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The rebound effect on water extraction from subsidising irrigation infrastructure in Australia

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ABSTRACT

Over the past decade, Australia has been buying water entitlements and subsidising irrigation infrastructure to reallocate water from consumptive to environmental purposes in the Murray-Darling Basin (MDB). There is considerable evidence that irrigation infrastructure subsidies are not cost-effective, as well as questions as to whether water extractions are increasing (rebounding) as a result. We used 2481 on-farm MDB irrigation surveys and identified a 'rebound effect' on water extractions, with irrigators who received an irrigation infrastructure subsidy significantly increasing (21-28%) their water extraction, relative to those who did not receive any grants. Although the precise hydrological impact of this rebound effect on catchment and Basin-wide extractions remains unknown, publicly available water data suggest that reductions in extractions from the MDB – supposedly commensurate with increases in environmental flows – may have been overestimated, particularly in the Northern MDB. This overestimation may in turn be linked to issues with water measurement and extractions at the catchment and Basin-scale, which occur due to: (1) water theft and poor enforcement; (2) inaccurate or absent water metering; (3) growth in unlicensed surface and groundwater extractions and on-farm storage capacity; (4) legal and practical uncertainties in compliance tools, processes and water accounting; and (5) complexity of floodplain, evaporation and groundwater interactions. To respond to these water governance challenges, MDB water and rural policy actions must: (1) improve measurement of diversions and develop transparent and robust water accounting, independently audited and accounting for uncertainty; (2) improve compliance, fines and regulation; (3) use multiple lines of evidence for water accounting and compliance; and (4) prioritise the cost and environmental effectiveness of water recovery.

1. Introduction

Addressing the world's growing freshwater demand within the context of finite supply and climate change is one of the most significant challenges facing humanity (Grafton et al., 2018). Irrigated agriculture is the largest user of water globally. As such, an improved proportion of water consumed in beneficial crop evapo-transpiration, relative to the water extracted or applied to crops – or a greater 'crop per drop' – is required. Increased irrigation efficiency – which is subsidised and promoted by governments around the world – is supposed to accomplish this goal by moving from inefficient irrigation methods, such as surface irrigation, towards more controlled systems, such as drip irrigation. This can afford opportunities to reallocate 'water savings' to other uses, such as urban consumption and the environment (Perry et al., 2017). While modernising irrigation infrastructure can provide private farm benefits, with improved crop yields, nutrient delivery and reduced labour costs (Meyer, 2005; Loch and Adamson, 2015), it does not necessarily generate basin-scale water savings. This is in part due to a phenomenon known as the 'rebound effect' (Gómez and Pérez-Blanco, 2014; Perry et al., 2017). Put simply, the rebound effect on water use occurs when increased water demand from improved water productivity outweighs technical efficiency savings, resulting in increased extractions.

The rebound effect is examined through the lens of water policy and management in Australia's food and fibre bowl, the Murray-Darling Basin (MDB or Basin). This Basin was chosen because, first, Australia has invested approximately $AUDS4 billion of public money in the modernisation of irrigation infrastructure in the MDB as part of a policy to return water to the environment without compromising agricultural productivity (DAWR, 2019a). Second, it is commonly assumed that an

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overall Basin-wide limit on water extractions, known as 'the Cap', prevents increases in Basin-scale water extractions or water consumption. This is because gross water extractions from watercourses and regulated rivers were set at 1993–94 levels of development (except Queensland, set at 1999-2000 levels and South Australia where the Cap was set at an average use of 90% of entitlements) (MDBA, 2018; AAS, 2019). Third, the Basin has been a leader in the development of demand-management economic incentives such as water markets (Quiggin, 2008; Seidl et al., 2020a).

With the passage of the *Water Act* in 2007 and its delegated instrument, the MDB Plan in 2012, the Cap was replaced with Sustainable Diversion Limits (SDLs) for each surface and groundwater resource in the Basin. The Act and MDB Plan were a direct response to the over-allocation of surface-water, with more than half the flows in many rivers in the MDB being diverted (end of system measure) (CSIRO, 2008). Under the original Basin Plan, 2750 billions of litres (gigalitres-GL) in long-term average annual yield (LTAAY)[1] was to be returned to the environment, with an additional 450GL, to be exclusively acquired through on and off-farm infrastructure upgrades, agreed to between Basin governments on the basis that it would not have a negative impact on rural productivity (Grafton and Wheeler, 2018). In 2018, legislative amendments reduced entitlement recovery to 2680GL; this figure was revised down by a further 605GL (subject to the implementation of 36 'supply measure' projects that are meant to offset water in exchange for 'equivalent environmental outcomes') (PC, 2018; Grafton, 2019).

The first iteration of water recovery by the federal government commenced before the MDB Plan was legislated and was provided for under the auspices of the *Water for the Future* program, a national water management agreement by federal and state governments. Under this program, two mechanisms were developed to remove water from the consumptive pool and return it to the environment; both have continued under the Basin Plan. The first, 'buybacks', involves the direct purchase of water entitlements from willing irrigators at market prices through open tender. Buying through open tender effectively ceased in 2013, with the focus shifting to 'strategic purchases' via closed negotiations with large corporate entities (with some limited exceptions). Some of these purchases have been criticised due to their lack of transparency, questionable environmental value (owing to the nature of the entitlements purchased) and potentially inflated values (Grafton, 2019; Seidl et al., 2020b). The second mechanism, subsidised irrigation infrastructure upgrades, is enabled through the Sustainable Rural Water-Use & Infrastructure program (SRWUIP) and associated schemes. For a subsidy or grant, the beneficiary implements specific infrastructure works and transfers a share of the assumed water savings in entitlements to the Australian Government. An additional sub-category of this mechanism – namely off-farm infrastructure modernisation – may also contribute to assumed water savings for the purposes of the aforementioned 450GL (Grafton and Wheeler, 2018).

Australia is more than halfway through implementation of the MDB Plan and, as at the start of 2019, has spent AUD$2.5 billion on buybacks and AUD$3.9 billion on modernising irrigation infrastructure (DAWR, 2019b). As of 31 March 2019, the Australian Government had acquired 1227GL of water entitlements through buybacks and strategic purchases (DAWR, 2019c), along with 695GL under the SRWUIP program. Based on these figures, approximately one-fifth of the estimated consumptive pool of water in the MDB had been transferred to environmental use under the Act and MDB Plan (see footnote 6 for detail).

Voluntary buybacks of water entitlements remains a controversial policy, with some rural communities asserting that they have diminished local spending, employment and public services (Bjornlund et al., 2011). These claims and persistent lobbying by industry irrigation groups drove the Australian Government to introduce (in 2015) a statutory limit of 1500GL on the volume of water that could be acquired via buybacks. The Australian Government also increased the budget allocated to irrigation infrastructure upgrades (Grafton and Wheeler, 2018; Haensch et al., 2019). This policy decision was made despite strong support for buybacks (Quiggin, 2008; Lee and Ancev, 2009; Grafton, 2010; Wittwer, 2011; Crase et al., 2013). Further, and despite claims to the contrary, many supported buybacks on the basis that they not only benefited the environment, but provided irrigators with far greater financial flexiblity than irrigation infrastructure grants (Loch et al., 2014; Wheeler et al., 2014b).

The two justifications for subsidising irrigation infrastructure for environmental water recovery purposes include: (1) *salinity*: reducing saline return flows into the rivers (water quality issue) (Wang et al., 2018); and (2) *farm productivity*: increases farm productivity (DAWR, 2019a) and hence makes recovery more politically acceptable. The multiple arguments against irrigation infrastructure subsidies include: (1) *cost*: subsidies cost at least three times more per dollars per megalitre (ML or million litres) of water acquired for the environment than buyback (Grafton and Wheeler, 2018; DAWR, 2019b), partly because of the increased transaction costs of subsidy programs; (2) *governance*: the program has been plagued by a lack of transparency, with some schemes subject to corruption charges (Victorian Ombudsman, 2011); (3) *return flows*: reduces seepage into groundwater and flows to streams and rivers (water quantity issue) (Williams and Grafton, 2020); (4) *rebound effect*: increases the area of land under irrigation or the area of land growing crops, potentially increasing water extractions (Adamson and Loch, 2018); (5) *utilisation*: increases utilisation of water entitlements and allocations (Wheeler et al., 2014a); (6) *substitution*: groundwater substituted for surface-water (Wheeler et al., 2014a); (7) *equity*: benefits are not evenly spread, with large corporate entities having a much higher probability of securing irrigation subsidies (21 times) over family farms[2]; (8) *floodplain harvesting*: the program funds new dams that can increase floodplain harvesting and divert water that may have been returned to streams and rivers (ABC Four Corners, 2019; Slattery et al., 2019); and (9) *resilience*: encourages substitution to permanent crops, increasing both electricity costs and demand for water during drought (Adamson and Loch, 2018; Wheeler et al., 2018) and reduces community resilience.[3]

Our purpose was to investigate whether irrigators who received an infrastructure subsidy in the MDB sometime in the past decade extracted more water compared to those who had not received a grant, holding all other factors constant. A unique farm-level survey database of 2481 records was used to investigate the potential existence of the rebound effect of irrigation infrastructure subsidies. We also examined publicly available data on Basin-wide water extractions from two different sources: the Murray-Darling Basin Authority (MDBA) and the Australian Bureau of Statistics (ABS), to understand how the rebound effect and the return of water entitlements from consumptive to environmental use may be influencing total water extractions.

Our contribution may be summarised as the identification, through existing datasets, of the intended and unintended consequences of using

[1] LTAAY is the long-term annual average volume of water permitted to be taken for consumptive use under a water access entitlement. Currently, all LTAAY figures are calculated using the long-term diversion limit equivalent (LTDLE) factors, with these factors to be accredited in finalised state water resource plans. See Appendix A for further comment.

[2] 70% of subsidies for irrigation efficiency were directed to family farms in the MDB (Simson, 2019), hence corporates (representing 2% of Australian farms (ABARES (2019; 132)) received 30% of funding: meaning a corporate farm had a 21 times higher probability of receiving an irrigation grant than a family farm.

[3] However, at least one irrigation infrastructure scheme (e.g. SA River Murray Sustainability Program) allowed for other (non-irrigation infrastructure) farm activities to be subsidised that may increase a farm's resilience in the face of future water scarcity.

subsidies to increase irrigation efficiency. Our findings are relevant beyond Australia insofar as many countries are attempting to increase overall water availability by subsidising and/or modernising irrigation infrastructure (Perry et al., 2017) – albeit it must be kept in mind that the institutional, physical, environmental, social and cultural conditions strongly influence all countries' water management and governance challenges.

2. Water use efficiency and the rebound effect

Increased irrigation efficiency is often presented as an opportunity for large-scale water savings, particularly in the agricultural sector. Perry et al. (2017) defined water use as: (1) consumptive use (conversion of water into water vapour), comprising beneficial consumption (i.e. crop transpiration; evaporation from wetlands) and non-beneficial consumption (i.e. evaporation from water surfaces and wet soil; transpiration by weeds); (2) non-consumptive use (water that remains in liquid state), comprising recoverable flows (returning to a river or aquifer for potential reuse) and non-recoverable flows (flowing to the sea or other unviable sink); and (3) storage change. Field water efficiency is often defined as the ratio of the volume of all irrigation water beneficially used on a farmer's field to the total volume of irrigation water applied; while water productivity is the output (yield or revenue) produced to water used (either water extracted, applied or consumed) (Grafton et al., 2018).

Irrigation efficiency programs, widely promoted by many governments as a panacea to overallocation, can result in contrary outcomes or rebound effects (Loch and Adamson, 2015). The rebound effect, also known as the Jevons' Paradox (Gómez and Pérez-Blanco, 2014), was first identified by Jevons in 1866 in a seminal study of coal use during the Industrial Revolution (Perry et al., 2017), revealing that increases in the work or energy produced per tonne of coal had increased, not decreased, coal use. The rebound effect also occurs in energy use (Wei et al., 2019).

The rebound effect is the change (increase) in water extraction from increased irrigation efficiency and its presence is determined by the interaction of three different water demand effects: (1) a *technical shift* allowing water extractions to decrease from the adoption of modernised irrigation infrastructure which allow far greater control over water movements and use (decreasing water demand); (2) a possible *variable cost* increase from modernising infrastructure and subsequently increasing electricity costs from increased pumping – especially if upgrading from previous flood irrigation by gravity (decreasing water demand); and (3) a *water productivity/relative value* increase as modernised infrastructure allows water to be used in new ways and often means crop mix change, increased utilisation of under-utilised water assets (including groundwater) and increased irrigation land (therefore increasing water demand). The increase in water extraction (namely the existence of a rebound effect) from (3) can outweigh the decreases in water extraction from (1) and (2) when: water is scarce; there is little idle irrigation capacity; irrigation infrastructures are subsidised; and water and energy costs are low (Gómez and Pérez-Blanco, 2014; Perry et al., 2017). The next issue is whether a rebound effect occurs only within a farm-level, or at a Basin-wide level. Deficiencies in water measurement, water accounting and compliance processes undermine attempts to contain these increases within any designated limit on catchment and Basin-wide extractions. The following section investigates the extent to which Australia falls into being a completely capped system (hence where irrigation water extractions cannot increase at the Basin-wide level even if there is existence of a rebound effect at the farm-level); or whether there is evidence of significant deficiences in water measurement and accounting that signal the potential for increased extractions to occur.

2.1. The Murray-Darling Basin literature

There is considerable literature that has signalled issues with water measurement and accounting in the Basin. For example, two recent studies have estimated the reduction in return flows (not currently fully accounted for in reductions in water inflows in the Basin) associated with subsidisation of MDB irrigation infrastructure. As a result of subsidies, reductions in return flows have been estimated to be 16% (Wang et al., 2018) or 40-90% (Williams and Grafton, 2020) of the amount transferred to environmental entitlements, with each study using different assumptions. Other studies have highlighted the possibility of the rebound effect and return flows in the MDB (e.g. Young et al., 2002; Qureshi et al., 2011; Grafton, 2017), while Loch and Adamson (2015) suggested that subsidies for irrigation infrastructure could increase consumptive water extractions and irrigation area in the Northern MDB.

Schirmer (2017) found that MDB irrigators who received an infrastructure grant were significantly more likely to increase irrigated and farm areas; increase irrigation efficiency; purchase new land; intensify production; 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). Most grant recipients (65%) believed that on-farm irrigation upgrades positively impacted farm profitability, however no information was available on how their water extractions changed over time (Schirmer, 2017).

ABC Four Corners (2019) and Slattery et al. (2019) suggested at least 20-30 new private dams, some capturing overland flows that previously flowed into rivers, have been built in the MDB with irrigation infrastructure subsidies. This finding is important because the common assumption by the Australian Government, the MDBA, state governments and some water scientists is that through imposing a limit (Cap) on the volume of water extracted within the MDB, and requiring the trade of water entitlements to occur within that limit, irrigators cannot increase water extractions (e.g. see Simson, 2019; Vertessy et al., 2019). In fact, various legal and illegal factors likely allow increased water extractions to exceed the statutory limits of the Cap. Factors that can result in increased water extractions, even within the Cap, include: (1) theft of surface and groundwater resources; (2) inaccurate or missing measurements of surface and groundwater extractions (Matthews, 2017; MDBA, 2017; NSW Ombudsman, 2017; Holley et al., 2018, 2020; Nelson, 2019); (3) unlicensed extractions such as the interception of overland flows during floods (AAS, 2019) and unmeasured 'stock and domestic' uses of water (MDBA, 2019a); (4) legal exceptions, water accounting issues and uncertainties in compliance tools, processes and river modelling; and (5) the complexity of accounting for water in complex floodplains, with evaporation, return flow and groundwater interactions (Kingsford, 1999; Ren and Kingsford, 2011; MDBA, 2018; AAS, 2019; Holley et al. 2020).

Each of these issues are discussed in further detail in Appendix A, and here we highlight their potential effect on water extractions. In particular, while the scale of illegal water extractions is unknown, one potential very broad indicator may include the New South Wales (NSW) *General Purpose Water Accounting Reports*. For example, the report for the Border Rivers Catchment in the MDB in 2017–18 cited an 'unaccounted difference' of 71,651ML, representing 19% of total catchment inflows (Burrell et al., 2018) – this difference was in addition to accounted evaporative losses. Other large percentages of 'unaccounted difference' are also found in other MDB catchments, and although modelling differences are to be expected, it may also highlight other more serious issues. The MDBA has also estimated actual surface-water extractions at the Basin-scale under SDL accounting arrangements, which includes estimates on unreported forms of take (i.e. extraction from watercourses for activities not previously included under the Cap – including interception activities, basic rights[4] and floodplain harvesting). For the 2017-18 irrigation season, actual extractions under SDL accounting were estimated by the MDBA at 2755.5GL (namely 35% higher) than that reported under the Cap (MDBA, 2019a; p. 92).

More broadly, ongoing resource limitations, as well as fluctuations

[4] MDBA (2019a) describes take under basic rights as water used for stock and domestic purposes in the MDB as well as native title rights in some Basin states. While the licencing requirements vary across states, a licence or approval is generally required to construct a bore for stock and domestic purposes, but not always to extract water from it.

Fig. 1. Murray-Darling Basin in Australia. *Source*: Adapted from MDBA (2015).

in resources for prosecutions, illegal bores and the potential overuse of stock and domestic water, historically received minimal inspectoral attention (Matthews 2017; NSWO 2017).

3. Data and method

Two types of data (secondary and primary) were used to investigate changes in MDB water extractions. We first provide a snapshot of available secondary data (using two different sources: MDBA for information on surface-water extractions under two different water accounting frameworks and the ABS for a snapshot of on-farm water extractions) to understand how water extractions have changed in both the Southern and Northern Basins (see Fig. 1) since the return of water entitlements from consumptive to environmental use from 2007-08 onwards. The primary data source uses information from three large-scale and representative irrigator surveys in the Southern Basin.

3.1. Location

The MDB has 65% of Australia's irrigated land, incorporating the Murray (South) and Darling (North) regions (Fig. 1). The Southern Basin comprises irrigation districts in South Australia, NSW and Victoria; while the Northern Basin includes irrigation districts in northern NSW and southern Queensland. The percentage of agribusinesses that irrigated in 2017-18 in the Southern Basin was 38% and 13% in the Northern Basin (ABS, 2019). Irrigators in Southern Basin districts historically received water allocations, regulated by government, determined by security/reliability restrictions and other factors such as history of use, environmental conditions and upstream storage (Wheeler et al., 2014). Irrigators in the Northern Basin own less regulated water entitlements and more unregulated licences; pay less in water charges; have larger irrigated areas and more homogenous production (namely cotton); trade water less; and rely more on groundwater and off-river storages for water extractions (AAS, 2019; Wheeler and Garrick, 2020).

3.2. Secondary water extractions data

Our first source of secondary data came from the ABS (multiple years) water use survey of irrigated farms in the MDB. This survey tracked irrigation business numbers, volume of irrigation water extracted per hectare of irrigated land, land area irrigated and crops grown (ABS, multiple years). The second source was from the MDBA water audit monitoring, SDL accounting and transitional water take, and Cap register compliance data (MDBA, 2018, 2019a).

ABS water extraction data for the MDB is presented from 2005-06 onwards, until 2017-18. Water extractions per hectare for the Northern and Southern Basins (Table B2, Appendix B) were calculated from 2005-06 to 2017-18 (ABS, multiple years), however, information was not available on which farms received an infrastructure subsidy. Challenges with ABS data included: it represented only a sample of agricultural operations over a certain size; farm sizes varied over time during non-census years; water extraction data was self-reported; and the natural resource management region boundaries differed over time, making direct region comparisons difficult. Nevertheless, ABS data included an estimate of all water extractions on-farm (including groundwater, floodplain harvesting, irrigation and private diversions).

The MDBA provides two water accounting estimates of MDB surface-water extractions: (1) *Cap data*: water extraction data, available in consistent form from 1997-98 to 2017-18, where water extractions were referred to as 'diversions' (and Basin state compliance was reported in the annual Water Audit Monitoring reports by the MDBA); and (2) *SDL accounting*: 'annual actual take' (volume of water used for consumptive purposes from watercourse or land-surface diversions), currently available from 2012-13 to 2017-18 (e.g. includes Cap data plus additional proxy estimates of surface-water extractions not measured under the Cap). The SDL water accounting and compliance framework was introduced in the MDB Plan 2012, and came into force on 1 July 2019. It is expected that, once all water resource plans are accredited and enforced, the Murray–Darling Basin Ministerial Council will recommend ending compliance against the Cap. Actual extractions were the annual volume of water calculated under either the Cap or SDL rules, which was actually used for consumptive (meaning *not* for environmental) purposes in a Cap valley or SDL resource unit (held environmental water entitlements are excluded) (MDBA, 2019a).

MDBA data includes surface-water extractions for metropolitan Adelaide and country towns (MDBA, 2019a), but does not include groundwater. MDBA (2019a) reported groundwater extractions as increasing from 1223 GL in 2012–13 to 1627 GL in 2017–18 – the highest level of extraction of any year within transitional groundwater SDL accounts – however this is considered an underestimate, given inadequate measurement and monitoring (Holley et al., 2018; Nelson, 2019). Appendix A outlines how the transitional SDL data is still subject to considerable underestimation due to floodplain harvesting, other interceptions of stream-flows, lack of measurement of stock and domestic extractions (i.e. basic rights), and also illegal extractions. Although 18 individual region audits have been triggered since 1997-98 (MDBA 2019e), the only independent overall audit conducted on Cap water extraction data since 2010 was a review by Turner et al. (2019), which recommended the MDBA should consider establishing a quality assurance process, as well as periodic independent audits. They also outlined 23 areas of concern (Turner et al., 2019: see their Appendix G).

3.3. Primary on-farm survey data

The data used was from three representative University of Adelaide irrigator farm surveys (2010-11; 2011-12; and 2015-16). These collected information on irrigation water extractions and irrigation infrastructure grant subsidies in the Southern Basin (see Zuo et al., 2015 and Wheeler et al., 2018 for survey details). Our irrigator survey areas in the Southern Basin (Fig. 1) included the Murray and Murrumbidgee River regions (NSW), where mostly annual cotton and rice crops are grown; the Goulburn–Murray Irrigation District and Murray River regions (VIC) where mostly dairy and livestock production takes place; and the Riverland (SA) where mostly citrus, wine grapes, fruit and nuts are grown.

The irrigator surveys were representative, and had high response rates. For example, the 2015-16 telephone-based survey had an initial response rate of 51%, and a response rate of 73% including those who agreed to be surveyed at a later date. Total cross-sectional sample size across the three-year period was 2481 records, with 1830 records having information on previous year water extractions in 2010-11 and 2015-16. In addition, some farmers were surveyed in all three years, providing temporal data (see Appendix C). Not all questions were asked in the 2011-12 survey, a follow-up to the 2010-11 survey. Water extraction and farm income and production data was provided for the previous season. The surveys included information on irrigators who had received an infrastructure irrigation grant[5], plus information on their water entitlements (owned and traded), management behaviour, water extractions by source, and other socio-economic demographics and attitudes. The temporal dataset tracked water extraction for the same irrigators over the five-year period, allowing the drivers of water extractions to be investigated. Importantly, overall water extractions varied significantly year by year, depending on seasonal conditions and allocations (as illustrated in Fig. 2). Hence, our modelling attempted to control for as many of these seasonal conditions as possible, as well as controlling for individual farm and farmer characteristics. Table B1 in Appendix B provides descriptive statistics.

Treatment effect estimators (Imbens and Wooldridge, 2009) were used to determine the average treatment effect (ATE) of infrastructure grants on water extractions and rates (see Appendix C for additional methodological details). Estimating the causal influence of receiving a grant on water extractions from observational data was difficult due to potential for non-random selection of irrigators between treatment and control groups. To correct for this non-random selection, treatment effect estimators were therefore used to estimate the causal effect – namely the mean of the difference between Y_1 (water extraction for a grant recipient) and Y_0 (water extraction for the counterfactual) – the ATE (see Imbens and Wooldridge (2009) for further methodological details).

4. Results

4.1. Secondary data results: ABS agricultural water extraction, MDBA Cap and SDL water accounting data

Irrigation water extractions reflected climate variations; seasonal water allocations to water entitlements; and land, crop, farm and irrigation characteristics (Fig. 2; Wheeler et al., 2015). In times of extended drought, irrigated crop water needs increase but irrigators, typically, have access to less water from storages, as water allocations assigned to water entitlements decline, especially for lower reliability entitlements.

The two measures of MDBA water extraction data (SDL accounting of 'actual take' and Cap data 'diversion' as described in Section 3.2) were plotted along with ABS farm water extraction data (Fig. 2). The three extraction estimates followed very similar patterns, with ABS irrigation data (as expected) estimating lower overall water extractions compared to the MDBA's estimates for the total MDB. In the ABS data, MDB farm water extractions varied considerably, but only decreased marginally from 2005-06 to 2017-18; whereas the number of irrigated businesses fell over time, but with no consequent trend in falling irrigation area or water extractions per hectare (Table B3, Appendix B). The ABS data occasionally generated higher estimates of water extractions in the Northern Basin, compared to the MDBA data (Fig. B1, Appendix B). This was especially the case from 2012-13 onwards (notably after key changes to Northern Basin water-sharing plans were made (AAS, 2019)).

Most (64%) of the environmental water entitlements owned by the Australian Government were already held by the time the Basin Plan was implemented in 2012 (Fig. 2). Taking 2017-18 as our reference for a comparison against available MDBA and ABS data, the Australian Government then owned just over 1900GL of entitlements in LTAAY (e.g. representing around a 19% reduction in consumptive water extractions (MDBA, 2013, 2019a)).[6] Here, we compared water extractions

[5] Note, we did not collect specific information on what type of grant was received, and our database therefore includes other possible irrigation infrastructure subsidies other than Commonwealth subsidy programs.

[6] The difficulty in calculating the percentage of water entitlements that have been returned to the environment is choosing the inital year reference given a) changing estimates of MDB baseline diversion limits (BDL) given water resource plan changes; and b) whether to include urban/other interception water extraction data in the BDL (MDBA, 2013). For example, the MDB BDL at 22 November 2012 was 13,623GL, and was 13,578GL at 29 March 2019. The SDL at Basin Plan implementation was 10,873GL, while as at 29 March 2019 it was

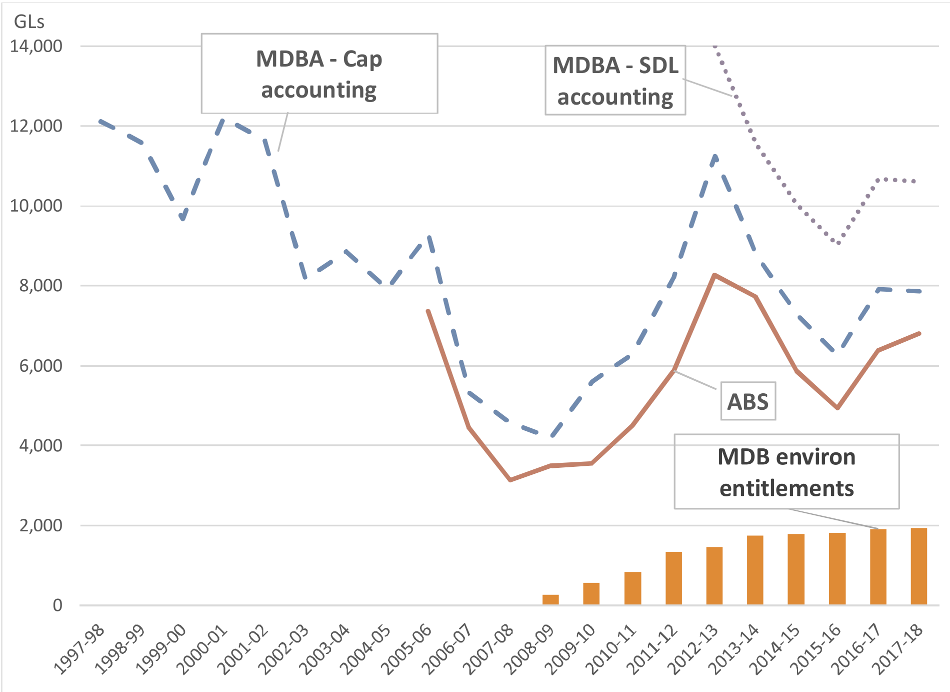

Fig. 2. Annual MDBA Cap and SDL total MDB water extractions, ABS irrigation water extractions and Commonwealth MDB environmental entititlement ownership data. *Sources*: MDBA (2019a) and ABS (multiple years). MDB environmental entitlements are the long-term average annual yield owned by the Australian Government, it does not represent the use of entitlements by environmental water holders.

Table 1
ATE of grants on water extraction rates and volumes.

Estimator	Water extraction rates (ML/Ha)		Water volumes (ML, natural log form)	
	Estimated ATE of grant	*p*-value	Estimated ATE of grant	*p*-value
OLS	0.344	0.024	0.249	0.004
Treatment-effects: regression adjustment	0.349	0.022	0.248	0.004
Treatment-effects: augmented inverse-probability weights	0.350	0.023	0.232	0.008
Treatment-effects: inverse-probability-weighted regression adjustment	0.351	0.023	0.232	0.008
Treatment-effects: nearest-neighbour matching	0.338	0.033	0.211	0.008
Treatment-effects: propensity-score matching	0.375	0.022	0.186	0.050

Note: Covariates used in the models listed in Appendix B, Table B1.

over two time-periods: 1997-98 to 2011-12 (i.e. the point at which the majority of environmental entitlements were owned by the Commonwealth) and 2012-13 to 2017-18 (both periods included wet and dry years), using MDBA Cap data, to evaluate whether there had also been a decline in water extractions of around one-fifth, as expected.

MDBA Cap diversion data indicated an average 2% reduction in MDB surface-water diversions (with decreases in the Southern Basin outweighing increases in the Northern Basin – see Appendix B) during these two time-periods, although it should be noted that, as measured, water extractions were currently within the Cap as at 2017-18 (MDBA, 2019a). Nevertheless, the measurement of water extractions is contested (as outlined in Appendix A), as is the process of current water accounting measurement. Slattery et al. (2019) observed that since 2010 there has been no independent audit to verify overall water extraction figures – with Turner et al. (2019) providing the first overall semi-audit. MDBA (2019e) reports that 18 individual audits on seven Basin valleys have been triggered since 1997-98, with a number of serious breaches found. ABS data (albeit over a shorter different time-period and therefore, again, it is difficult to compare the two) also does not show a 19% fall in irrigation water extraction. We suggest that further ongoing auditing of all water extraction data (under either Cap or SDL reporting) is required, along with a need for greater transparency regarding the water accounting method. However, future research should analyse water extraction by valley, controlling for water allocations, under different rainfall scenarios, carryover, and government intervention – especially given the prevalence of the Millennium drought during the above periods. Our broad findings identify important questions in relation to the change in irrigation water extractions.

4.2. *Primary Data: Southern MDB irrigators' water extractions and utilisation rates*

Our regression analysis, accounting for self-selection bias due to observed covariates, identified that grant recipients had statistically significant higher water extraction rates and volumes than non-recipients. Table 1 provides the estimated ATE of grants for water extraction volumes (ML) and rates (ML/ha) from five treatment effect estimators, using ordinary least squares (OLS) as a benchmark for the survey years in question. The ATE ranged from 0.34-0.38 ML/ha, depending on the estimator used, and all are statistically significant ($p < 0.05$). Therefore, this implies that overall, an irrigator who received a grant extracted 0.34-0.38 ML per hectare more than an irrigator who did not receive a grant, everything else being equal. Regarding water extraction volumes, the ATE ranged from 0.19-0.25

(*footnote continued*)
11,442GL (MDBA, 2019a). We followed MDBA (2013) and excluded urban/other interception data and used the original BDL 2012 estimate to estimate a 19% reduction in consumptive use as at 2017-18. Using the current water recovery amount of 2100GL as at 31 March 2019, this represents a 21% reduction in consumptive water extractions.

Table 2
Strategies of irrigators in the previous five years by grant status.

In the past five years, have you undertaken the following (% answering yes)	2010-11 Grant (n = 421)	Non-grant[a] (n = 525)	2015-16 Grant (n = 471)	Non-grant (n = 529)
Purchased water entitlement	30.1***	20.1***	24.6***	16.3***
Sold water entitlement	34	30.7	41.4***	27.2***
Increased irrigated land area	15.7	13.3	31.8***	23.1***
Decreased irrigated land area	44.7	45.5	20.2	22.9
Changed crop mix	53.2	51.6	57.7***	41.8***
Purchased farm-land	21.6	21.5	30.6***	22.9***
Sold farm-land	8.3	8.8	11.5	11.7
Improved irrigation efficiency	83.6***	62.7***	83.9***	68.1***
Purchased water allocations	N.A.	N.A.	58.2**	51.4**
Sold water allocations	N.A.	N.A.	50.3***	40.0***
Diversified production	53	52	40.6**	33.5**

Notes: ***, **, *Difference in strategy % between 'grant' and 'non-grant' is statistically significant at the 0.01, 0.05 and 0.1 significance level, respectively. N.A. indicates question not asked in survey.

Table 3
Planned strategies of irrigators in the next five years by grant status.

In the next five years, do you plan to undertake the following (% answering yes):	2010-11 Grant (n = 421)	No grant (n = 525)	2015-16 Grant (n = 471)	No grant (n = 529)
Purchase water entitlement	34.7	32.7	31.7	29.0
Sell water entitlement	37.8	36.8	14.3	15.2
Increase irrigated land	34.0	32.4	31.1	27.3
Decrease irrigated land	24.7	24.6	13.0*	16.9*
Change crop mix	59.4	61.9	54.7***	42.1***
Purchase farm-land	32.5	30.5	34.9**	28.5**
Sell farm-land	31.8	31.2	21.1	23.7

*Note:****, **, *Difference in strategy between 'grant' and 'non-grant' groups was statistically significant at the 0.01, 0.05 and 0.1 significance level, respectively.

($p < 0.05$). Given that water volumes are in natural log transformation, the ATE results imply that water extractions were 21-28% greater, on average, for a subsidy recipient, compared to a non-recipient (see Table 1) in our survey years.

We further analysed the water extractions data, using temporal data (i.e tracking the same recipients and non-recipients from the first survey in 2010-11 onwards, and obtained qualitatively similar results (Table C1, Appendix C)).

4.3. Primary data: Southern MDB irrigators' strategies

The full dataset was also investigated to understand whether the behaviour of irrigators who received grants differed from those that did not. In other words, what were grant recipients doing differently? First, we found significantly different (and rising) water utilisation rates (percentage of water received and used for irrigation[7]) by grant recipients as compared to non-recipients during 2009-10 (62% for recipients and 57% for non-recipients, t-stat = 1.99, df = 891, p-value < 0.05) and 2014-15 (85% for recipients and 79% for non-recipients, t-stat = 3.37, df = 919, p-value < 0.01) water seasons.

We also sought to understand the strategies that had been employed by irrigators on the farm. Grant recipients were statistically more likely to increase irrigated land, change crop mix, diversify production, and increase purchase of permanent and temporary water in the previous five years to the survey date (Tables 2 and 3). They were also more likely to have sold their water entitlements than non-recipients, though it is important to note that irrigators may have interpreted a 'sale of water entitlement' as the transfer of a share of their water entitlements to the Australian Government for the subsidy. The actual behaviour of irrigators on their farms over the past five years is provided in Table 2, and their planned strategies in the following five years from the survey date is provided in Table 3.

Irrigators planned for more farm management activities than they actually undertook (Table 3, see also Wheeler et al. (2013) for the difference between actual and stated/planned behaviour). When irrigators originally planned a farm management strategy (i.e. increase/decrease irrigated area, change crop mix, buy/sell water and buy/sell farm land), this did not necessarily eventuate due to multiple reasons (e.g. unplanned life/weather events, debt, health) (Wheeler et al., 2013). Additional analysis (Table C2 in Appendix C) indicated that receiving a 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. No information was available in the surveys on groundwater substitution, floodplain harvesting or illegal use.

In the 2015-16 survey 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 be spent on on-farm irrigation infrastructure by the Commonwealth (Australian Government) – while both groups disagreed on average that more money should be spent on buybacks (Table 4). But, 28% of irrigators overall did agree (or were neutral) that more money should be spent on water entitlement buybacks. In contrast, there was a significant difference in attitudes found between the two groups, in terms of their responses to whether irrigation infrastructure money had been

[7] Water utilisation rate represents the water extracted by irrigators as a percentage of the water allocated/received for a given year (which takes into account entitlement reliability). Mean water utilisation rate can be significantly influenced by observations receiving little water from entitlements owned but using much larger volumes through purchased water. Hence, water utilisation rates for irrigators purchasing temporary water were capped to 100%, as they used 100% of their own water entitlement use.

Table 4
Mean agreement score[1] 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: 1. 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.

wasteful and inefficient – irrigators who had not received any grants were more likely to agree it had been wasteful. Similarly, the two groups had significantly 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).

In summary, 51% of all irrigators surveyed agreed (or 75% agreed or were neutral) that expenditure on irrigation infrastructure had been wasteful and inefficient; while 21% of all irrigators agreed (or 54% agreed or were neutral) that infrastructure subsidy money should have been spent on rural education and health services instead.

5. Discussion

MDB irrigators have increasingly upgraded and modernized their irrigation infrastructure over the past few decades (Meyer, 2005; Crossman et al., 2010), using private funds and government subsidies. Irrigation infrastructure upgrades are an incremental adoption for irrigators (Wheeler et al., 2013) that allow farms to increase production value-add. This change has been supported by the Australian Government with subsidies for on-farm irrigation infrastructure intended to deliver a 'win' for the environment (more water returned to the rivers) and a 'win' for irrigated agriculture (increased value of agricultural production) (DAWR, 2019a). However, government infrastructure irrigation subsidies seem to have had a perverse water extraction outcome and actually increased water extractions by recipients. Subsidies reinforced, increased and hastened adoption of irrigation infrastructure and methods that altered cropping patterns – encouraging irrigation of permanent plantings, thereby increasing irrigated areas and on-farm water extractions. Increased water extractions were made possible because of increased utilisation of all types of water entitlements[8] and the interception of overland flows, creeks and floodways (Steinfield and Kingsford, 2013; Slattery et al., 2019). It is important to note that we are not suggesting that our identification of a rebound effect of 21-28% increase in water extractions has occurred at the Basin-level, as our effect was identified with existing irrigators only (and does not include those who have left the industry). What we are suggesting is that because of the rebound effect, and the considerable issues that have been documented with governance, transparency, floodplain harvesting, return flows, substitution and illegal use, there is a real question regarding whether the actual reductions in irrigation water extractions are the same as the amount of water returned to environmental use. The limitations of our analysis include the fact that we only had access to a couple of years of self-reported water extraction and infrastructure subsidy data for irrigators still irrigating at our survey points in time in the Southern Basin, and only a small sample size to conduct temporal data analysis. Information was missing/incomplete on groundwater substitution, illegal water use and floodplain harvesting. Further temporal data analysis on a larger sample in the Northern and Southern Basins would be warranted. In addition, further modelling on the drivers of regional water extractions at the community level would also be beneficial, along with the need for the ABS to geocode farm water extraction at consistent boundaries and farm definitions over time in the MDB.

To date, some AUD$4 billion has been spent on irrigation infrastructure subsidies, and billions more expenditure on irrigation infrastructure and supply projects is in the pipeline - yet there exists a far cheaper alternative to acquiring water for the environment: the direct purchase of water entitlements from willing sellers via open tenders. Thus, why is the idea of adopting (and subsidising) irrigation infrastructure to save water so entrenched, not only in Australia but worldwide? Perry (2018) explains it is because of entrenched views from eight groups: farmers (it increases their on-farm income and saves inputs); engineers (they benefit from (and like) money spent on modern infrastructure); equipment suppliers (they sell the infrastructure); politicians (they can claim they are saving the environment, solving food security and pleasing constituents); planners (believe water is released to other urban or environmental uses); experts (they can recommend large schemes and make optimistic predictions); donors (gives them something 'concrete' to fund and easier to implement than policy reform); and environmentalists (believe water is released for the environment). The current call to 'build dams' across Australia to solve its current water scarcity crisis emphasises the simplistic and short-term mindset that unfortunately the debate often falls into. Grafton and Williams (2019) also suggest that the irrigation sector and its peak body groups have been highly successful at influencing government decision-making.

Policy complacency also comes partly from a perception that current technical tools adequately track water extraction and flows in floodplain rivers. Governments primarily rely on mechanistic hydrological models to understand the water balance. Existing hydrological models used for auditing diversions can also perform poorly by overestimating low flows and under-estimating high flows on the floodplains – primarily because the data that they rely on originates from gauges in the river channel (Ren and Kingsford, 2011) and the significant complexity of natural hydrological and ecological interactions (Appendix A). This is of greatest concern for the Northern Basin, where irrigators are more likely to rely on both floodplain harvesting and groundwater compared to the Southern Basin (Wheeler and Garrick, 2020). Indeed, Turner et al. (2019) report a Cap data model confidence indicator of 'very low/low' for Northern Basin valleys. Thus, there is an urgent need to better understand historical and current water extraction (using satellite measurement and thermal imagery, along with metered data to track actual water extraction estimates), for both surface-water and groundwater sources across all types of entitlements; and to model changes in water extractions as the result of the transfer of water entitlements across river valleys, holding all other influences constant. These tools can support current modelling of flow regimes. There is also a need for greater harmonisation/standardisation of various water extraction data sources across river valleys.

As Holley et al. (2020) point out, resources for prosecutions, illegal

[8] Increased utilisation of water entitlements and increased modernised irrigation will generally reduce return flows and also reduce dam storage levels. In terms of growth in use impact on BDLs and SDLs, given the requirement that it has to be 'made good', the probablility is that low-priority users may suffer disproportionately in water allocations. Turner et al. (2019) note that the 'make good' provisions have not been tested and it is a future key area to monitor.

groundwater bores and stock and domestic water use have historically received minimal attention. Although we recognise that substantial improvements in compliance have recently occurred – such as an appointment of an interim Inspector General for the Basin (1 October 2019); the establishment of the NSW Natural Resources Access Regulator in 2017; and the increased use of some satellite monitoring of water extractions – this increased scrutiny and institutional change has only occurred after considerable external agitation and media exposure (e.g. ABC Four Corners, 2019). There is a review of the Basin Plan, including the level of water extractions, scheduled for 2026. While there is some support for this schedule (e.g. see Vertessy et al., 2019), our analysis suggests that a detailed and independent investigation is required much sooner as to why water extractions are not falling, as expected – and this without even considering the effects of climate change (Colloff and Pittock, 2019).

Basin-wide environmental improvements do not seem to have occurred and serious concerns exist about the impacts of 36 proposed supply projects – intended to provide the same environmental benefits with less water recovered for the environment, in terms of water entitlements owned by the Australian Government (PC, 2018; Colloff and Pittock, 2019). In our view, the evidence indicates that transformational rather than incremental change is required and the focus should be on reducing (and effectively monitoring, enforcing and regulating) water extraction. This transformational change will require a serious reconsideration of how Australia shares its water, with greater emphasis given to 'hands off' water and minimum flows in rivers (AAS, 2019; Young, 2019).

To date, water recovery for the environment, through irrigation infrastructure subsidisation, has been used as a de facto rural development strategy. Indeed, more than half of 1000 surveyed southern Basin irrigators in 2015-16 believed that this expenditure was wasteful and one-fifth of them believed it should have been spent on other services in the community. It is time that economic and social development in rural communities was taken more seriously (e.g. Wittwer and Dixon, 2013), and proper rural economic development (and structural and exit adjustment) strategies be developed (Wheeler et al., 2020). This approach needs to integrate with appropriate climate change, mental health, drought and natural resource management strategies, to help create more resilient rural communities.

6. Conclusion

Our research suggests that subsidised irrigation infrastructure upgrades in the MDB have not reduced water extractions or water consumption at a Basin-scale. While Australia has returned one-fifth of water entitlements in the MDB from extractive to environmental use, this does not seem to have translated into commensurate reductions in water extractions. This conclusion is based on comparisons of available public data by MDBA and ABS at the Basin-scale, and the identified existence of a rebound effect at the farm level. Indeed, our surveys of 2481 irrigators across a number of years indicate that those who received an irrigation infrastructure grant increased their water extraction volumes (by 21-28%) and rates, relative to other irrigators. This increase occurred because of crop mix changes and increased irrigated area, and was partly enabled through increased surface-water entitlement utilisation and water trade participation. Our research further suggests that current policy settings and compliance tools have failed to adequately constrain water extractions at a Basin-scale due to: (1) theft and poor enforcement in certain areas; (2) inaccurate and missing metering and measurements (including in relation to poorly regulated groundwater substitution); (3) growth in unlicensed diversions and on-farm storage capacity (notably in relation to floodplain harvesting); (4) legal exceptions and uncertainties in compliance tools and processes; and (5) complexities of natural hydrological and ecological interactions on floodplains, particularly in relation to evaporative processes and interactions between surface-water and groundwater systems. These problems are greatest in the less-regulated Northern Basin.

Our findings highlight the behavioural implications that arise from subsidising irrigation infrastructure. This requires reform that facilitates the use of technology to generate robust, fully audited water accounts, underpinned by sound measurement of all inflows; water consumption; recoverable return flows; and flows to sinks. Further, uncertainties should be appropriately accounted for in key water parameters such as: 1) improved compliance at the licence-holder and catchment scale (including close monitoring of, and response to, growth-in-use; independent and properly resourced enforcement agencies; and providing for third party enforcement of the *Water Act* and MDB Plan); and 2) using multiple lines of evidence for accounting and compliance. Finally, there needs to be a prioritisation of cost-effectiveness and environmental effectiveness for water recovery.

CRediT authorship contribution statement

S.A. Wheeler: Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Validation, Writing - original draft, Writing - review & editing. **E. Carmody:** Investigation, Writing - original draft, Writing - review & editing. **R.Q. Grafton:** Funding acquisition, Investigation, Writing - original draft, Writing - review & editing. **R.T. Kingsford:** Investigation, Writing - original draft, Writing - review & editing. **A. Zuo:** Data curation, Formal analysis, Investigation, Methodology, Validation, Visualization, Writing - original draft, Writing - review & editing.

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Appendix A

This appendix explores different factors driving increased water extractions (exceeding the MDB Cap and in many cases also the SDLs water accounting frameworks) in certain catchments across the MDB. These are: theft of both surface and groundwater resources and poor enforcement; inaccurate or absent measurement of water diversions (Matthews, 2017; MDBA, 2017; NSW Ombudsman, 2017; Holley et al., 2018, 2020; Nelson, 2019); unlicensed diversions (in particular floodplain harvesting, interception activities and basic rights) (MDBA, 2019a); growth in on-farm storage capacity in floodplain zones (AAS, 2019); legal exceptions, inadequate governance assessment of extractions from the main channels of rivers, and uncertainties in SDL compliance tools and processes (MDBA, 2018); and natural complexities of floodplain, evaporation and groundwater interactions (Ward, 1999; Parsons et al., 2008). Furthermore, Turner et al. (2019; see their Appendix G) outlined 23 areas of concern with SDL accounting. Of these areas, they considered the following to be of greatest concern (in terms of being rated high in materiality and impact): floodplain harvesting; unauthorised extraction; climate change; preparation for growth-in-use responses; and cultural flows.

Theft of both surface and groundwater resources and poor enforcement; inaccurate or absent measurement of water diversions

As water theft, poor enforcement and inaccurate or absent measurement of diversions are inextricably linked, we address these matters concurrently with an emphasis on two Basin States – NSW and Queensland. The choice of these States is based on evidence that they are "…bedevilled by patchy metering, the challenges of measuring unmetered take and the lack of real-time, accurate water accounts," affecting compliance and enforcement (MDBA, 2017; 14). It is also based on alleged and proven non-compliance with water laws in north-western NSW and the QLD MDB. By way of background, allegations of water theft first came to light via a series of media exposés in 2017 and 2018. This led to numerous formal inquiries, reviews and reports (inquiries) into alleged water theft, misconduct by bureaucrats and maladministration of water laws. These inquiries found, *inter alia*, inadequate enforcement of relevant laws by water agencies within these two States (Matthews, 2017; MDBA 2017; NSW Ombudsman, 2017).

NSW has taken steps to remedy this lacuna, establishing the Natural Resources Access Regulator (NRAR) by an Act of Parliament in November 2017 (Holley et al., 2020). The NRAR began operating in April 2018 and at the time of writing, has taken a significant number of enforcement actions (including nine prosecutions, 50 penalty infringement notices and 107 statutory notices) (NRAR Report, 2019). While this reflects well on the NRAR, it also highlights issues with compliance (including water theft, as reflected in a number of prosecutions). While the extent of water theft is unclear (due to lack of publicly available data), the *NSW General Purpose Water Accounting Reports* classify substantial volumes of water as 'unaccounted differences' in a number of catchments for several accounting years (e.g. Burrell et al., 2018). Although most of this difference is probably associated with modelling issues, some of these unexplained 'losses' could be attributed to theft – particularly in valleys with historically low and/or inaccurate metering (such as the Northern MDB) (MDBA, 2017; 137), requiring further investigation. Metering and telemetry are slated to be rolled out across NSW between now and 2023 (MDBA, 2019b) which, if successfully implemented and maintained, will arguably improve compliance with water laws (assuming the NRAR continues to be properly funded to enforce these laws).[9] However, metering and telemetry will not be universal, and in particular are not legally required for thousands of bores located across the State (MDBA, 2019b).

Queensland also has a poor enforcement record, as well as the large percentage of diversions (over two-thirds) that are unmetered or otherwise unmeasured (MDBA, 2017). This leaves considerable scope for unauthorised diversions, particularly given the large volumes of unmeasured water that are taken from floodplains, low levels of telemetry and the widespread practise of self-reading meters (MDBA, 2017). While water accounting reports for each catchment in Queensland are not publicly available, *Water Auditing Monitoring Reports,* which continued to 2011-12 (detailing compliance by each Basin State with the Cap), indicate that diversion figures for Queensland are the least accurate of any Basin State (MDBA, 2016). While the Queensland Government has announced plans to improve measurement of water diversions, it remains to be seen what percentage of extractions will be accurately measured and subject to telemetry, particularly given the large volumes of diversions attributable to floodplain harvesting. It also remains to be seen whether the government will meet its stated commitment to ensure all water entitlements in the Queensland MDB are accurately metered by 2025[10] (noting that not all legal extractions are linked to an entitlement). Further, the ability of floodplain earthworks to divert water is poorly identified and not measured. It is also unclear whether these proposed improvements will translate into enforcement of water laws and reductions in possible non-compliance.

Groundwater diversions merit particular attention because the monitoring of groundwater extraction in the MDB remains weak (Holley et al., 2018, 2020; Nelson, 2019). Of the approximately 270,000 bores in the Bureau of Meteorology's (BOM) groundwater dataset within the MDB surface-water area, 99,436 are in the NSW MDB and 34,548 in the QLD MDB. Of these, 13,180 are for irrigation, 22,444 are for stock and domestic use and 38,744 are used for unknown purposes (with the remainder attributed to a variety of purposes including monitoring, industrial use and water supply).[11] While this dataset does not exclude bores extracting water from aquifers in the Great Artesian Basin (which underlays the top third of the MDB), it is still reasonable to conclude that the number of bores extracting water for consumptive purposes from the NSW and QLD MDB groundwater sources is in the thousands. Relevantly, in the NSW MDB, many categories of bores are not required by law to be metered (for example those used for stock and domestic purposes) and all are exempt from the telemetry provisions that were introduced in 2018.[12] Similarly, in the Queensland MDB, diversions from most groundwater sources are not metered.[13] Further, a 2019 review by the MDBA into the self-meter read process, applicable to the Condamine Alluvium Groundwater Source, found that "the control environment is insufficient to provide adequate assurance that the measurement and reporting of water take in the Condamine Alluvium is reliable" (MDBA, 2019c; 2).

Uncertainty regarding extractions from groundwater sources is of particular concern during times of reduced surface-water availability/allocations (Nelson, 2019). According to the BOM, "[d]rought can cause groundwater use to increase as surfacewater becomes scarce and allocations are reduced…" The BOM noted: "…this appeared to occur in the upper and lower Namoi catchment, in 2017–18, with licenced extractions using 200,049 ML of groundwater from aquifers. This was almost the full allowable take of 203,800 ML. This number does not include unlicensed extractions for stock watering or domestic purposes." (BOM, 2019; 20). The final point is of broad significance: stock and domestic extractions in NSW and QLD are neither metered nor strictly controlled by a volumetric limit, adding an additional layer of uncertainty regarding the overall volume extracted from both surface and groundwater for this purpose (Nelson, 2019). The combination of probable reductions in surface-water availability in the MDB due to climate change (MDBA, 2019d) and the likely, subsequent shift to diversions – both legal and illegal – from poorly regulated groundwater sources (Nelson, 2019) may result in unaccounted SDLs exceedence. This hypothesis is linked to our analysis of the adequacy of SDL compliance tools and processes, detailed further below.

Unlicensed diversions (in particular floodplain harvesting, interception activities and basic rights) and growth in on-farm storage capacity in floodplain zones

Floodplain harvesting in the Northern MDB is inadequately regulated and measured, giving rise to a high level of uncertainty regarding volumes extracted in this manner (MDBA, 2017; AAS, 2019; Walker, 2019). In NSW, it remains an unlicenced form of water extraction which is estimated using hydrological modelling (NSW DPI, 2013). In QLD, it is either licenced or subject to a more general authorisation.[14] As in NSW, extractions are estimated using modelling, which is in part based on potentially out-of-date data regarding private off-river storage capacity (AAS, 2019) (noting that storage meters are only used in one sub-catchment) (Queensland Government, 2019). There is also evidence that laws regulating the building of levees and storages in both states have been poorly enforced, which combined with other factors has led to an overall growth in development in certain catchments (Kingsford 1999; Steinfield and Kingsford, 2013; AAS, 2019; Walker, 2019). The net effect is a low level of confidence that these

[9] *Water Management (General) Regulation 2018 (NSW)*, Part 10; s. 231(1); Schedule 8(6); s. 231(1)(c)(d).
[10] Murray-Darling Compliance Compact (2018); 6. https://www.mdba.gov.au/sites/default/files/Basin-Compliance-Compact-180702-D18-31184.pdf.
[11] Data provided by the Bureau of Meteorology (by email), 16 September 2019.
[12] Water Management (General) Regulation 2018 (NSW), s. 231(1)(d); Schedule 8(6)(2)(b).
[13] Water Act 2000 (QLD); Water Regulation 2016 (QLD).
[14] Water Act 2000 (QLD), s. 101.

extractions are in all instances capable of being contained by statutory limits or compliant with the Cap.

Both NSW and Queensland intend to improve the management and measurement of floodplain harvesting (NSW DPI, 2013; Queensland Government, 2019; 23). Specifically, NSW is proposing to licence floodplain harvesting in five Northern Basin catchments by 2021,[15] with hydrological modelling being used to determine the volume attributed to each licence. However, an independent review of the modeling found that "…there is a lack of transparency in the steps undertaken to develop the numerical models used in the implementation process, largely because of a lack of coherent, complete and up to date documentation outlining the methodologies, calibration, verification and assessment of scenarios" (Weber and Claydon, 2019; 5). Thus, the overall intent of the policy, which is to reduce floodplain harvesting to an historic level of extractions (most commonly that which was diverted in 1999-2000) (AAS, 2019), is potentially being undermined by technical and methodological flaws. Uncertainty regarding historic and current levels of floodplain harvesting is compounded by a lack of clarity regarding the fate of levees capable of diverting water beyond licenced volumes. That is, it is unclear whether all such structures will be permanently decommissioned and, if not, how this will be managed in light of volumetric limits. In Queensland, the government has made a very general commitment to improve measurement of these diversions[16], however details have not yet been made available. In summary, important questions remain regarding volumes attributable to floodplain harvesting, and whether or not they can be contained within SDLs.

Legal exceptions, inaccurate data, water accounting and inadequate governance arrangements

SDL compliance provisions and processes merit particular attention. First, the Basin Plan allows for the cumulative exceedance of the permitted take since 30 June 2019 – to reach a trigger equal to 20% of the long-term average annual volume of the SDL for a catchment, before a Basin State is required to provide a "reasonable excuse" for the exceedance. Where a reasonable excuse is provided, the State will be deemed compliant. Further, where the exceedence is found to be beyond the Basin State's control, the excess will be removed from the accounting record[17]. In other words, non-compliance will only occur when an exceedence in excess of the 20% trigger is not accompanied by a "reasonable excuse" (which is not defined in the Water Act or Plan, although policy guidance regarding this matter is available (MDBA, 2018)). In such instances, the MDBA is entitled to take enforcement action under Part 8 of the *Water Act 2007* (seeking, for example, an injunction preventing an ongoing breach of the SDL for a particular water resource plan area). However, it has indicated that it would seek to avoid the courts to the extent possible, relying instead on diplomatic pathways (including a range of possible management actions) and the use of its auditing functions[18]. Further, as the *Water Act* does not include open standing provisions, third parties (including those directly affected by water management decisions) are not entitled to enforce the Act or Plan through the courts.

It is understood that the 20% "exception", previously in place under the MDB Cap, was included in the Basin Plan to reduce the probability of a catchment being declared in breach of annual permitted extractions on the basis of inaccurate hydrological modelling. Further, and according to the MDBA (2018; 4), "…regardless of whether a Basin state is determined to be non-compliant or compliant with a reasonable excuse, if an excess growth-in-use is identified, the Basin state will be required to 'make good' and reduce actual take to the SDL". However, this is a policy position rather than an explicitly stipulated legal requirement[19] and one that depends on successful identification of growth-in-use, which can be "difficult to quantify" (Wentworth Group of Concerned Scientists, 2017; 58) – particularly given the issues with measurement of water and associated hydrological modelling in certain parts of the Basin, as discussed below. It also depends on the cooperation of any affected Basin States to reduce allocations. Given the frequent and well-documented threats by the current NSW Government to "walk away" from the Basin Plan if its demands to maintain or increase the volume of water available for consumptive use are not met (NSW DPIE, 2019), this cannot be assumed.

Second, the method for determining compliance with SDLs (via an assessment of annual actual take and annual permitted take), set out in each water resource plan[20], may be based on modelling that draws upon inaccurate datasets. For example, approximately 20% of extractions in the MDB are unmeasured (MDBA, 2018) – thereby rendering assumptions regarding these diversions inherently unreliable, complicating assessment of SDL compliance. Indeed, the MDBA has identified "issues with the accuracy of hydrologic models used for SDL compliance", with confidence levels (which reflect error rate) for models for Northern Basin catchments rated as "low" or "very low" (Turner et al., 2019). This general conclusion can be usefully supported by a more specific case study. To that end, we have analysed the Water Accounting Methods Report for the Condamine-Balonne Water Resource Plan (Queensland Government, 2019), relevant only to the Queensland section of this catchment, and identified major factors capable of undermining the accuracy of SDL compliance assessment for that water resource plan area. These factors are evidenced by the manner in which the long-term annual average extractions across different categories in the Condamine-Balonne are estimated. In the first instance, 398.07GL out of a total of 870.6GL of take is not measured. Second, few water allocations are measured with meters that comply with Australian Standard 4747 (0% of supplemented water allocations (which total 106.97GL) and 1% of unsupplemented water allocations (which total 326.99GL)) (Queensland Government, 2019; 18). Third, estimation of overland flow-extractions is modelled, with this being partly based on survey data regarding the capacity of overland flow storages that is approximately 20 years old (Queensland Government, 2019; 24). While a moratorium was imposed on the construction of new storage works in the Condamine-Balonne in 2000, compliance cannot be assumed over the last two decades. In summary, data regarding at least half of all known extractions in the area is likely to be more or less unreliable due to absent or inaccurate measurement. This is significant insofar as the calculation of annual actual extractions for the three aforementioned categories is based on the same method as that which was used to determine long-term annual average take (although we do note that additional lines of evidence are used to determine annual actual take for overland flow diversions) (Queensland Government, 2019; 44). There is, therefore, an extremely high probability that estimations of actual extractions for these categories are not accurate. We can also conclude that the modelling used to assess SDL compliance (annual permitted take) is in part based on data that is either unmeasured or inaccurately measured. Hence, there is reason to question the accuracy of SDL compliance processes for this water resource plan area, undermining compliance with the applicable SDL.

Water accounting issues also demand attention. Given that water entitlements across the MDB comprise varying securities and rights, they are often converted to LTAAY figures using LTDLE or cap factors. There are currently a number of LTDLE versions, namely 1) a version used in the Living

[15] https://www.industry.nsw.gov.au/water/plans-programs/healthy-floodplains-project/harvesting (accessed 25.09.19).

[16] https://www.dnrme.qld.gov.au/land-water/initiatives/rural-water-management/projects/measurement-overland-flow (accessed 25.09.19).

[17] Basin Plan, s. 6.11(5), s. 6.12; 6.12C.

[18] Basin Plan, s. 13.10, and notes under s. 6.12(5) and s. 6.12C(5). See also the non-compliance 'escalation pathway' set out in: MDBA (2018b; 13).

[19] Under s. 71(1)(h) of the Water Act 2007 (Cth), the MDBA can require that a Basin state provide it with information regarding the actions that it proposes to take to ensure that SDLs are complied with in the future. However, this provision is discretionary.

[20] Basin Plan, ss. 10.10; 10.15.

Murray, which was subsequently applied in the *Water for the Future* program from 2007-08 onwards; 2) a version used by the MDBA to develop the baseline diversion limits in the Basin Plan, first applied by the Commonwealth government in 2011 (and which had the impact of reducing the amount of water recovered, causing the Ministerial Council to order that the Authority revert to the original LTDLEs) (Slattery and Campbell 2018); and 3) a current version being developed for the purposes of water resource plans (PC, 2018; AAS, 2019). Shifting between LTDLE versions is significant insofar as it changes the LTAAY of water entitlements , therefore potentially increasing or decreasing the need for water recovery. The PC (2018) provides an n actual example for the changes in water accounting in the Gwydir. Specifically, changes in cap factors for supplementary and general security licences mean that the current estimate of CEWH holdings in that valley have increased from a LTAAY holding of 40,623GL to 47,932GL (without buying or selling any water). AAS (2019) has suggested that the change in cap factors in the latest version of the LTDLE was considerably different to previous estimates of LTAAY by the MDBA; it has further suggested that this version requires urgent reassessment as it could be used to reclassify some resource units as over-recovered against relevant SDLs.

Finally, current technical tools such as mechanistic hydrological models, focusing primarily on the main channels of rivers, do not adequately track water extraction and flows in floodplain rivers. Existing hydrological models used for auditing diversions can over-estimate low flows and under-estimate high flows on the floodplains as the data they rely on is primarily derived from gauges in the river channel (Ren and Kingsford, 2011). Modelling concerns were also raised in the SA MDB Royal Commission. For example, Close (2018) noted that on-farm storage in the Barwon-Darling increased by 50% between 1993-94 and 2000, with the region in breach of the Cap only three years after its introduction. However, the 2013 model that was developed by Bewsher (2013) for the MDBA under the Northern Basin review found that the Barwon-Darling was no longer in breach. This finding was not accompanied by any explanation, which was perplexing given the aforementioned increases in development. Despite experts recommending that the model not be accredited for Cap compliance purposes, it was used as part of the Northern Basin Review (Close, 2018).

In summary, there is considerable evidence to suggest that, where water extractions are not measured or are inaccurately measured (which is particularly relevant in the Northern Basin), SDL compliance tools may underestimate extractions and in turn fail to properly identify breaches of the Cap or the SDL under all forms of water accounting.

Natural complexities of floodplains, evaporation processes and groundwater systems

Finally, there is considerable natural complexity in the hydrological and ecological behaviour of floodplains, including in relation to evaporation and surface-water - groundwater interactions (Ward, 1999; Parsons et al., 2008). For example, there is evidence that in the lower valleys of the MDB, groundwater bores that are located next to rivers may at times allow for the extraction of surface flows due to the high level of connectivity between sources (Braaten and Gates, 2003). Further, the building of strutures on floodplains allows for diversion of floodplain flows into private storages (Steinfield and Kingsford, 2013; Steinfield et al., 2013). This complexity is not currently adequately captured in water accounting models or for compliance purposes, with relatively limited use of satellite imagery to track flows across the floodplains of the MDB. In summary, these issues generate further uncertainty regarding actual water extractions.

Appendix B

Table B2 illustrates our classification of ABS farm water extraction data into consistent areas to allow comparison of Northern and Southern Basins over time. Water extractions on farms is based on gross application rates and is not 'net use', but does not include diversion losses associated with transporting water to farms. From 2015-16 onwards, water extractions were estimated from farm busineseses undertaking agricultural activity above a minimum threshold of the estimated value of their agricultural operations (AUD$40,000). Agricultural census level data was available for 2005-06; 2010-11; and 2015-16. Before 2015-16, the ABS used an estimated agricultural value of AUD$5000 (ABS, 2019). The impact of this change was that from 2015-16 onwards, irrigation business numbers are estimated to be reduced by 22%, and water volumes by 4%. Thus, ABS water

Table B1
Descriptive statistics of variables in regression analysis, 2010-11 and 2015-16.

Variables	Mean	Standard deviation	Min.	Max.
Water extraction rate (ML/irrigation ha)	4.51	3.34	0.0001	19.77
Water extraction annual volume (ML)	616.81	1682.94	0.1	45000
Age (years)	57.10	11.28	24	90
Male dummy (1 = male; 0 = otherwise)	0.88	0.32	0	1
Low education level dummy (1 = education below year 10; 0 = otherwise)	0.17	0.37	0	1
Farming years	35.94	13.53	1	80
Whole farm plan dummy (1 = plan; 0 = otherwise)	0.74	0.44	0	1
Horticulture industry dummy (1 = horticulture; 0 = otherwise)	0.34	0.47	0	1
Broadacre industry dummy (1 = broadacre; 0 = otherwise)	0.30	0.46	0	1
Dairy/livestock industry dummy (1 = dairy; 0 = otherwise)	0.36	0.48	0	1
Farm size (ha)	677.42	2171.12	0.40	36826.43
Number of full-time equivalent employees	2.30	2.45	0	40
Irrigation area (ha)	198.83	435.41	0.40	8000
Productivity level[1]	2.72	1.31	1	5
Debt equity ratio	0.36	0.43	0	7
Attitude towards technology[2]	4.18	0.82	1	5
Net farm income at AUD 2015 prices ($)[3]	63679.61	71498.06	0[3]	250000
Total high security water entitlements (ML)	267.40	495.89	0	9604
Total general/low security water entitlements (ML)	532.49	1505.86	0	48000
Percentage of income from off-farm work (%)	31.87	35.49	0	100
Year dummy (1 = year 2015; 0 otherwise)	0.56	0.50	0	1

[1] Self-rating of productive output of farm over the last five years on a scale where 1 = strongly decreasing and 5 = strongly increasing.
[2] A Likert scale variable from 1 = strongly disagree; 2 = disagree; 3 = neutral; 4 = agree; 5 = strongly agree for the statement 'Knowing about technology is important to me'.
[3] All negative net farm income was coded as zero net farm income.

Table B2
Natural resource management (NRM) region classification in the Northern and Southern Basin.

NRM region	2005-06 to 2012-13	2013-14 to 2015-16	2016-17 to 2017-18	Note
Western	Northern	Northern	Northern	NRM boundary changed 2016-17
Border Rivers-Gwydir	Northern	Northern		NRM existed before 2016-17
Lachlan	Southern	Southern		
Murray	Southern	Southern		
Murrumbidgee	Southern	Southern		
Namoi	Northern	Northern		
Lower Murray-Darling	Northern	Data not reported		
Central Tablelands			Northern	NRM existed since 2016-17
Central West			Northern	
North West NSW			Northern	
Northern Tablelands			Northern	
Riverina			Southern	
South East NSW			Southern	

Table B3
MDB farm irrigation water extractions in ABS data.
Sources: ABS water extractions (multiple years).

Year	Agricultural businesses (no.)	Irrigation businesses	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
2006-07	59,864	17,063	1,101,000	4,458,279	4.05
2007-08	56,585	15,875	957,753	3,141,659	3.28
2008-09	54,096	15,476	929,074	3,492,409	3.76
2009-10	53,681	15,486	975,660	3,564,480	3.65
2010-11	54,023	15,794	1,194,253	4,518,369	3.78
2011-12	53,946	14,684	1,411,612	5,875,449	4.16
2012-13	51,203	13,361	1,597,454	8,283,439	5.19
2013-14	50,929	14,496	1,559,565	7,736,385	4.96
2014-15	49,096	14,587	1,366,738	5,868,785	4.29
2015-16	35,465	9216	1,238,106	4,938,381	3.99
2016-17	36,083	9196	1,347,592	6,355,072	4.72
2017-18	35,203	9496	1,460,054	6,797,678	4.66

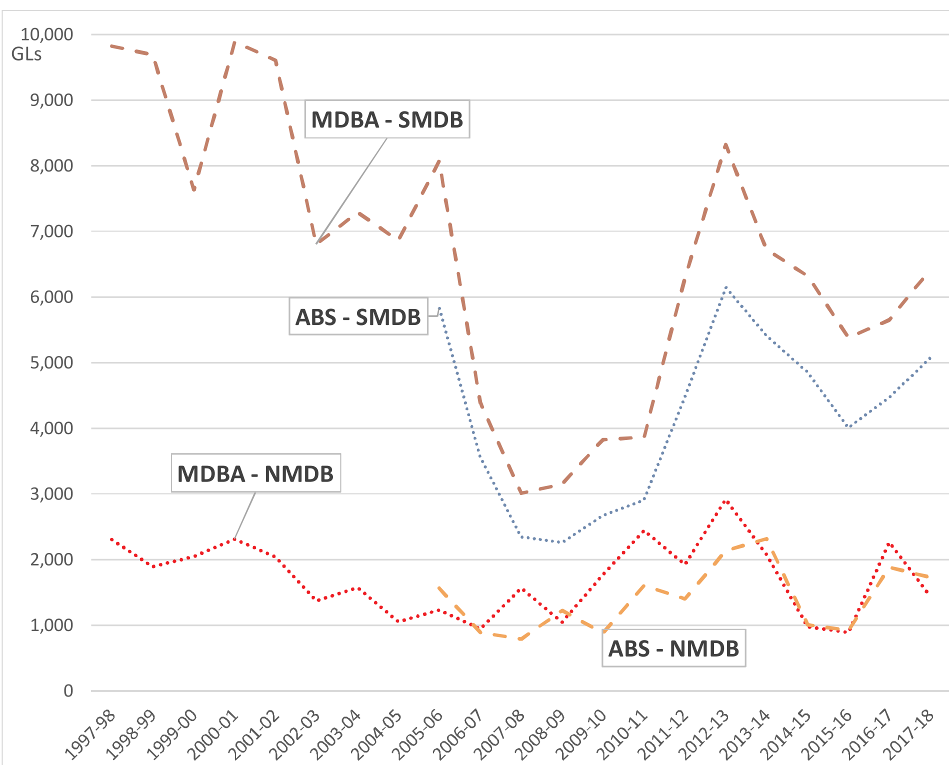

Fig. B1. MDBA Cap diversion data and ABS water extraction data for the Southern (SMDB) and Northern (NMDB) MDB. *Sources*: MDBA (2019a) and ABS (multiple years).

volumes and extraction rates from 2015-16 onwards should have been higher in Table B3 and Figs. B1 and Fig. 2 – if following the same method definitions as used prior to 2015-16.

Also, due to changes in Natural Resource Management boundaries, it was not possible to have Northern and Southern Basin areas align exactly with standard Basin boundaries used by the MDBA. In order to have Northern and Southern Basin boundaries derived from NRM areas as consistently as possible over the whole period shown here, the Lower Darling is included in the Northern Basin from 2005-06 to 2012-13 (although technically it should be in the Southern Basin). From 2013-14 to 2015-16, the Lower Darling was not reported by the ABS. From 2016-17 to 2017-18, the Western NRM area in the Northern Basin includes the Lower Darling. In addition, the Western, Central West and Central Tablelands in NSW are all in the Northern Basin, and since 2016-17, have included a part of Lachlan (Southern Basin).

Appendix C

C1. Regression methodology

Water extraction rate was used to illustrate the treatment effect estimator below, but this method was also applied to model total water extractions as well. The only difference was that total water extraction was tranformed by natural logarithm given its large dispersion (whereas the water extraction rate did not have such a large dispersion). Rescaling by taking a natural logrithm resulted in a difference of water extraction in percentge terms between grant recipients and non-recipients. Consider an irrigator that did not receive a grant has a water extraction rate Y_0. Y_1 is defined as the potential water extraction rate or counterfactual for the same irrigator if he/she received a grant. For an irrigator that did receive a grant, Y_1 is observed, so Y_0 would be the counterfactual outcome for the same irrigator. A missing-data problem arises since both Y_1 and Y_0 cannot be observed simultaneously for the same irrigator. Treatment-effect methods can account for this and estimate the ATE – the mean of the difference (Y_1-Y_0). Given receiving a grant is not likely to be random, recipient status could be related to covariates that also affect water extraction (i.e. a potential self-selection bias). Hence, the difference in sample means does not estimate the true average treatment effect. The treatment effects estimator specifies a number of covariates so that any remaining influences on the treatment are not related to the potential outcome, namely the water extraction rate.

Assume x_i is a vector of observed covariates that affect the outcome (water extraction rate), and w_i is a vector of covariates that affect the treatment assignment – receiving a grant (x_i and w_i may have elements in common) – the functional forms for Y_0 and Y_1 are:

$$Y_0 = x'\beta_0 + \varepsilon_0$$

$$Y_1 = x'\beta_1 + \varepsilon_1$$

where β_0 and β_1 are coefficients to be estimated and ε_0 and ε_1 are error terms that are not related to x_i or w_i. The treatment assignment is:

$$t = \begin{cases} 1 & if\ w'\gamma + \omega > 0 \\ 0 & otherwise \end{cases}$$

γ is a coefficient vector and ω is an error term that is not related to either x_i or w_i. Estimates of these coefficient vectors can be used to estimate the ATE = E(Y_1-Y_0). There are several different estimators to accomplish this, including regression adjustment (RA), augmented inverse-probability weights (AIPW), inverse-probability-weighted regression adjustment (IPWRA), propensity-score matching (PSM) and nearest-neighbour matching (NNM) (Imbens and Wooldridge, 2009). It should be noted that there is no definitive way to select one estimator over the others. However, the AIPW and IPWRA estimators have a double-robust property that if either the outcome model or the treatment model is correctly specified, the ATE can be consistently estimated. Similar ATE estimates across the various estimators is a necessary condition that the models are correctly specified (note Table 1 shows very similar ATE estimates).

We removed outliers where values > 20 ML/ha were excluded, and also controlled for sample selection bias. The choice of independent variables (e.g. x_i and w_i) in the model were driven by past research findings (e.g. Wheeler et al., 2014a, 2015), including age, gender, education, farming experience, agricultural industry, farm size, number of employed farm workers, irrigation area size, farm productivity condition, water right ownership, net farm income and off-farm work. Given the above, as well as missing covariate responses, the final sample size was 1604. Serious multicollinearity was tested for via correlations and variance inflation factors, with no serious issues identified. Robust standard errors were used to mitigate heteroskedasticity. In addition to accounting for the observed covariates, receiving a grant was tested for endogeneity to account for unobserved variables influencing both receiving a grant and water extraction rate/volumes, using an additional exclusion restriction – a Likert scale variable from strongly disagree to strongly agree for the statement 'Knowing about technology is important to me'. A test for endogeneity (chi-2 statistic = 1.98, df = 2, p-value = 0.37 for water extraction rate; and chi-2 statistic = 1.63, df = 2, p-value = 0.44 for water extraction volumes) indicated the treatment and outcome unobservables are uncorrelated – hence receiving a grant was not endogenous to the water extraction rate.

C2. Analysis using the same irrigators over time (panel dataset)

To track water extractions over our time, we followed the same grant recipients and non-recipients from the first survey in 2010-11, for their observations in 2011-12 (535 records) and 2015-16 (338 records) (Table C1). This tracks what happened to their water extraction over time (one and five years after our initial survey and by grant status as at 2010-11).[21]

Trends for all groups of irrigators' water extraction in the panel analysis rates mirrors the cross-sectional results reported in the body of the paper (namely recipients have higher absolute extraction rates and are increasing their extraction at a faster rate). But, the difference in water extraction rates *between* recipients and non-recipients is not significant for any particular industry (Table C1). Hence although grant recipients, as of 2010-11, had a relatively larger water extraction rate, it was not statistically significantly different than non-recipients, either one or five years later. This

[21] 59 of the non-recipient group originally surveyed in 2010-11 indicated they received a grant in 2015-16, implying they may have received their first infrastructure grant anytime between 2010-11 and 2015-16. Therefore, these 59 were excluded from the 2015-16 calculation. However, sensitivity tests indicated that results did not significantly differ when included.

Table C1
Mean water extraction (ML/ha) and associated standard error (SE) in 2011-12 and 2015-16 by industry and grant status (panel analysis).

		2011-12	2015-16	Change %
All irrigators		3.66	5.24	
	Recipients as of 2010-11	(SE = 0.18, n = 230)	(SE = 0.26, n = 168)	43.2
		3.57	4.85	
	Non-recipients as of 2010-11	(SE = 0.21, n = 225)	(SE = 0.37,n = 106)	35.9
	Two sample t-test of equal mean (p-value)	0.75	0.38	
Horticulture		4.57	7.00	
	Recipients as of 2010-11	(SE = 0.21, n = 95)	(SE = 0.38, n = 71)	53.2
		5.17	6.49	
	Non-recipients as of 2010-11	(SE = 0.39, n = 60)	(SE = 0.70, n = 29)	25.5
	Two sample t-test of equal mean (p-value)	0.15	0.50	
Broadacre		3.49	3.79	
	Recipients as of 2010-11	(SE = 0.41, n = 69)	(SE = 0.49, n = 35)	8.6
		4.06	3.38	
	Non-recipients as of 2010-11	(SE = 0.41, n = 82)	(SE = 0.66,n = 35)	-16.7
	Two sample t-test of equal mean (p-value)	0.33	0.61	
Dairy/ livestock		2.49	4.03	
	Recipients as of 2010-11	(SE = 0.40, n = 40)	(SE = 0.38,n = 62)	61.8
		2.26	4.94	
	Non-recipients as of 2010-11	(SE = 0.29, n = 51)	(SE = 0.52,n = 42)	118.6
	Two sample t-test of equal mean (p-value)	0.62	0.15	

Note: SE = standard error.

Table C2
Planned strategies of irrigators by grant status, between 2010-11 and 2015-16 (panel analysis).

Strategies	Grant	Non-grant	Null hypothesis: Grant = Non-grant
Increased irrigated area as planned	50% (n = 32)	32% (n = 83)	Reject (z-score = 1.79, p = 0.07)
Bought permanent water as planned	42% (n = 38)	21% (n = 70)	Reject (z-score = 2.31, p = 0.02)
Changed crop mix as planned	76% (n = 66)	55% (n = 155)	Reject (z-score = 2.93, p = 0.003)
Bought farm-land as planned	59% (n = 34)	39% (n = 77)	Reject (z-score = 1.85, p = 0.05)
Decreased irrigated area as planned	20% (n = 35)	34% (n = 56)	Cannot reject (z-score = -1.44, p = 0.15)
Sold permanent water as planned	47% (n = 36)	31% (n = 95)	Reject (z-score = 1.71, p = 0.09)
Sold farm-land as planned	17% (n = 35)	29% (n = 68)	Cannot reject (z-score = -1.33, p = 0.18)

finding is potentially driven by the smaller sample size in the panel dataset than the cross-sectional analysis. Nevertheless, the change in the water extraction rate between 2011-12 and 2015-16 was largest for the grant recipient group, and especially for horticulture and broadacre industries. For the dairy/livestock industry, the change was smaller for the recipient group.

Table C2 shows that using the panel dataset suggested that receiving a grant increased the likelihood of irrigators expanding their enterprises (i.e. increased irrigated area, buying farmland and water entitlements) as well as changing crop mix, compared to non-grant recipients.

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