDB of those that sold water entitlements to Federal government, as at 2012



Source: Haensch et al. (2019, p. 737)

Wheeler et al. (2021) investigated the links between ground and surface-water resources in the Goulburn-Murray Irrigation District and identified the following: 1) groundwater bores located closer to surface-water sources were associated with more extraction; 2) higher surface-water allocations, an indicator of surface-water availability, were negatively associated with groundwater extraction; 3) an increase in the price of surface-water allocations was associated with an increase in groundwater extraction; and 4) an increase in trading volumes for both water allocations and entitlements in the surface-water market was associated with an increase in groundwater extraction.

Zuo et al. (2022) used principal component analysis to identify five clusters of sMDB irrigators from a set of twenty possible farm and water strategies during 2015-16 (n=977); and then used multinomial logit regression to identify influences associated with each cluster. The five clusters of irrigators include those: expanding the farm (includes buying temporary water); expanding and diversifying (includes buying permanent water); downsizing (selling both temporary and permanent water); transitioning (switching away from irrigation to dryland); and saving (using carryover – water saved in storages for the following year). Around a third of irrigators can be classified as trying to expand the farm (the Expanders and the Expanders and Diversifiers); another third, of typically older irrigators, have lower levels of debt and spare water (the Savers); while a final third, who also tend to be older but also face financial and/or psychological stress, are placing more emphasis on dryland production (away from irrigation) or decreasing agricultural production altogether (the Transitioners and Downsizers). Higher long-term temperature of a farm area's location increased the probability of the irrigator being a downsizer – which may indicate a future trend of irrigation farm exit, given the predicted rise in temperature from climate change.

5.2.2.4 Influences on irrigator preferences for water recovery expenditure

Using a mail-out survey across the southern MDB in 2011, Loch et al. (2014a) and Loch et al. (2016) analysed irrigator preferences to allocate federal water recovery budget funds in the sMDB (n=535). These market-based water policy programs included water entitlement purchasing, temporary water market products and exit-based packages to recover water, and were compared against irrigation infrastructure on and off-farm programs. **Error! Reference source not found.**shows irrigator preferences for government expenditure on market-based programs to reallocate water toward environmental uses by state. Overall, irrigator preferences were 56% for infrastructure and 44% for all types of water market purchases.

Table 5.3 Water recovery irrigator budget expenditure preferences in the sMDB in 2011/12

Policy Options	**Mean %**[c]				**One-way ANOVA F-test**
	NSW (n=176)	**SA (n=205)**	**VIC (n=154)**	**Weighted Average**	
Upgrading on-farm irrigation infrastructure	<u>32</u>[b]	21	<u>34</u>	31	17.44***
Upgrading off-farm irrigation infrastructure	<u>28</u>	<u>23</u>	<u>25</u>	26	2.09[a]
Water entitlement purchases	<u>18</u>	34	<u>19</u>	21	21.71***
Water Allocations/Entitlement leases/option contracts	<u>12</u>	6	<u>11</u>	10	3.95**
Exit Packages & revegetation payments	<u>6</u>	11	<u>7</u>	7	4.69**
Standard Exit Packages	<u>5</u>	<u>5</u>	<u>5</u>	5	0.42[a]

<u>Notes</u>: a Represents the robust test of equality of means (Welch) due to heterogeneous variances, and *p-value<.1; **p-value<.05; ***p-value<.01.
b Underlined state mean % indicate they are not significantly different at p<0.05 using Bonferroni post-hoc comparisons.
c Calculations do not include 'no answer' responses.
<u>Source</u>: Loch et al. (2014a, p. 400)

SA irrigators were significantly different in their preferences to NSW/Victorian irrigators regarding most water recovery options. Specifically, SA irrigators favoured higher spending on water entitlements and exit packages compared with NSW or Victorian irrigators, and less spending towards on-farm infrastructure. Irrigators' main reason for their budget preferences was the need to improve irrigation efficiency (49%). Irrigators preferring trade and exit packages (15%) were driven by various policy options, or the belief that policy needed to be more flexible. Irrigators favouring water markets were more likely to be interested in retirement options and to indicate environmental water needs as a reason (as a more cost-effective option).

Loch et al. (2016) used zero-one inflated beta regression analysis to investigate irrigator engagement with market-based programs. The significant influences found were state regional influences, the type of farm production and recent stress that the farmer has incurred (i.e., debt, low income, or low water allocations). In particular, NSW irrigators (primarily annual cotton and rice farmers) prefer farming over water trade, whereas perennial viticultural and horticultural farmers show positive engagement with market-based programs.

Also, SA irrigators were linked with moderate preferences for market-based options. Broadacre and dairy cropping farmers were associated with moderate lower proportional preference outcomes (dairy farmers are more flexible with risk management and thus less reliant on market mechanisms). Furthermore, increased farm debt and water extraction levels as well as higher holdings of high security water entitlements showed decreased preferences for market-based programs. Farm income variables (both on- and off-farm) showed positive associations with proportional preferences. Price variables appeared to be less relevant but higher water entitlement prices paid in 2010/11 prompted irrigators toward positive proportional water market preference outcomes.

5.2.2.5 Future MDB land patterns

Aither (2020) used a scenario approach to estimate how consumptive water supply in the sMDB in any given future year will be required by permanent irrigated horticulture and the ‘headroom’ above that (namely the amount available to other industries). However, the model assumes that there is no reduction in permanent horticultural plantings that may occur due to water availability. Their conclusions were that existing permanent horticulture in the connected Murray region is expanding and will grow from their estimated 1,230GL per annum to 1,400GL at full maturity.

5.2.3 Other applied descriptive studies on the influences of water recovery programs

5.2.3.1 Community economy level impacts

Arche Consulting (2012) developed case study assessments to explore the net impacts arising from the implementation of SDLs and water recovery programs at a local community scale. The study highlighted the negative effects of an SDL of 2,750GL, with water recovery achieved through the buyback and infrastructure investment. The negative effects included decreased output, reduced income, and reduced employment. The results also highlighted the longer-term benefits of investing in infrastructure, which resulted in water savings being retained on farm, and contributed to direct employment in agriculture.

MJA et al. (2010) assessed the socio-economic impacts of Basin Plan and related changes in water availability. The analysis involved face-to-face interviews with 250 stakeholders across the 12 regions and other available surveys and datasets. The analysis was based on simple statistics; thus, the overall quality ranking is defined as low. The report found that the reduced water availability scenarios will reduce agricultural production and cause financial losses to farmers and communities with varying impacts depending on the sector. For example, substantial socio-economic flow-on impacts are expected for the cotton sector (especially for reduced water availability greater than 40%), especially for remote cotton-dependent areas. Also rice production would decline at a substantial pace (i.e., a 40% reduction in water availability would lead to a 60% reduction in rice production).

5.2.3.2 Influences on farmers' water entitlement sales to government

Qualitative studies based on farmer interviews found similar influences on water entitlement selling, as reported under Section 5.2.2.3. For example, Thampapillai (2009) identified that farmers in financial hardship, close to retiring, with off-farm income availability, and having no successor were more likely to sell permanent water to the government. In general, irrigators unwilling to sell expressed concerns about the rural viability, rising costs of the irrigation infrastructure system, government management of environmental water and transparency of the government's buy-back program. Kuehne et al. (2010) emphasised the relevance of non-profit maximising values for the decision to sell water to the government, such as plans for staying in farming, years left to retirement, succession arrangements, being full-/part-time or hobby farmer, future employability, whether the water sale included the land, conditions of the farm exit grant package, and the price on offer. A more pessimistic attitude towards the future resulted in a higher probability of water sales. While confirming debt as a dominant reason for selling permanent water, Bjornlund et al. (2011) likewise emphasised the role of irrigators' values, attitudes and wellbeing (financial security is only one driver of wellbeing).

Wheeler and Cheesman (2013) was the largest ever survey of sellers to the *Restoring the Balance* program to date – over 500 sellers – and their key results included: 70% of the survey participants remained in farming, after they had sold parts (60%) or all (10%) of their permanent water, and 30% exited farming after they had sold all of their permanent water. Thus, exiting farming was not a major driver for the decision to sell permanent water to the government. Dominant reasons for selling were debt (30%) and cash flow (30%). Cash flow was mainly used to support farm income and increase viability (22%) and also to fund on-farm investment (8%). Other reasons for water sales were farm exit (15%), having surplus water (9%), age, and death/divorce. Also, few participants responded with environmental reasons, family support, frustration with local IIO or the government, channel upgrades, unbundling of land and water as well as decreased water quality levels.

5.2.3.3 Farmers' views of impact from water infrastructure programs

A series of reports (Schirmer 2015, 2016, 2017b, 2019) and DAWR (2018) examined the socio-economic effects of the Commonwealth's water recovery investments, as part of the University of Canberra's annual Regional Wellbeing Survey (RWS). The reports summarised in the following are based on irrigators' views and opinions about the investments as well as their likely impact on their farm. Results are based on simple descriptive analysis of irrigators' farm outcomes and their views on government programs.

Schirmer (2015) reported on the results of the RWS for the years 2013 and 2014, focusing on the socio-economic effects of investments in water infrastructure. In 2013, 900 irrigators were included, which grew to 1,000 irrigators in 2014. Irrigators located in areas where off-farm infrastructure improvements were undertaken mainly (63.8%) reported positive effects for their farms, while 14.6% noticed negative effects. Specifically, it seemed that off-farm upgrades were associated with an improvement in on-farm financial performance. The majority of irrigators receiving on-farm water infrastructure grants found them to be either very useful (80%) or moderately useful (20%) mainly regarding better farm financial performance. Survey results also indicated that on-farm infrastructure grants are associated with some improvements in irrigators' views about their community.

Schirmer (2016) reported the results of the RWS regarding the socio-economic effects of all three parts of the Commonwealth's water program (on- and off-farm infrastructure and water entitlement sales) for the year 2015. In this year, the survey collected data from 13,303 people living in rural and regional Australia during September-November, including 833 MDB irrigators. Irrigators who received investments for on-farm infrastructure reported largely positive socio-economic effects. It was also suggested that flow-on effects for communities were likely to be mostly positive. This may not be the case when external pressures hinder the adequate use of such grants. Opinions regarding the impact of off-farm infrastructure modernisation was more (spatially) varied with irrigators mainly reporting better timing of water delivery, but also increased costs of water delivery. Effects of water entitlement sales to the government were also perceived differently (half reporting positive and half reporting neutral or negative effects). Irrigators were perceived to be more positive when exiting the industry as compared to irrigators who remained in irrigated agriculture.

Schirmer (2017b) focused on examining the effects of the infrastructure grants using the RWS for the time period 2014-2016, particularly for 2016. In these years the RWS included 869 (2014), 833 (2015) and 631 (2016) MDB irrigators. The three-year survey results showed again overall a positive response to on-farm infrastructure grants by irrigators, even though there is great diversity noted amongst irrigators, along with the higher debt levels and increased power costs some irrigators were experiencing as a result of the program. Generally, irrigators found that increased farm productivity and water use efficiency outweigh negative effects. Irrigators receiving grants were more likely to be expanding their farms, except farms that were affected by market downturn. With the help of on-farm grants irrigators were able to realise works that are larger in scope and scale. The survey further found that irrigators who invest in on-farm modernisation were more likely to also invest in other actions to improve water use efficiency. On the other hand, the effects of off-farm upgrades were difficult to assess as they involved differing types of modernisation, undertaken at different points in time. Furthermore, irrigators were less aware of off-farm upgrades, and were less likely to report positive effects for their farms. It was generally reported that off-farm works impacted positively on water delivery timing, efficiency of water use and farm productivity, but also many irrigators found them to have a negative effect on the costs of water delivery. The survey showed higher stress levels of irrigators located within off-farm infrastructure modernisation regions, potentially relating to higher costs and reducing the ability to gain from the positive effects of off-farm upgrades.

In DAWR (2018), MDB irrigators' views and experiences of the government's water recovery programs between 2013 and 2016 were studied. Between 600 and 850 MDB irrigators were included in the RWS during this time. DAWR (2018) reported on the results mainly from the survey year 2016 and found that upgrades to on-farm irrigation infrastructure under the government's program resulted in positive effects for farms in more than 80% of the cases (namely water use efficiency, timing of water delivery, on-farm workload and farm productivity and profitability). On the other hand, for some irrigators upgrades resulted in negative effects, including energy costs and farm debt. The government's investments in off-farm irrigation infrastructure resulted in positive effects for farms, reported in around 50% of the farms located in relevant regions (with regard to improving the timing of water delivery and water use efficiencies). However, many irrigators reported they resulted in increased farm costs, especially with regard to water delivery. Irrigators also reported varied effects of water entitlements sales to the government: irrigators remaining in irrigated agriculture after the sale noticed positive effects for irrigation efficiency, but negative effects from water allocation prices and water delivery costs (DAWR, 2018).

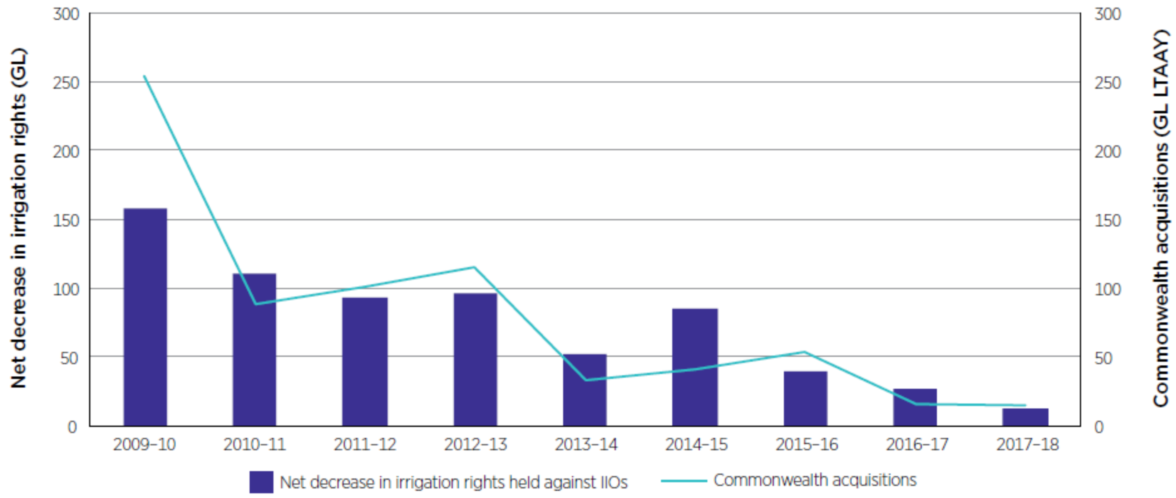
Schirmer (2019) reported from the 2018 RWS results focussing on the social effects of investments in water infrastructure. In 2018, 657 irrigators participated in the RWS, including 412 MDB irrigators. The report concluded that investments in on-farm infrastructure provided for many modernisation works to be undertaken earlier than they would have otherwise (52% said they would have undertaken the work anyway, but in a longer timeframe). Irrigators modernising on-farm infrastructure tended to be farm expanders, while 80% did not expand overall volume of water use and 20% did. Overall, on-farm modernisation was associated with positive effects regarding farm productivity and production, farm outcomes and farmer wellbeing. On the other hand, off-farm modernisation was not associated with many direct impacts for individual irrigators.

5.2.3.4 Other consequences of water infrastructure programs (e.g., stranded assets, Swiss-cheese and future irrigation delivery and operating impacts)

Finally, another cost that is often raised in relation to water recovery is the issue of stranded assets and/or redundant infrastructure. A stranded asset is any component of the water delivery system (e.g., meter, off-take wheel, channel diversion box, etc.) that reduces in value on the market as compared to its value on a balance sheet because it has become obsolete (or unused) before being fully depreciated by an IIO. In irrigation areas, when there is a permanent decrease in the demand for water delivery services the assets of IIO can become unused or underused (or stranded). This is also known as the ‘Swiss-cheese’ effect from infrastructure removal and the spreading of operational costs across a reduced irrigator membership (Walsh, 2012). Empirical evidence from Wheeler and Cheesman (2013) found that of the farmers who owned water in irrigation areas, 60% of them kept their delivery rights, while 94% of those who stayed farming after selling water kept their delivery rights.

IIO areas now impose termination or exit fees to cover the ongoing costs associated with stranded assets. These are a charge imposed on entitlement trade and subsequent loss of a water access entitlement out of an irrigation district or area. These fees are set by the ACCC and charged to maintain the delivery infrastructure or any stranded assets that remain after the water access entitlement has left the area. These results indicate uncertainty about the reality of stranded assets, but at the same time there needs to be a recognition that a severe rationalisation of irrigation areas needs to be considered anyway, with perhaps large amounts of an area removed from the system. Following on from above, work has been conducted by the ACCC to understand the net decrease in irrigation rights over time, following government acquisitions. Figure 5.2 Australian Government environmental water acquisitions and net decrease in irrigation rights, 2009–10 to 2017–18illustrates the net decrease in irrigation rights held against irrigation infrastructure operators (where terminations were highest in 2009-10), and that terminations have fallen over time, and overall been significantly less than Commonwealth acquisitions.

Figure 5.2 Australian Government environmental water acquisitions and net decrease in irrigation rights, 2009–10 to 2017–18



Source: ACCC (2019; 53)

The largest price increases occurred in modernised IIO schemes (e.g., pressurised systems in particular), where infrastructure modernisation impacts upon irrigators' future delivery charges and energy costs (ACCC, 2019).

MJA (2019) discuss how off-farm irrigation infrastructure is 'gifted' (i.e., the infrastructure is excluded from the regulatory asset base for the duration of its life) in return for water savings. However, operational, maintenance costs and tax on the asset still have to be paid, and therefore impacts on irrigators' delivery and other charges. The exclusion of infrastructure from the regulatory asset base means irrigators are not charged for infrastructure depreciation or financing costs, which will have implications when it needs renewal in the future, causing a range of concerns. Irrigation delivery charges are expected to rise considerably in the coming decade, hence the real ongoing cost of irrigation upgrades may be hidden from irrigators.

The Productivity Commission (2018) investigated four aspects of the costs of water recovery: 1) the premium associated with recovering water through infrastructure; 2) the potential costs of recovering 605GL if supply measures do not deliver; 3) the potential costs of recovering 450GL through efficiency measures for different entitlement recovery portfolios; and 4) the costs of recovering the 450GL of environmental water before it can be delivered effectively[14]. Drawing on data provided by the Department of Agriculture and Water Resources, the Productivity commission estimated the premium of recovering water through infrastructure programs at $2,800/ML of LTAAY. The cost of recovering 605GL depends on the method of recovery chosen, with the costs of supply measures at $678 million, water purchases at $709 million, and infrastructure modernisation at $1.2 billion. The cost of recovering 450GL through efficiency measures with a market multiple of 1.75 is estimated between $2.061 - $2.271 billion depending on the structure of the entitlement portfolio. This is $486 - $696 million higher than the WESA budget of $1.575 billion. Finally, delaying the recovery of 450GL through efficiency measures until 2030 instead of 2024, so that the necessary

[14] Effective delivery of the 450GL depends on the easement and removal of constraints, which is unlikely to be fully operational by 2024 (Productivity Commission, 2018).

easements and constraint removals are in place saves the Australian government $203 million, assuming a 5% discount rate and water recovery costs of $4,966/ML (in LTAAY).

5.3 Other key MDB water scarcity studies that study economic outcomes

This section summarises some of the other key studies in our literature review that looked at the relationship between various forms of water scarcity and economic outcomes. We break this down into: water market time series modelling; water market participation; land use change; farmer exit; and farmer mental health studies.

5.3.1 Water market time series modelling studies

There have been many water market studies in the MDB that have sought to examine impacts of prices and volumes. Although the previous sections have detailed the particular studies that attempted to directly estimate the impact of water recovery on the market, many other studies have just focused on general water scarcity variables. Some findings from this broader literature are detailed here (but were not officially in our systematic literature review).

Nguyen-ky et al. (2018) forecasted monthly NSW Murray Irrigation Area temporary water prices from 1999–2015, using Artificial Neural Network and hybrid Artificial Neural Network-based Bayesian modelling approaches. Similar to other studies, the results indicated that current water allocation prices, general water security volumetric allocations and commodity price data of cereal and meat prices were significant determinants of future water temporary prices. Errors in estimation were greater in periods of high uncertainty (e.g., Millennium drought). Plummer and Schreider (2015) developed a climate driven regression model to estimate the effects of volume of water in storage and winter rainfall on water allocation price jumps in Northern Victoria between irrigation seasons. They found that both seasonal rainfall and volume in storage significantly influenced the price jumps between irrigation seasons. Other studies have examined whether water markets exhibit characteristics similar to other financial markets (e.g., market depth in Brooks et al., 2009; price clustering features in Brooks et al., 2013 and Zuo et al., 2014; and price leadership in Brooks & Harris, 2014). They all also confirm the link between water scarcity and prices.

De Bonviller et al. (2020) estimated groundwater temporary water price elasticity, using ten years of monthly surface and groundwater temporary market data (2008-2018) in the Murrumbidgee. Using two-stage least square regression, they found a close to unit price elasticity (-1.05) – namely increases in groundwater temporary prices led to almost very similar decreases in groundwater temporary market demand. The study also found a significant price leadership phenomenon from surface-water allocation markets to groundwater allocation markets.

De Bonviller et al. (2019) studied daily water allocation price and volume data (2008–2017) (n=28,983) to identify abnormal price movements preceding water allocation announcements in the Greater Goulburn trading zone in the southern MDB, to investigate the existence of insider trading. Given that under the Basin plan, water market rules were introduced in 2014 in the MDB that officially regulated insider trading, it allowed a natural experiment test within the data. The study used a moving average time-series regression models and found that scarcity and seasonal factors were the most important influences of water allocation market price movements. Specifically, daily water allocation trade amount and total storage in major dams were negatively statistically significantly associated with water allocation prices. Furthermore, a commodity price index received by irrigators showed a statistically significant positive impact on water allocation prices after 2014, as higher commodity prices tended to increase irrigation water demand. In particular, there was evidence of abnormal price movements (in the hypothesised direction) preceding water allocation announcements, suggesting the presence of insider trading, especially before 2014. There is also some evidence that the new water trading rules introduced in 2014 may have decreased (or eliminated) the incidence of such abnormal price movements, although there is still some very weak evidence of abnormal price movements post-2014. Overall, the study suggested that water allocation market traders are becoming more sophisticated and speculative.

Haensch et al. (2016), in a random-effects panel model time series (n=126 from 2000-2011) in the sMDB, found that larger volumes of permanent water were likely to be sold from regions with higher dryland salinity in soils and lower groundwater salinity issues. This suggested that groundwater entitlements may act as substitutes for surface-water entitlements in recent years (where they are viable substitutes) (Haensch et al., 2016). Furthermore, Haensch et al. (2021), modelling 2010-11 to 2013-14 from a leading private water broker at this time (Waterfind) in the sMDB using a random effects tobit panel model, found little evidence that rural community decline measures (i.e., disadvantaged communities) are associated with higher permanent water sales – in contrast to some views held by communities. Overall, key spatial influences such as net rainfall, groundwater use and dryland salinity, were determining influences on the volumes of water entitlement sold, while water entitlement purchase volumes were much more likely to be associated with water market prices, location and soil productivity. On the other hand, no statistically significant relationship was found between very remote areas and areas with lower socio-economic classifications with higher volumes of water entitlements sold. However, there did seem to be a link between more disadvantaged areas and higher volumes of water entitlements purchased. Water allocation trading was more associated with water scarcity factors.

5.3.2 Water market participation studies

Zuo et al. (2015) used regression analysis of irrigator surveys from 2006-07 to 2009-2010 (n=1,232) in the sMDB and found that farmers experiencing higher variability in profit and facing more downside risk purchased greater volumes of temporary water. Using an updated dataset from 2006-07 to 2011-12, Nauges et al. (2016) modelled by industry (horticultural (n=963) and broadacre farms (n=543)) and found that horticultural irrigators used temporary water trading because they are averse to the risk of large losses (downside risk) while broadacre irrigators use water trading as they are averse to the variability (variance) of profit. This confirms that water trading was used by irrigators as a risk-management strategy.

5.3.3 Land use change studies

Connor et al. (2009) employed a Danzig two-step programming framework with recourse, with the first stage as farmers' choice of long-run capital investments, and the second stage as choices regarding water application rate and fallowed area – to model the impact of different climate scenarios (mild, moderate, severe) on agricultural production and profitability for two subregions (South Australia and Victoria) in the lower MDB. The overall model maximises profits for each subregion subject to land and water constraints. The model predicts that a 30% reduction in water allocation (moderate climate scenario) is associated with only -5% revenues and -9% profits in Victoria. Impacts of a reduction of 70% or more in water allocation (severe climate scenario) lead to -42% revenues and -52% profits in Victoria;

revenue and profit impacts for South Australia are higher than for Victoria in all climate scenarios (i.e., reductions are larger), but follow the same overall pattern in regards to different climate scenarios.

Connor et al. (2012) built on the study by Connor et al. (2009) to investigate the combined impacts on irrigated agricultural production of reduced, more variable and more saline water supply in three subregions of the lower MDB. The Danzig two-step programming framework is extended by including volatility of water supply and salinity, and by including additional variable production inputs and choices of long-run capital investment in the different modelling steps. Modelled cropping area and farm profit contract with the more severe the climate change scenario. This result is consistent across the three model specifications of water scarcity only (-46% area, -69% profit for severe climate); water scarcity and increased volatility (-59% area, -76% profit for severe climate); and water scarcity, increased volatility, and changing salinity (-68% area, -87% profit for severe climate). Considering water scarcity, variability and salinity simultaneously, leads to greater contractions in cropping area and profit for all climate change scenarios, as compared to the other model specifications. However, the impact on water use per hectare is less uniform across models, with the salinity model showing an increase in water use with more severe climate scenarios, whereas the other models predict a decrease in water use in these cases.

Rowan et al. (2011) mirrors the findings by Connor et al. (2009; 2012) regarding the impacts of climate change on farm profits. Rowan et al. (2011) developed an integrated stochastic dynamic modelling framework designed to assess irrigation farm viability under climate change scenarios. The framework is applied to a theoretical Sunraysia perennial irrigation farm for four future climate scenarios (baseline, mild, moderate, and severe). The model maximises irrigation farm profit, while accounting for short-run water use decisions, and long-run investment decisions in farm capital by means of two objective functions. Results show that farm profitability decreases with increasing climate change severity, with farm profitability under moderate climate change reducing by 31% and 64% for the steady state and stochastic model respectively. Averages of the marginal value of irrigation water (MVIW) for each scenario were calculated as \$5/ML under baseline conditions, \$22/ML under mild climate change, and \$147/ML under moderate climate change. Average values of MVIW for each climate change scenario were calculated as \$5/ML under baseline conditions, \$22/ML under mild climate change, and \$147/ML under moderate climate change.

Qureshi et al. (2013a) considered the relationship between water availability in different climate change scenarios and agricultural production, using a positive mathematical programming model to investigate the impacts of reduced rainfall, water allocations, and increased crop water use on agricultural production, within twelve MDB catchments. Reviewing theoretical and technical model details, special attention is given to address calibration and model parameterisation issues. Farm profits are maximised in each catchment under four climate change scenarios (baseline, dry, medium, wet), considering annual establishment costs, variable costs, and land and water availability constraints. Results show that the expected mean gross value of irrigation decreases with increasingly dry climate scenarios, with reductions of 14% for the dry climate scenario compared to the baseline case when irrigators can engage in water trade – as opposed to 26% reduction if neither water trade nor crop-management adaptations are possible.

Agbola and Evans (2012) explored a different angle of water access and supply: they were interested in the impact of water allocation prices on cotton and rice planting areas. They used a Nerlovian partial adjustment model with data from the Murrumbidgee area from 1965-2008 to estimate the response of cotton and rice area to water allocation prices in the MDB.

The area of cotton or rice planted is the dependent variable of a Fully Modified Ordinary Least Squares regression model, controlling for lagged cotton/rice area/yield/price, and lagged prices for wheat, barley and water allocation. Cotton and rice acreage were found inelastic in the short and long run to changes in water allocation prices, yet changes in water allocation price have a statistically significant effect on rice acreage (at the 1% level) and cotton (at the 5% level). Rice acreage is statistically significantly impacted at the 1% level by all independent variables, whereas cotton acreage is statistically significantly impacted at the 1% level by lagged cotton area, cotton price and barley price, and by the water allocation price at the 5% level. A one percent increase in water prices lead to a 0.06% decrease in rice acreage in the short term, and a 0.52% decrease in the long term (hence was highly inelastic). The biggest influences on changes in rice hectares were the price of wheat, followed by the price of rice, the price of barley, the price of cotton, the rice yield, and finally water prices. The greatest influences on changes in cotton hectares were the price of barley, followed by the price of cotton, the price of rice, cotton yield, wheat price, and finally – again – water prices. A 1% increase in water prices lead to 0.07% decrease in cotton in the short term, and 0.60% decrease in long term (hence was highly inelastic).

Connor et al. (2014) used logistical regression to model the influences on irrigation area and irrigation revenues in seventeen MDB regions from 1997-2010, across nine industries (n varied from 24-41). Explanatory variables included allocation %, area irrigated, commodity price evapotranspiration less rainfall. Their results estimated a far smaller marginal revenue decline per unit decline in water allocation than other macro models such as CGE and partial equilibrium, but a more consistent impact on water irrigated area. Their elasticities for irrigation area varied from 0.09-0.42 – suggesting that for a 1% decrease in water allocations, irrigated area decreased by 0.09-0.42% across various industries. For irrigation revenue, for a 1% decrease in water allocations, all commodities revenue only decreased by 0.1%.

Loch et al. (2020) and Adamson and Loch (2021) also modelled scenarios of water use in the MDB. In particular the paper studied the current modelling of uncertainty with respect to investment choices (e.g., technology adoption to improve water use efficiency). They used a combination of cost-benefit analysis and state-contingent analysis, to model uncertainty as alternative states of nature. Water inputs were modelled under two categories: water that is required (fixed) to keep capital (e.g., tree-crops) alive, and water that allows for productive crop yields (where annual crops do not require fixed water, as all inputs are used to create productive yields). They found that systems with greater rates of fixed water input requirements are at far greater risk of exceeding tipping points. Adamson and Loch (2021) focused on an example of the almond industry in California and identified: i) water use efficiency is typically not economically attractive to private investors due to relatively low savings; ii) subsidies are needed to incentivise uptake; but iii) risk remains high and both public and private exposure increases as a result of the co-investment choices.

Doolan et al. (2019) assessed MDBA's development of a modelling tool under the *Capacity and Delivery Shortfall Project* (representing consumptive and environmental water demand, and River Murray capacity to deliver water to satisfy this demand). This tool is a long-term daily simulation model of the River Murray, based on 125 years of hydrological data and the National Hydrological Modelling Platform, also known as the Source Murray Model (SMM). The SMM was configured to closely represent 2018-19 water demand and trading conditions (including water policy and IVTs) as the reference scenario, and is intended to allow for the simulation of future water demand and River Murray flow capacity under different scenarios, identifying water supply shortfall, associated drivers, and their development over time. Doolan et al. (2019) concluded that risk of delivery shortfall to regions downstream of the Barmah Choke will increase over time, given: 1) increased areas of horticultural plantings in the Murray Valley; 2) increasing water demand of maturing existing horticultural plantings; 3) environmental water delivery requirements under the Plan; and 4) a drying future climate.

HARC (2020) was commissioned to examine water use patterns and areas planted for different crops in the region from the Barmah Choke to SA from 1993-2018. The report used two data sets: 1) SunRISE crop area data (http://www.sunrisemapping.org.au/) for NSW, SA, and VIC; and 2) water use data from MDBA account sheets. They found an increase in planted areas in the Lower Murray, driven by an increase in permanent plantings, with areas of seasonal crops almost identical in 2003 and 2018. Although grape vines remain the dominant horticultural crop by area, with just over 50,000 ha in 2018/19, their planted area has steadily fallen over time, coinciding with large increases in areas planted to nut trees, particularly in Sunraysia (VIC) where they are now the dominant horticultural crop.

5.3.4 Farmer exit

Wheeler et al. (2020b) employed panel data over a 20-year period from 1991 to 2011, and applied spatial regression modelling at the regional level to assess the impact of economic, and water factors on net farmer number changes in the MDB. The panel data consisted of four periods, namely, 1991–1996, 1997–2001, 2002–2006 and 2007–2011, with 996 total observations. Contrary to expectations, they found changes in irrigation water diversions had no significant effect on MDB farmer exit.

Wheeler and Zuo (2017) examined how drought and water scarcity impacted irrigator exit intentions and found that there was only weak evidence to suggest that irrigators' exit intentions were higher in times of drought – however there was strong evidence to support the influence of a lagged water scarcity impact on farm exit intentions during periods of non-drought (i.e., in other words, irrigators were more likely to intend to exit at times when the property market was less depressed). There was also strong evidence that poorer performing farms (measured by rates of return and higher debt over a certain level) were more likely to have exit intentions during drought periods, but not necessarily so in non-drought periods. Older age was the most consistent predictor of farm exit intentions across all industries, especially during drought periods.

5.3.5 Farmer mental health studies

Climate change decreased water inflows and intense competition for water supply will exacerbate the stresses inherit in farming and impact wellbeing and mental health (Wheeler et al., 2018b). The pathway that this leads to includes: 1) worsening farmer mental health; 2) declined agricultural production and livelihoods; 3) changed environmental conditions; 4) reduced employment and depressed rural community; 5) migration and separation of family; and (6) physical health harm.

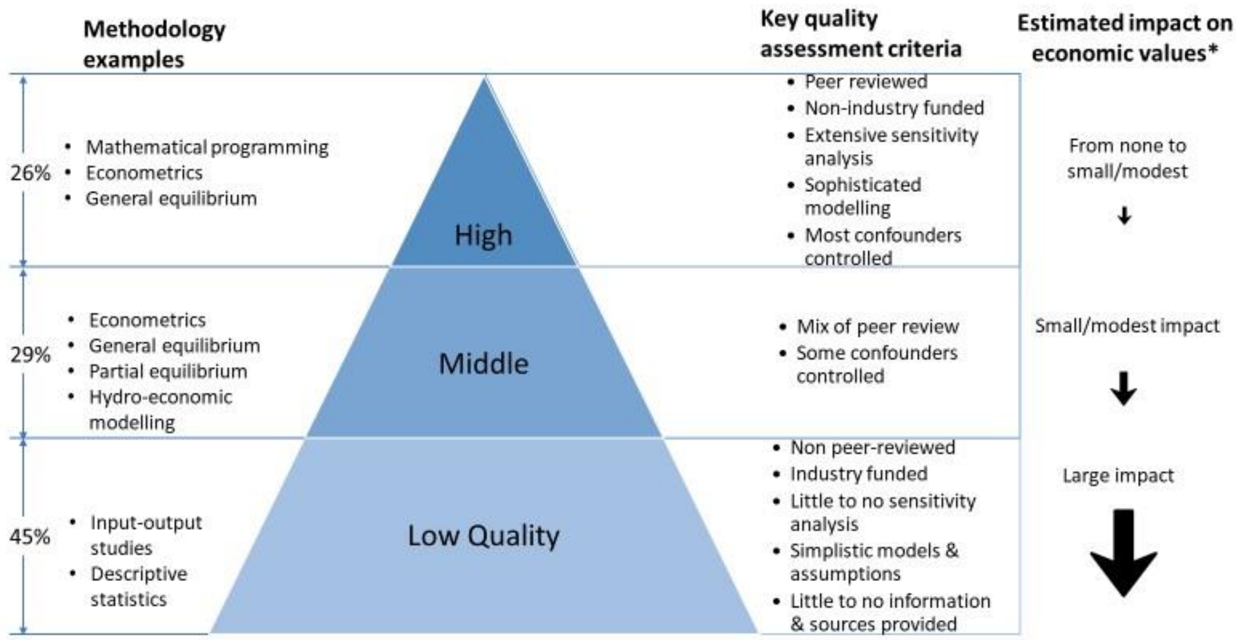
Wheeler et al. (2018b) analysed the mental health of 1,000 MDB irrigators in 2015/16 and, using descriptive statistics only with some testing, found that higher psychological distress was most related to: finances; drought; water availability; commodity prices; and time. Horticulture irrigators had higher distress, followed by broadacre, dairy and then livestock irrigators. Both Wheeler et al. (2018b) and Zuo et al. (2022) showed that there was a strong association between farmers citing financial stress and worse psychological distress, and consequently then planning on leaving the farm. Wheeler et al. (2018b) also established a strong positive association between psychological distress, and farmers acting very negatively towards issues such as water trading; government; the environment; Basin Plan; and optimism about the future. In particular, farmers were more likely to disagree that water trading had been a good thing for farming, if they were suffering moderate to very high levels of psychological distress.

Using ordered probit modelling, Daghagh Yazd et al. (2019) found that the main drivers of MDB irrigator psychological distress in 2015-16 were worsening financial capital (namely lower farmland value, higher farm debt, lower % of off-farm income, lower productivity change over the past 5 years and lower net farm income). More recently, Daghagh Yazd et al. (2020) used a different dataset over 14 years and examined whether climatic conditions and water scarcity were associated with worsening farmer mental health in the MDB. The sample used 2,141 observations (for 235 farmers) from a national longitudinal survey between 2001-02 to 2014-15 and was modelled using correlative random effects panel data regression. Key findings were that farmers' located in areas with reduced rainfall, water allocations less than 30% and mean daily summer temperature over 32°C had significantly worse mental health than farmers elsewhere. In addition, farmers who had lower income during drought were much more likely to have worse mental health than in non-drought times.

5.4 Summary

This chapter has focussed on the studies that have sought to investigate the impact of various water recovery programs on economic outcomes. It also provided a quality assessment of such studies, looking at both internal and external validity issues. Figure 5.3 provides a graphical summary of all the results. It highlights that the majority of studies conducted are classified as low quality (and dominated by input-output and descriptive statistics studies), and these are the studies that tend to find large negative impact on various economic values.

Figure 5.3 Overview of water recovery studies by quality assessment and impact on economic values



Source: Authors' analysis
Note: * Economic values include GDP, GRP, GRIAP, employment numbers, farm production, farm gross margins (which may decrease with water recovery). Other economic values such as water market prices have the opposite sign as some studies suggest they increase under water recovery. Diagram is not to scale.

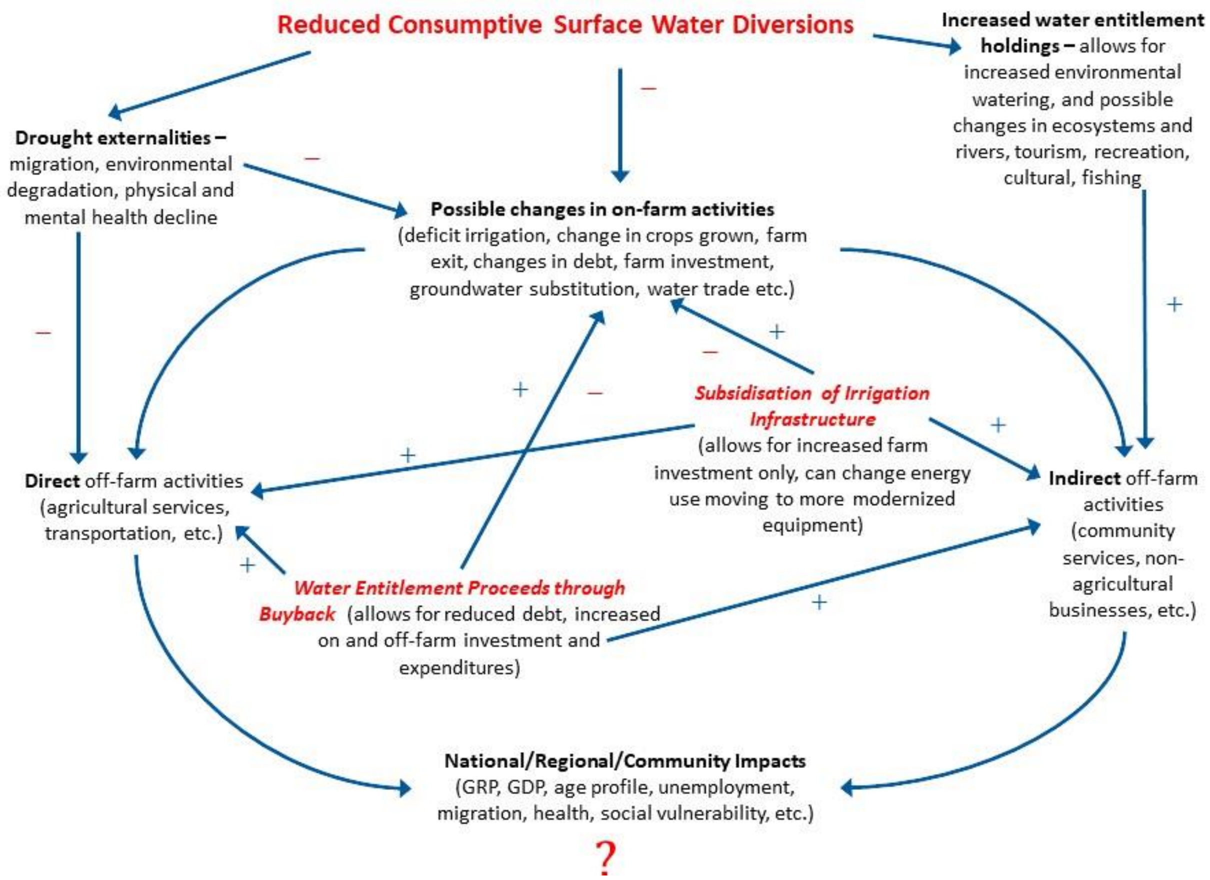
6 Discussion and Conclusion

6.1 Introduction to evaluating the economic studies conducted

This report has reviewed and highlighted the range of economic values within the Basin. Generally, the MDB provides substantial direct values such as agriculture and community economic values, recreational and tourism economic values, mining and energy economic values, high indirect economic values, and considerable non-use economic values. However, this range of economic values varies and is primarily impacted by current and future water policy, climate change and other drivers.

Among these drivers, the impacts of water recovery on economic outcomes in the MDB remain one of Australia's most contentious areas of interest. There are many that associate the Basin Plan and water recovery with being the sole/main reason for rural community decline. A cut in water allocated to agriculture is hence perceived by many MDB communities as the root cause of reduced irrigated areas, reduced irrigated production, lower irrigated value of production, fewer jobs in irrigated agriculture and local communities due to decreased spending overall, along with a decline in rural population due to out-migration. Figure 6.1 below illustrates the links between water and other outcomes.

Figure 6.1 The relationship between reduced water diversions through water recovery and various socio-economic outcomes



Note: Socio-economic outcomes to the environment, tourism, recereation, etc. are not included in diagram above

Source: Adapted from BDA Group (2010; 29)

The figure above illustrates that when analysts do not account for all the interlinkages that exist between different sectors, and allow for adaptation and reinvestment (especially of buyback), then it is impossible to truly evaluate regional and community impacts.

While there are direct and indirect costs of water recovery for irrigated communities, there are tradeoffs and costs under any policy. Where water recovery is fully voluntary, and adequately compensated for willing irrigators, irrigators are making choices that selling water is their best alternative at that time. Governments must focus on the most effective regional socio-economic programs that can be implemented to deal with reduced water allocations.

The flip side of the negatives associated with water recovery is that it can lead to significant benefits: improved environmental outcomes; tourism benefits; recreational benefits; cultural and heritage benefits; other socio-economic benefits, etc. – many of which are not valued by markets directly and can be extremely difficult to estimate. Indeed, the money received by farmers for water recovery has also been shown to have significant on-farm benefits – both from a reinvestment point of view from the sale of water entitlements (Wheeler & Cheesman, 2013) and farm productivity point of view from upgrading irrigation infrastructure and returning some water entitlements to the government (Hughes et al., 2020). It is far more difficult (and at times, not possible) to quantify the indirect benefits of water recovery for communities, than it is to identify the direct costs for rural communities from water recovery. However, our review of studies reveals that even quantifying the direct costs of water recovery in rural communities has been challenging, and extremely variable in quality.

It is notable that there is a clear divide between findings from the peer-reviewed academic literature, versus much of the consultancy literature, newspaper articles, or submissions to Parliamentary enquiries. Why is this the case? We do believe that most consultants have done the best they can, within the time-period, data available and available skills/knowledge that they have. But, it is also worth noting that some of these socio-economic reviews have cost a great deal of money.[15] The MDBA five-year Basin plan assessment work including the 2016 Northern Basin Review and 2017 Basin Plan Evaluation, in particular, cost over one million dollars in consulting fees. This work was shown to have significant faults (e.g., see the review in Wheeler et al., 2018a). Hence, one must judge the value (quality) gained from such an expensive economic analysis.

If we focus on the impact of water recovery on agricultural economic outcomes only (ignoring the difficulties of estimating indirect and non-market environmental values), the problem that **all analysts** face with trying to model the multi-faced outcomes is seeking to identify the **causal impact of water recovery**. What do we mean by causal impact? Causal impact is the relationship between two variables – how change in one variable impacts change in another variable. Correlation between two variables is not causality. Indeed, an association between two variables is not always causality. Causality is actually very difficult to determine, and Appendix B provides an overview of the different methods that attempt to

[15] For example, in the lead up to the Basin Plan five-year assessment in 2017, significant socio-economic consultancy work was commissioned by the MDBA. As noted on AusTender, Deloitte Access Economics was tendered the following to work on the Assessment of the Social and Economic Impacts of the Basin Plan: $238,600 in 2015 and $868,000 in 2014. Deloitte never finished this socio-economic assessment, with some funds transferred to KPMG to complete the work. In addition, KPMG was also paid to work on nMDB modelling in 2016 with $143,000, and $83,490 in 2015-16 to support advisory and modelling services on water use. In 2019, Aither were paid $934,861 for expert consulting and services to the Sefton et al. (2020) panel, and in 2019 MJA were paid $711,484 for modelling analysis on the effects of the water efficiency program. Indeed, other consultancy contracts include the four year 'integrated river modelling uplift program' of $27.4+ million with PwC.

identify causality. The more a method controls for as many influences as possible, the more a method has a large sample of data over time, and the more a method uses proper inference techniques, the more confident we can be about whether a causal relationship can be identified. In addition, there is a key difference between good quality economic analysis and economic impacts. Economic impact derived from a form of input-output analysis is rated as a very low-quality analysis. For example, the ABS stopped producing updated input-output multipliers in 1998-99, given their concerns with the misuse and application of such numbers. These concerns have yet to manifest in the minds of many consultants and departmental staff, given the number of input-output multiplier studies still conducted. The ABS outlines the following issues with input-output analysis: lack of supply-side constraints (overstates impact); fixed prices; fixed ratios for intermediate inputs and production; no allowance for people to adapt and respond to changes; absence of any budgetary constraints; not applicable for small regions. To summarise, "input-output multipliers may be useful as summary statistics to assist in understanding the degree to which an industry is integrated into the economy, their inherent shortcomings make them inappropriate for economic impact analysis. These shortcomings mean that input-output multipliers are likely to significantly overstate the impacts of projects or events. More complex methodologies, such as those inherent in Computable General Equilibrium models, are required to overcome these shortcomings" (ABS, 2021).

Similarly, we need to know whether all studies conducted have internal and external validity. Internal validity is associated with the existence of selection bias (e.g., have only some areas or years been selected for study); detection bias (e.g., has there been biased assessments of outcomes); attrition bias (e.g., have methodologies been undertaken correctly with proper testing and analysis). External validity is associated with the extent that any analysis can be generalised to other areas, which comes back to selection bias and the extent of the analysis conducted. Hence – the quality assessment framework that we applied to all key identified quantitative studies that have been conducted on the links between water scarcity and water recovery in the Basin is critical in guiding our understanding of what results we can rely upon for policy purposes (see Figure 5.3).

6.2 Key problems with many economic studies of water recovery to date

The bulk of the large-scale reviews to date (e.g., EBC et al., 2011; RMCG, 2016; Sefton et al., 2020; Productivity Commission, 2018; Wentworth Group of Concerned Scientists, 2017; KPMG, 2016; 2018; TC&A & Frontier Economics, 2017; Frontier Economics., & TC&A, 2022) have not managed to identify a causal relationship between water recovery and economic outcomes.

How various areas react to water recovery can be complex, as it is related to a variety of characteristics of rural viability. EBC et al. (2011) highlighted that rural communities least at risk of drought and water scarcity impacts were associated with the following characteristics: a larger population size; increased diversity in industry and regional economy; less dependent on irrigated agriculture; and locational factors. MJA (2020) showed that in the last decade most Basin communities followed the same trajectory 2006-2016 as to the path previous to 2006. In particular, MDB towns that had populations over 14,000 grew, and also became more diversified over the past decade. Those with populations 8,000-14,000 had mixed outcomes for their economies, with some growing and others shrinking, while those with populations under 8,000 were often found to be shrinking.

Appendix B provides an overview of all the different economic methods that exist and can be applied in order to investigate the impact of water recovery. It also provided a framework to investigate the quality and reliability of each study – to enable us to break down studies into low, medium, and high reliability. By definition, we can place much more reliability/trust on studies that attempt to model the dynamic nature of change in agriculture and the economy, than studies that do not. Descriptive studies obtain the lowest possible score in general – this is because by definition, they are based on very simple statistics and assumptions. They do not account for any adaptation, any confounding variables (e.g., anything other than the two variables in question), or other changes.

Our ranking and assessment of internal and external validity issues with various economic studies revealed some key consistent problems. Such problems have been previously identified in Walker (2019); Wheeler et al. (2018a); AAS (2019) and MJA (2019). We build on the review started in Wheeler et al. (2018a), and outline and discuss the two biggest problems below, as well as highlight the difference between the quality of various studies.

1. *Falsely assuming a proportional relationship between Water Use and Farm Production; while Overestimating Buyback costs and Underestimating Irrigation infrastructure costs*: Failure to recognise the true production relationships between water and agricultural outputs and characterising production changes as directly proportional to water availability. This is not borne out in practice or in tested theoretical contexts. There are a number of positive economic impacts of adjustment mechanisms, such as buyback, and the consequent positive impacts of spending within communities, while at the same time there are a number of negative impacts of infrastructure subsidies (such as return flows). Studies often ignore the benefits of buyback while also ignoring the full social costs of irrigation infrastructure.
2. *External and Internal Modelling Validit*y: Sample selection exists where specific ill-affected (in terms of reduced irrigation use) communities or community members are chosen and then presumed to be representative of a wider population (while not including other communities that may have benefitted from increased environmental water). In addition, less-than-rigorous statistical approaches that confound mis-specified assumptions about hydrological, agricultural and/or economic relationships contribute to the lack of validity.

6.2.1 Falsely assuming a proportional relationship between Water Extractions and Farm Production and consequently overestimating Buyback costs whilst underestimating Irrigation infrastructure subsidy costs

6.2.1.1 Studies that were ranked low quality and unreliable to be used in policy advice

The key issue when assessing water recovery is that reductions in farm revenue are much less proportional than the reduction in water availablility. The studies that are cited a lot in terms of water recovery's impact on regional economies, irrigated production and irrigated employment include KPMG (2016, 2018); MDBA (2016c); RMCG (2016, 2021); TC&A and Frontier Economics (2017); and Frontier Economics and TC&A (2022). The majority of these studies assume unit elastic response - a direct proportional relationship, namely – a 1% decrease in water extractions leads to an equal 1% decrease in irrigated hectares, which subsequently results in an equal 1% decrease in irrigation production. Often an input-output model is then applied – to suggest loss of regional economic value and jobs.

For example:

- TC&A and Frontier Economics (2017) suggested that water use in the GMID without buyback would have been 29-31% higher from 2013/14 to 2015/16, and GMID milk production could be expected to have been about 30% higher than was observed.
- Frontier Economics and TC&A (2022) assume a counterfactual case, and state that if it hadn't been for buyback (adjusted for trade), then water use would have been 46% higher in the GMID. However, this is an assumption only and not borne out by the figures reported in Figure 20 of the same report which suggests around a 25-30% decrease in the counterfactual. Needless to say, the authors use this figure and say that if there had not been this 46% reduction in water use, there would have been a 46% increase in milk produced.
- RMCG (2016; 36) states a counterfactual of a drop in 234GL in water available for dairy use in the GMID due to the Basin Plan, and that this translated directly into 440ML of lost milk production (based on average amounts of water needed per dairy cow). They then assume an 'average' milk price, and say this is a reduction in the annual farm-gate value of dairy production by $200 million, with the mixed farming and cropping sector losing a value of $25 million, and then an input-output model is used to suggest a total loss of $580 million per year and the loss of 1,000 jobs across the region.

Similarly, DPIE input-output modelling reported in Aither (2021) suggested over a $7 million loss to regional economies plus substantial job losses from implementing floodplain harvesting policies in NSW.

These figures quite rightly upset many people in rural and regional communities (and urban communities) when they are discussed and circulated, as no one wants rural communities to suffer. In addition, these are the only sorts of figures that are repeated in rural newspapers, with very little to zero commentary ever provided on more balanced assessments.

But are such figures of socio-economic impact correct? The answer is **unequivocally, no**. Indeed, they have all also been rated as 'low quality' in our quality assessment. The reason why is that the majority of farmers make decisions every year on how to maximise their farm production and they regularly adapt to changed situations. These situations include a changed climate; changing commodity prices; changing input prices; water use; technology; irrigation infrastructure; trade; diversification; off-farm income; reinvestment etc. As such, because of this, it means that when there is a shock (or a decrease) in water allocations, this does not fully translate into a) the same amount of reduced irrigated area; b) the same amount of reduced irrigated farmgate value; and c) a larger impact on regional economic value. Studies such as the ones described above were all rated in our review as low quality because of their method. Their static nature does not account for dynamic effects, for factor movement, for structural adjustment and for productivity change. Essentially, the counterfactual in these studies measuring the impact of water recovery is not valid because of the static, simple and unrealistic assumptions.[16]

Let's provide a very simple worked-through example of the data provided in Table 2.1 of this report. This table provides information on the agricultural businesses, irrigation businesses,

[16] The counterfactuals put forward are usually incorrect due to a) surplus water; b) farm exit and downsizing, and c) farmer adaptation; hence trade would not be in the direct proportion of the environmental buyback. The second assumption regarding the relationship between water and production, and profits is also incorrect.

irrigated area, extraction volumes, gross value of agricultural production (GVAP) and gross value of irrigated agricultural production (GVIAP) by the ABS from 2005-06 to 2020-21 for the MDB. Undertaking some simple descriptive statistics on this data, we learn:

- The correlation between irrigated area and water volume applied in the MDB is 0.97 – hence, this is an extremely high correlation, and tells us that we have an almost perfect positive correlation.
- The correlation between GVIAP and water volume applied is 0.44 (medium correlation), and between GVAP and water volume is 0.21 (namely low correlation).
- The correlation between accumulated water bought back in the MDB and volume extracted is 0.44; and between water bought back and irrigated area is 0.36 (medium to low correlation).

What does this tell us? It tells us that water volumes, irrigated area, and GVIAP move in the same direction (which is the relationship that consultants will cite in their studies). However, the correlation factors also tell us that accumulated water recovery volumes are positively associated with both volumes extracted and irrigated hectares – which seems to contradict the view that recovery decimates both extraction volumes and irrigated area. Hence, these results emphasise that **correlation is not causation**, nor is it an understanding of proportionality in the relationship. In other words, a 1% change in water extractions does not lead to a 1% change in hectares irrigated, to a 1% change in farm production, to a 1% change in farm profitability, and so on. **Hence, making simple assumptions based on such relationships can be significantly misleading.**

We can broadly illustrate this point by providing a very simple regression analysis of the elasticity relationship of how production (in terms of what irrigators do with irrigated area and GVAP or GVIAP) responds to changes in water volume. This is calculated as the percentage change in the quantity of production (namely irrigated area/GVIAP or GVAP from 2005-06 to 2020-21), divided by the percentage change in water extracted. We also check how changes in the commodity price index[17] affect both GVAP and GVIAP. Hence, estimating a number of very simple log-log and linear-log regressions between various pairs of two variables, we find:

- Every 1% increase in water extraction in the MDB was associated with a 0.68% increase in irrigated area.
- Every 1% increase in water extraction in the MDB was associated with a 0.18% increase in GVAP, and a 0.36% increase in GVIAP.
- Every 1% increase in commodity prices in the MDB was associated with a 1.2% increase in GVAP; and a 0.9% increase in GVIAP.

So – *just using simple regression analysis of elasticity responses* – irrigated area changes were only about two-thirds of any water extraction change, GVIAP change was only about one third of any absolute change in water extractions, and GVAP change was only associated about one fifth with any water extraction change during this time period. At the same time, a 1% change in commodity prices is associated with around six (three) times more change in GVAP (GVIAP) than water extraction change. Please note: these examples are provided for simple illustrative reasoning only, and more sophisticated analysis must be conducted to fully understand elasticity responses, using larger sample sizes, by industry and controlling for various confounders to be able to more accurately predict the actual relationships. It is highly

[17] Commodity price index was sourced from ABARES (2021b). Price received index is the simple average for hay, horticulture, cotton, viticulture industries for Australia, with base year in 2019-20.

probable that these simple elasticity estimates between water extracted and the economic outcomes above are overestimated (especially given peer reviewed work in this space – which we describe further below). Some industries will be impacted more than others. Modelling over the longer time period is left for future research.

6.2.1.2 Key Findings from Alternative Theoretical and Empirical Studies Results

In this section below, we provide six key findings from the peer-reviewed high-quality literature.

1. *The relationship between water extracted and farm economic outcomes (irrigated area and revenue) is not unit elastic*

Many of the dynamic and empirical modelling studies of water security have employed a range of MDB historical data, over time and by industries, and have also accounted for other variables that may influence the relationship between variables. Some studies use dynamic and partial equilibrium and theoretical modelling, and others use empirical modelling. For example, Connor et al. (2014) modelled elasticities for irrigation area varied from 0.09-0.42 – suggesting that, for a 1% decrease in water allocations, irrigated area decreased by 0.09-0.42% across various agricultural industries. In terms of irrigation revenue, for a 1% decrease in water allocations, all commodities revenue only decreased by 0.1%. Modelling of impacts prior to the Basin Plan implementation by Adamson et al. (2011), Dixon et al. (2009) and Wittwer and Griffith (2011), estimated reductions in revenue to be about 0.4% and 0.2% respectively for each 1% reduction in available water. Similarly, ABARES (2011a) found minimal impacts as a result of buybacks. Wentworth Group of Concerned Scientists (2017) modelling found the following: a 1% increase in commodity prices increased GVIAP and GVAP by 0.8% and 0.6%, respectively; a 1% increase in water recovery reduced GVIAP by 0.04% and was insignificant on GVAP; a 1% increase in water recovery reduced irrigated area by 0.8%; while a 1% increase in water allocations increased irrigated area by 0.2%; KPMG (2018) modelling found that the relationship between a 1% increase in agricultural irrigated production was associated with a corresponding increase in agricultural labour FTE of between 0.01-0.44%. Grafton and Jiang (2011) revealed that the decline in irrigated agriculture profits is much less than the proportional decline in surface water extractions.

Indeed, the difference between GVIAP and GVAP is critical, as outlined by Wittwer (2019). First, dryland farming contributes to the local economy two times greater than irrigated agriculture. It is estimated that dryland agriculture accounts for 70% of the agriculture output within the MDB (ABS, 2008). Second, farmers adapt to increased water scarcity. Furthermore, rather than revenue or gross value of production, profitability is the key factor in determining the ongoing resilience of farm businesses. Very little studies to date have focused on overall farm profitability.

In the long run, elasticities of change will always be greater (more elastic) than in the short run – as changes that were impossible to make in the short term (e.g., restructure, capital adjustment etc.) become more realistic over a longer time frame. Hence, it is important to look at elasticities over the entire time period, rather than focussing on individual years.

2. *Impacts of buyback expenditure within the local economy are often ignored*

Following on from above, one of the key reasons studies such as KPMG (2016), RMCG (2016), TC&A & Frontier Economics (2017), Frontier Economics & TC&A (2022) overestimate the cost of MDB water buyback is that they ignore local benefits from local expenditure by farmers resulting from water buyback compensation, and other farm adaptation in general. This is an important omission, as assumptions are made in CGE modelling for example to account for expenditure within local economies, as though one particular sector may decrease, another sector may benefit. As outlined by Wittwer (2019), for every lost dollar of irrigation output, farm factor movements result in an increase in dryland production of about half a dollar. And, there is evidence to suggest buyback compensation was, in fact, spent locally (e.g., studies such as Wheeler & Chessman (2013) show this).

3. *Not all farmers who sold water entitlements left farming, or suffered changes in production*

Wheeler and Cheesman (2013) and Zuo et al. (2022) – who surveyed thousands of irrigators across the sMDB between 2008-09 and 2015-16 – provided convincing evidence that many irrigators who sold water to the Australian Government continued farming in the sMDB. These studies also highlight the difference between selling water entitlements, consequent reductions in water extractions, consequent reductions in irrigated area, and consequent reductions in irrigated value. Wheeler and Cheesman (2013) highlighted that farmers predominately sold their surplus/buffer water (water not used in production). Furthermore, water sales proceeds have been used to reduce debt (and hence interest payments), or to restructure and reinvest on farms. Irrigation infrastructure subsidies do not allow for debt repayments. Consequently, for reasons such as this, and the use of CGE modelling, Dixon et al. (2011) suggested that buyback would increase economic activity in the sMDB (e.g., positive effect on household consumption), with little effect on aggregate sMDB farm output and national macro-effects on GDP. This conclusion is very similar to what others have found, namely that the outward trading of water may have had a minor impact on declining productivity during the assessment period, but it was small compared to the influence of the drought (NWC, 2012). This is also why Wheeler et al. (2014a) found only very weak to no significant impact of selling water entitlements on MDB farm viability, though it is important to note, these questions need updating with more recent data when available.

4. *Climatic and socio-economic factors are often a lot more important than water allocations for socio-economic outcomes*

Wittwer (2019) emphasised that droughts have much larger impacts on rural economies than buyback. Wheeler et al. (2020b) analysed farmer exit in the MDB from 1991 to 2011, using spatial regression modelling at the statistical local area level to assess the impact of weather, economic and water factors on net farmer number changes. They found that the direct drivers of farmer exit in local areas were climatic (e.g., increases in maximum temperature and increased drought risk (through decreased long-term precipitation skewness and increased long-term precipitation kurtosis)) and socio-economic (e.g., decreases in commodity output prices, increased urbanisation and higher unemployment). On the other hand, absolute rainfall, changes in irrigation water diversions and water trade movements had no significant impact on MDB farmer exit. This study focused on total farmers – namely both dryland and irrigated farmers – given that when many farmers exit irrigation they often turn to dryland farming instead (Wheeler & Cheesman, 2013). Although the Commonwealth had recovered over 60% of water entitlements by 2011, further water reform may continue to have a more significant impact on rural communities, especially smaller irrigation communities, and further high-quality research is needed to quantify this impact. Such research is not possible without water recovery data, by program, being provided at the smallest area possible.

5. *Buyback negative impacts are often overstated, whilst broader impacts from irrigation infrastructure subsidies are often understated*

Points 1- 4 above provide many examples of why buyback impacts are often overstated. Schirmer (2016), (2017a), etc. reported that the majority of irrigators answered that selling water had had a positive impact on their farm, particularly in relation to reducing debt, reducing stress levels, and improving their life, finances, and farm enterprise as a whole. Irrigators were slightly more likely to rate better on-farm outcomes for transfers of water entitlements compared with irrigation infrastructure upgrades.

Unfortunately, rural and regional media have played a significant role in enforcing this negative perception of buybacks – with at least one article every couple of weeks discussing the danger of buybacks. The concept of buybacks is almost always reported negatively, and with a lot of fear mongering. On the other hand, while there has been research on the farm productivity benefits of irrigation infrastructure subsidies (e.g., it has been found that on-farm irrigation infrastructure expenditure has had a positive economic impact on farm productivity (Hughes et al., 2020) and the community (Banerjee, 2015), and that it has increased employment and real GDP in the local economy and on-farm productivity (DAWR, 2017; MJA, 2017)); many of the negatives of irrigation infrastructure subsidies have been ignored. Wheeler et al. (2020a) provided a summary of this, such as increased cost; poor governance; reduced return flows; a rebound effect; increased entitlement utilisation; increased substitution, reduced equity; increased floodplain harvesting take; and reduced resilience. A number of people have argued that the water infrastructure subsidies have resulted in smaller increases in the net stream and river flows than is estimated by the Australian Government and may even have reduced net stream and river flows (Williams & Grafton, 2019).

Wheeler et al. (2020a) found that those who received an irrigation infrastructure grant increased their water extraction volumes and rates (by 21-28%) relative to other irrigators. Receiving an infrastructure grant increased the likelihood of irrigators expanding their enterprises (i.e., increased irrigated area, buying farmland and water entitlements), as well as changing their crop mix, compared to non-grant recipients. Grant recipients were less likely to adhere to contractive strategies, such as decreasing irrigated area and selling farmland, although these differences were not statistically significant. Hughes et al. (2020) also found a similar impact, and a higher farm water demand. Schirmer (2017a) found that MDB irrigators who received an infrastructure grant were significantly more likely to: increase irrigated farm areas and irrigation efficiency; purchase new land; and intensify production. They were more likely to experience negative impacts such as increased farm debt and electricity/power costs from modernisation; and experience a loss in the last year (and over the previous three years).

Analysing this with CGE modelling, infrastructure upgrades were found to be inferior to public spending on health, education and other services in the Basin (Wittwer & Dixon, 2013), with at least 2-3 times more jobs possible through such public spending than on irrigation infrastructure. Indeed, it is strongly argued that communities conflate the impacts of buybacks and drought. Using CGE models, it is clearly shown that drought is more severe in the regional impact than buybacks (Wittwer, 2011a; Dixon et al., 2009).

6. *Healthy rural communities depend on many other factors than water for irrigation*

As noted earlier, Wheeler et al. (2020b) found that the direct drivers of farmer exit in local areas were climatic (e.g., increases in maximum temperature and increased drought risk (through decreased long-term precipitation skewness and increased long-term precipitation kurtosis)) and socio-economic (e.g., decreases in commodity output prices, increased urbanisation and higher unemployment).

Wittwer and Young (2020) is one of the latest economic modelling studies of water recovery, undertaken for the Independent Socio-Economic MDB Panel. Wittwer used an updated version of TERM-H2O to model two scenarios: 1) obtaining the remaining water recovery target through infrastructure only; and 2) spending the same amount of money on regional services instead, between 2020 and 2024. It was found that scenario one had a net present value (NPV) welfare loss of $1.1 billion (but increased jobs up to 1,000 in the short term, and 100 in the medium term); while scenario two found that each dollar spent on education, health and community services created four times as many jobs as spending on infrastructure, and had an NPV welfare loss of $0.125 billion (nine times less than spending on infrastructure). Note, no welfare benefits to increased environmental water were considered.

6.2.2 External and Internal Modelling Validity Issues

Within our review, there were many internal and external validity issues identified in the economic modelling studies. We list some of the issues here (though note this is not fully comprehensive):

1. *Small sample sizes*: Numerous studies have used small sample sizes to model economic relationships. A case in point is our own very simple analysis in this chapter. Basically, it is generally recommended that a minimum of 100 observations are needed to be able to draw reliable results. Modelling of very low numbers of observations has occurred in modelling water markets, irrigated area, and irrigated businesses (e.g., Agbola & Evans, 2012; MDBA, 2016b; Wentworth Group of Concerned Scientists, 2017; Aither, 2017a; RMCG 2016 etc.). If the timeseries nature is not taken into account, and the small sample size precludes other variables that can be included, then the high value of R^2 and the significant independent variables in the OLS model are potentially due to spurious regression (Granger & Newbold, 1974).
2. *Statistical modelling issues*: As has been outlined in the previous section, many studies (e.g., KPMG, 2016, 2018; MDBA, 2016b; RMCG, 2016; Frontier Economics, & TC&A, 2022) employ less than rigorous statistical approaches that confound mis-specified assumptions about hydrological, agricultural and/or economic relationships. There is no noted checking regarding issues around collinearity, heteroscedasticity or serial correlation (or where tests were done, substantial concerns surround the tests conducted) – bringing into question the validity of the modelling results.
3. *Causal policy impact*s: Following on from the above point, methods used to estimate the impact of water recovery on economic outcomes cannot provide a causal impact because there was not a proper model for the counterfactual. For example, if we take the case of water markets, to be able to derive a causal impact of recovery on prices, then the water allocation price model should be estimated based on the period without Commonwealth purchases, which can be used to predict the price for the period with Commonwealth purchases (see Baerenklau et al. (2014) for an example in another setting). The difference between the predicted price and observed price for the period with Commonwealth purchases can be concluded as the impact of Commonwealth purchases. However, to be able to use this approach, sufficient data are needed to generate a robust prediction model for the period without Commonwealth purchases – and small sample size observations mean this is impossible. Much of the current economic modelling in this space does not model the counterfactual at all well, and hence their predictions are suspect.

4. *Sample selection biases*: Sample selection biases, where only certain regions, areas etc. are selected for modelling purposes and then used to represent wider community results. An example of this is KPMG (2016), where out of a total population of 67 communities in the nMDB, the authors selected 15 specific 'ill-affected' communities for modelling regarding SDL reductions. By not modelling all available data, any spatial or spill-over influences are ignored, and the results cannot be deemed representative.
5. *Inadequate documentation and no independent peer review*: There was a lack of referencing and attention to detail in most consulting reports, which made it hard to review and check data sources. Many studies were rated poorly in this area. Although some consultancy studies were peer reviewed, there were not many, and indeed, there were also issues in the fact that the peer review was not conducted by experts in either MDB water economics or CGE modelling. In addition, at times peer review comments were not properly addressed (e.g., Blackwell et al., 2018 provided a summary of the response to their peer review).

6.2.3 Summary of Quality of Studies that Predict Large Socio-Economic Impacts of Water Recovery versus Studies that predict more Nuanced Impacts

In summary, within our quality assessment and ranking exercise, all studies that assumed a direct proportional relationship, improperly estimated counterfactuals of water extraction, applied input-output assumptions of multipliers to estimate impacts on GVAP and GVIAP were ***all rated as low quality***. In other words, they did not receive enough points to clear 40% under our quality assessment. On the other hand, when evaluating the studies that did not find the same large socio-economic impacts of water recovery, only a small proportion of these studies were rated as low quality, with the remainder split between high and mid quality.

6.3 Further research, recommendations and concluding comments

It is impossible to do justice to a section that identifies future research needs, hence this section is kept deliberately brief. Based on our review of the literature, what is obvious is that the bulk of the research money has been spent on investigating the direct costs to irrigation communities, and trying to identify the costs to regional areas stemming from water recovery to the environment. However, what has been highlighted by this review, is that the bulk of work in this space has been of low quality, largely because of their short-term nature and the methods used. This is especially the case of funded consultancy research by various governments and industries. This leads us to our first recommendation for the MDBA:

> **Recommendation 1: In assessing water buyback socio-economic impacts, the MDBA should work together with the Department of Agriculture, Fisheries and Forestry, the Department of Climate Change, Energy, Environment and Water, along with relevant state governments, to consider adopting a standard for research evidence – thereby gaining more assurance from peer-reviewed studies that sufficiently address causality issues with creditable methods.**

Part of these criteria should consider recommendations from groups such as NSW Treasury (2017) and ABS (2021) to avoid using input-output modelling wherever possible to measure socio-economic impact. There also needs to be attention given to increasing the availability of datasets for research entities to access, and providing researchers with enough time to produce high-quality research. ABARES provision of its water trade and water recovery data in ABARES (2021a) is a very good example of increasing and facilitating data openness and transparency, although there needs to be greater attention towards providing researchers unit-level access to their irrigation farm datasets.

There has been far less attention paid to understanding the economic benefits of water recovery to other groups – such as First Nations, downstream communities (e.g., dredging costs, marine fishing costs, benefits of salinity export), recreational, tourism and other economic benefits. The lack of economic research on the outcomes of regional diversification fund expenditure is also another area that needs significantly more work, which is our second recommendation:

> **Recommendation 2: Additional research on (a) economic benefits of water recovery for First Nations people and country, and downstream communities; and (b) outcomes of regional diversification fund expenditure are of high priority to assist in providing a balanced and evidence-based view of Basin regional economy development implications of water recovery for the environment.**

In addition, there has been some important knowledge gaps, such as the impacts of infrastructure on return flows, and the rebound effect on water extraction from subsidies – such that some policy options can markedly overstate water recovery. Although it is critical to understand the socio-economic costs of water reform, as this is important for rural communities, because without knowing causal impacts and drivers, it is difficult to then fully develop effective policies that can be employed to deal with existing distributional issues. At the same time, it is not warranted to invest significant resources in only one side of the water recovery equation. While it has been acknowledged that attempting to monetise environmental/economic/social benefits of water recovery at higher levels is complex (e.g., MDBA, 2016b), it is worth noting that there are a range of different existing stated and revealed preference economic techniques that could be employed and invested in. Not making communities aware of these results – while instead reporting what we consider to be overinflated and incorrect measures of job losses and GDP reductions – means that a balanced perspective of the possible outcomes of water recovery is prevented.

Areas of research into the economic impacts of water recovery that we think should be invested in further include: water market dynamics (impacts on prices and volumes traded across the MDB); socio-economic outcomes of, and participation in, water markets and water reform across regions, specifically addressing causality issues; cultural water recovery benefits; path dependency of selling water entitlements by farmers; stranded asset and restructuring needs; climate change adaptation; water consumption and extraction changes; surface-groundwater interaction and substitution; irrigation area changes; longer on-farm and off-farm productivity impacts of infrastructure investments; corruption and governance issues in irrigation infrastructure programs; key insights into factors associated with stronger irrigation infrastructure recovery programs; and environmental outcomes.

In particular, careful attention must be paid to the methods (and data) used to assess the causal impact of water recovery. The focus must be given to high quality, longitudinal, large sample sizes, and dynamic assessment that attempts to robustly identify the causal impact of water recovery – considering as many other confounders as possible. Work should be independently reviewed by external experts.

In addition, as has been emphasised numerous times in the economic literature, to ensure greater welfare from optimised economic outcomes, it is essential that attention is given to strengthening the MDB’s economic and social water institutions. Going forward, meta-governance frameworks and the sequencing of any water reform are crucial, given the feedback loops that exist between stakeholder behaviour and any resource consumption or change in policies (Wheeler, 2022).

Finally, it is critical to note a passage from the *Water Act 2007*, which is considered by some to be one of the greatest water reform achievements by Australia. In Part 2, Division 1, Subdivision B, Section 21, Subsection 4 (b), "*Basis on which Basin Plan to be developed:*

> *(4) Subject to subsections (1), (2) and (3), the Authority and the Minister must, in exercising their powers and performing their functions under this Division:*
>
> *(a) take into account the principles of ecologically sustainable development; and*
>
> *(b)* ***act on the basis of the best available scientific knowledge and socio-economic analysis****"* [bold emphasis added].

In the past decade or so, we have heard a lot about the need for the best available scientific knowledge to guide water recovery. On the other hand, there has been less emphasis/discussion on the need for the best available socio-economic analysis. There has also been a lack of understanding and focus on trying to create the best socio-economic analysis, despite the millions of dollars that has been spent on consultancy studies. This leads us to our final recommendation for the MDBA:

> **Recommendation 3: A focused socio-economics research program to address high priority questions can effectively inform MDBA planning implementation, and communication. This will involve assessing:**
>
> **a) How can the impacts of existing water recovery on communities – both positive and negative – be creditably documented and communicated to the public?**
>
> **b) What is the best way to achieve further water recovery – considering implications for rural community socio-economic wellbeing costs and benefits across the whole impacted population?**

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Appendix A: NRM codes

MDB region 2011 and 2016 NRM codes

NRM codes 2016-2021		NRM codes 2007-2015	
Code	**Region label**	**Code**	**Region label**
101	Central Tablelands	101	Border Rivers-Gwydir
102	Central West	102	Central West
105	Murray	105	Lachlan
108	North West NSW	106	Lower Murray Darling
109	Northern Tablelands	107	Murray
110	Riverina	108	Murrumbidgee
112	Western	109	Namoi
204	Goulburn Broken	113	Western
205	Mallee	204	Goulburn Broken
206	North Central	205	Mallee
207	North East	206	North Central
210	Wimmera	207	North East
303	Condamine	210	Wimmera
309	Queensland Murray Darling Basin	301	Border Rivers Maranoa-Balonne
312	South West Queensland	305	Condamine
407	South Australian Murray Darling Basin	312	South West Queensland
801	ACT	407	South Australian Murray Darling Basin
		801	ACT

Appendix B. Details of Economic Modelling Techniques

Optimisation and mathematical models

Computable general-equilibrium (CGE) models are increasingly used to estimate the whole-economy economic impact, with the availability of various economic data, software, and computing power. CGE models specify a system of equations representing production, consumption and investment components of the economy. CGE models can also specify economic sectors and geographic regions, in which factors of production move freely to sectors and regions where the returns are highest. CGE models' main advantage is their theoretical foundation and ability to capture all channels through which economic theory suggests policy interventions may operate, to generate counterfactual scenarios, and to conduct welfare analysis (Carbone et al., 2020).

Partial equilibrium models (such as hydroeconomic models), on the other hand, deals with how changes affect a particular firm, sector or region. They consider the equilibrium in one market, assuming exogenous prices in other markets. The main advantages of partial equilibrium models are simplicity and a more detailed representation of one sector than CGE models. However, the flow-on effects on other sectors cannot be captured and the assumption of independent markets may be unconvinced in some circumstances. Settre et al. (2017) provides an overview of hydroeconomic models that have been used to analyse water issues in the MDB.

Mathematical programming models are a popular approach for impact analysis in the literature. Mathematical programming models for irrigation agriculture are able to link crop simulation and other biophysical models with economic models of irrigator behaviour. Positive mathematical programming models have recently been widely used to analyse irrigation agriculture since they integrate the multitude of resource, policy, scenario, and environmental constraints often observed in reality (Qureshi et al., 2013b; Sapino et al., 2020)

An input-output (I-O) model is the simplest kind of general-equilibrium model in which all economic interactions are characterised by fixed coefficients. While the I-O model's main advantage is its simplicity, it has several distinct limitations, such as the strong assumption of linear relationships between inputs and outputs, and its static nature suitable only for simulating short-run impacts. The consensus regarding I-O models is that they tend to *overstate impacts and focus mainly on short-run outcomes*, and in particular, multipliers derived from I-O models m*ay be misleading and deviate from what might be expected to occur in reality* as a consequence of policy interventions to the economy (James, 2017). NSW government guide to CBA (NSW Treasury, 2017) highlight the significant limitations of I-O analysis, and suggest extreme care should be taken in any interpretation of its results, and ABS have stopped updating their I-O multipliers since 1998-99, due to their misuse in many policy advice (ABS, 2021).

Econometric models

Econometric models are a frequently used technique for observational and experimental data analysis to estimate how the change of an explanatory variable (i.e., the policy or intervention variable or a water scarcity measure) is associated with the change of the dependent variable (i.e., the outcome variable). Studies using econometric models can have nations, regions, farms and farm plots as observation units, analyse cross-sectional, time series and also panel datasets, and fit different functional forms of the data being analysed.

Valuation of non-market benefits and cost techniques

Within economics, there has developed a range of econometric techniques for investigating and valuing non-price impacts – which can be especially important for investigating impacts of water recovery for the environment. Examples of such techniques include both revealed or stated preference methods:

- **Revealed preference method** uses either survey observations of actual expenditure choices or information from market prices and characteristics to reveal the willingness to pay derived from differences of expenditures with marketed goods and services. The most popular method is the travel cost method that can reveal the willingness to pay, for example, for a recreational site according to the consumers' travel costs to that site. Another method is the hedonic pricing model, which is often used to value bundled products where the values are calculated based on market prices and environmental characteristics.
- **Stated preference method** is based on surveys that simulate market situations in which survey participants can choose what amount they would spend for certain environmental conditions. Popular methods include contingent valuation and choice modelling. Both approaches have been used to value native vegetation, as well as farmers willingness to pay for different environmental vegetation schemes.
- **Damage, replacement and substitute cost methods** are all related methods that estimate values of ecosystem services based on either: the costs of avoiding damages due to lost services, the cost of replacing ecosystem services, or the cost of providing substitute services. These methods are not providing an 'economic value', but they do give a firm foundation of the cost of avoiding damages or replacing ecosystems.

Econometric estimation models

Studies using econometric methods commonly use three types of data formats, i.e. cross-sectional, time-series, and panel (also known as longitudinal). The most frequently used cross-sectional datasets are farmer surveys that specify an individual farmer as a cross-sectional unit. Cross-sectional units can be surveyed repeatedly at multiple time intervals to generate a panel dataset. But panel farmer surveys are rare because it is expensive to survey the same farmers and attrition rate may be too high to ensure a representative panel dataset. Hence, another popular panel dataset format uses information repeated (usually annually) for the same geographic area (e.g., Natural Resource Management area, Statistical Area 2, water trading zones, etc.). Time-series data are often from aggregate water market transactions at different time intervals such as daily, weekly, and monthly. A few studies also use annual time series with less than 20 years of observations.

Estimation in econometric studies has a wide range of models depending on the nature of the dependent variable and data format. Ordinary least squares (OLS) is often used for a continuous dependent variable and cross-sectional dataset, given classical assumptions are not violated. Limited dependent variables are common in econometric studies. For example, probit models are used for binary dependent variables; tobit models for censored dependent variables; multinomial logit models for categorical dependent variables. Some models can extend the estimation to panel datasets such as fixed and random effects linear panel models, and random effects panel tobit models. Additional tests for time-series data are usually required to check stationarity of variables and co-integration relationships between variables.

Observational and experimental techniques

Observational studies and experimental studies are the two major research designs used to estimate the causal relationship between the intervention and the outcome. While experimental studies can ensure the random assignments of intervention among study subjects, which makes the causal estimation straightforward, they are usually more expensive and have a shorter study period than observational studies. Hence, much of the econometric literature has been devoted to analysing observational data with causal inference. Four popular techniques have been used in economics to estimate causal effects using observational data: (1) natural experiments, (2) instrumental variables, (3) regression discontinuity, and (4) difference in differences.

Natural experiments are circumstances under which there is a divergence in regulations or practices between studied subjects (i.e., nations, regions or individuals). Unlike experiments such as randomised controlled trials, researchers cannot assign participants to 'treatment' and 'control' groups under natural experiments. Rather, differences in regulations or practices provide the opportunity to treat specific subjects as part of an experiment. The validity of natural experiment studies depends on the assumption of random assignment of 'treatment' and 'control'. In practice, it is often complex, and the researcher must seek to determine the degree of randomness and any bias that may be introduced. There is a large literature on propensity scores, which are estimates of the probability of treatment as a function of "observed" characteristics and are used to correct the non-randomness of 'treatment' and 'control' assignment in natural experiments.

In the instrumental variable (IV) approach, one needs to identify a valid instrument for the endogenous treatment variable. A variable is called an instrumental variable for the endogenous treatment variable if it is uncorrelated with the error term of an IV model and it is correlated with the endogenous treatment variable. The 2SLS (two stage least squares) estimator is a commonly used IV estimator and can be obtained by two consecutive OLS regressions. The first stage predicts treatment as a function of instrumental variables along with other covariates, which obtains the predicted value of treatment; the second stage OLS regression predicts the outcome as a function of the predicted value of treatment and other covariates.

The regression discontinuity (RD) design is suitable when the treatment assignment depends on some threshold. We can estimate causal effects by comparing outcomes for studied subjects on each side of the threshold, and the difference in outcomes between the two sides is interpreted as the causal effect of treatment. Several assumptions are needed in order for the RD design to generate causal estimates. First, treatment is assigned based on an observed variable or index (such as age, asset value, etc.); second there is a discontinuity in the probability of being treated at some cut-off value of the assignment variable or index; third, the cut-off value for treatment is arbitrary and thus the only difference between studied subjects on either side of the cut-off is the treatment; and fourth the same cut-off is not used for determining treatment of other policies/interventions.

Difference in differences approach requires two groups, the treated and the untreated, and two time periods, before treatment and after treatment. There may be a number of covariates that affect the outcome for each group. This setting allows one to estimate what the outcome would be for the treated group if it were not treated, the causal effect of treatment. The validity of the difference-in-differences approach relies on the equal trend assumption, which states that no time-varying differences exist between the treatment and control groups.

Qualitative analysis, descriptive analysis

Qualitative methods are often used to assess impacts by collecting information about values, attitudes, views and behaviour, with the focus on 'why' and 'how' impacts are realised or not realised. Some qualitative studies collect information at two points in time, before and after the treatment. However, there is usually no control group since the 'why' and 'how' questions are only relevant for the treatment group. The attribution of impact to treatment is evaluated by participants' answers to whether and how the changes in outcomes are due to the treatment or to other factors at play.

Besides analysis of qualitative data, surveys are frequently used to collect quantitate data for impact assessments. Descriptive analysis such as two-way associations, two-sample mean/proportion comparison is used to compare the impacts between treatment and control groups. For example, Schirmer (2015) asked 'how has investment in off-farm water supply infrastructure in general affected your farm enterprises in the last five years' and provided seven Likert scale answers from very negative impact to very positive impact. Descriptive analysis was used to compare the Likert scale distribution between the respondents residing in areas without any infrastructure grants and those residing in areas with grants.

Benefit cost analysis (BCA)

Benefit cost analysis (BCA) is another economic technique commonly used to assess social welfare changes associated with a policy or project intervention, by evaluating all the relevant costs and benefits in present monetary terms. BCA was introduced in the United States *Flood Control Act 1936* as a means of appraising flood control projects. Benefits and costs in BCA are typically identified and valued in terms of the incremental change between alternative scenarios: a business-as-usual scenario, and the relevant policy scenario. Scenarios are compared in terms of the net present value, benefit cost ratio, and internal rate of return. BCA does not include secondary impacts in the cost and benefit calculations, such as economy-wide flow-on effects on output, employment, or income. The general principles of BCA can be generally formulated as a series of questions (Prest & Turvey, 1966). Which costs and which benefits are to be included? How are they to be valued? At what interest rate are they to be discounted? What are the relevant constraints?

All various economic methods described above can be combined within a BCA, in terms of informing the benefits and costs, and assumptions implicit within its framework. If undertaken correctly, all costs should be included as well as implicit costs that are not readily monetised when deciding to undertake an investment or not. The decision rule is that an investment should only occur in the discounted net benefits (benefits less costs) are positive and generate a rate of return in excess of alternative uses for the funds. Values that cannot be quantified must be carefully explained, and in some cases a threshold approach (i.e., estimating the minimum value that such benefits must be in order to justify the project being socially beneficial) applied.