Cost of uniform “cut”: Management of declining groundwater in the presence of environmental damages

Le Lan[a]*, MD Sayed Iftekhar[ab], James Fogarty[a], Steven Schilizzi[a]

[a]UWA School of Agriculture and Environment, The University of Western Australia, Crawley, WA 6009, Australia

[b]Griffith University

*E-mail address: lan.le@research.uwa.edu.au

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Abstract:

Globally, the agriculture sector is the largest user of groundwater, and reducing groundwater extraction by the agriculture sector is an active policy objective in many jurisdictions to manage a declining groundwater resource. Determination of the cost to agriculture in terms of lost gross margin due to implementing exogenously determined water extraction restrictions has been an active research area. In this paper, we contribute to the literature on groundwater management by developing a hydro-economic farm level optimization model that allows us to internalize the environmental externalities associated with groundwater extraction and compare with various levels of uniform proportional reduction in groundwater extraction. Our case studies are three sub-areas within Western Australia's most important groundwater system: the Gnangara Groundwater System. We find that when environmental externalities are considered, the reduction level of water extraction varied between 26% and 38% across the three sub-areas. Following the reduction, the total farm gross margin falls by 21% and the environmental damage falls by 98% relative to the current level of water extraction limits. We also find that to reach the same level of reduction in environmental damage, the uniform cut has to be between 40% and 50% and this results in a fall in farm gross margin by 29% to 39%. We present this contrasting result as evidence against using a policy of uniform proportional cuts to agriculture sector groundwater allocations

Key words: environmental externalities, groundwater-dependent ecosystems, groundwater management policy, uniform policy

JEL classifications: C60, Q15, Q25

Research categories: Environmental Economics and Policy, Resource/Energy Economics and Policy

1 Introduction

Groundwater for irrigation is an important agricultural input (Siebert et al., 2010). It also supports many ecosystems including wetlands, terrestrial vegetation, stygofauna, and caves. However, groundwater systems are declining due to climate change and unsustainable agricultural management practices (Famiglietti, 2014; Mulligan et al., 2014). This has produced far-reaching damages on groundwater-dependent ecosystems (GDEs), such as serious declines in biodiversity; mortality of vegetation species; river and wetland degradation; water quality contamination; and salinity intrusion in coastal aquifers (Castaño et al., 2018; Eamus & Froend, 2006; Kløve et al., 2011; Loáiciga et al., 2000; B. B. R. Murray et al., 2008). Given the huge environmental consequences of groundwater over-extraction, many countries are searching for approaches to compromise agricultural water extraction for ecological requirements.

In light of this, a common policy response has been for governments to impose restrictions on groundwater extraction for agriculture, with Brazil, Canada, Australia, China, New Zealand, Mexico, among many other countries, all imposing an extraction limit or 'caps' on extraction for agriculture (OECD, 2015). Often via reducing the 'cap', a fixed quantity or a fixed percentage of water extraction reduction (ie., 'cut') is uniformly implemented to achieve specified targets. Since this is a top-down regulatory approach, it could aim to secure a certain volume of groundwater for environmental objectives without the need of implementing any new institutional structures or legalisation (Tisdell, 2010). However, it has been suggested that this approach is not economically efficient because of the inflexibility in capturing the heterogeneity in economic and environmental values of water (Gao et al., 2013; Grafton & Ward, 2008). For example, Iftekhar and Fogarty (2017) found that large farms with high financial return would be disproportionately affected from a uniform cut policy. Further, responses to change in groundwater conditions of GDEs vary with size and ecosystem types (Esteban & Dinar, 2016; B. B. R. Murray et al., 2008); and the economic value of GDEs is also heterogeneous (B. R. Murray et al., 2006). Therefore imposing one-size-fit-all policies may result in unnecessarily high opportunity costs to farmers while not achieveing desired environmental benefits.

Fomulating spatially differentiated policies to protect ecosystems requires the internalization of the differences in environmental externalities associated with groundwater over-exploitation into decision making. There are several studies that have incorporated environmental externalities in their hydro-economic models to identify the appropriate policy instruments for preventing the damage of groundwater over-withdrawal to ecosystems (Esteban & Dinar, 2012; Kuwayama & Brozović, 2013). However, none of these studies compared the impact of internalization against various levels of uniform cut of groundwater extraction, which is often easier for the government agencies to set.

This paper builds upon the existing empirical studies on the cost-effectiveness of using uniform management policies to protect the GDEs, see for example: Gao et al. (2013); Kuwayama and Brozović (2013); Mulligan et al. (2014); Qureshi et al. (2006); and Tisdell (2010). A conclusion derived from these studies is that the performance of uniform policies vary with different contexts and circumstances. More empirical evidences on the performance of uniform policies - the most commonly used policies, in different contexts are therefore needed for policy makers to understand the relative performance of different policy instruments. This paper compares the performance of uniform proportional cut in water extraction for farmers and environmental externalities internalization approach in Australia. To date, no study in Australia has attempted to incorporate the spatially distinct damage cost of environmental externalities in the groundwater hydro-economic model to estimate the impact of different management approaches. Rather, most studies use a fixed quantity or percentage of water reduction, which is often preset via previous works as environmental targets and explore the costs imposed to agriculture sector by achieving that targeted reduction, e.g. (Gao et al., 2013; Tisdell, 2010).

In this paper, we develop a hydro-economic multi-year regional representative farm model. In the model, we internalize the cost of environmental externalities due to groundwater depletion. We expand the ecosystem damage function that is used by Esteban and Dinar (2012) by introducing heterogeneous spatial characteristics of three sub-areas (expressed in distance and hydraulic conductivity); and heterogeneous monetary values of GDEs (wetlands and terrestrial vegetation). This study focuses on the Wanneroo groundwater area, which is a part of Gnangara Groundwater System (GGS), the most important groundwater system in Western Australia (WA). Over-extraction, climate change and the expansion of pine plantation have caused the decline in water-table levels in GGS, leading to serious impacts on GDEs (DWER, 2017). In response to declining water table levels, the main water authority has determined that reduction in water allocations are required (DoW, 2015). However, both the extent of water allocation cuts and the method that will be used to reduce water allocations still need further investigation.

The paper is organized as follows. Section 2 describes the case study region. We then describe the methods used in this study in Section 3. Results from the case study are presented in Section 4, and finally Section 5 summarizes the results and discusses policy implications.

2 Case study area: Carabooda, Neerabup and Nowergup in the Gnangara Groundwater System

The GGS is the largest source of high-quality freshwater for metropolitan Western Australia (WA), as well as water for agriculture, and irrigation of public and private open spaces. The system provides water for almost 50% of Perth's public water supply and an extensive area of irrigated agriculture (DWER, 2017). The system also supports 69,000 ha of native woodlands and more than 2,000 ha of wetlands (Ranjan et al., 2009). The GGS consists of four main aquifers, with two shallow aquifers and two deep aquifers. The entire system covers approximately 2,200 squares kilometres along the Swan River, east to the Darling Scarp. The two shallow aquifers are the Gnangara Mound (unconfined superficial aquifer) and the Mirrabooka aquifer (semi-confined). The two deep aquifers are the Leederville and Yarragadee, and these two aquifers are defined as confined aquifers. Continuous reduction of rainfall and over extraction from the superficial aquifer has seen the water table fall, at an annual average rate of six cm per year, for over thirty-five years (Iftekhar & Fogarty, 2017). By 2030, if current extraction rates are maintained, it is predicted that the yield of groundwater might decline by more than one-third compared to the current level (Bennett & Gardner, 2014).

The increasing decline of groundwater table level in Gnangara has led to serious damage to the GDEs. According to Syme and Nancarrow (2011), more than 75% of the wetlands surrounding Perth have disappeared since the city's establishment. Groom et al. (2008) found that the fall in groundwater table level has been linked to a significant loss in vegetation species including Banksia woodlands, one of the native species in Western Australia. In addition, many other GDEs are currently under stress and require artificial water supply for maintenance (Ali et al., 2012; Environmental Protection Authority, 2007; Froend et al., 2004). In light of this, the WA Department of Water and Environmental Regulation (DWER) has issued the Gnangara Groundwater Allocation Plan, of which one of the three main objectives is to "limit the direct impacts of abstraction and use on GDEs and water quality" (DoW, 2009).

The DWER currently manages groundwater by dividing the whole region into eight groundwater areas and multiple sub-areas. Data analysed in this study relate to licenses held in the three sub-areas, namely Carabooda, Neerabup and Nowergup, in Wanneroo groundwater area (Figure 1). The three sub-areas represent one of the largest agricultural production in the Wanneroo region. According to the Australian Bureau of Statistic (ABS) year 2016-2017, the agriculture production in Wanneroo accounted for approximately 30% of the Western Australia production. Currently, there are nearly 1,040 employees working in the agriculture sector in Wanneroo (Taskforce, 2018). In addition, the area is home to numerous GDEs including Lake Nowergup: a wetland with high ecological value which is under risk of disappearing if groundwater levels continue dropping in current rate (Froend et al., 2004).

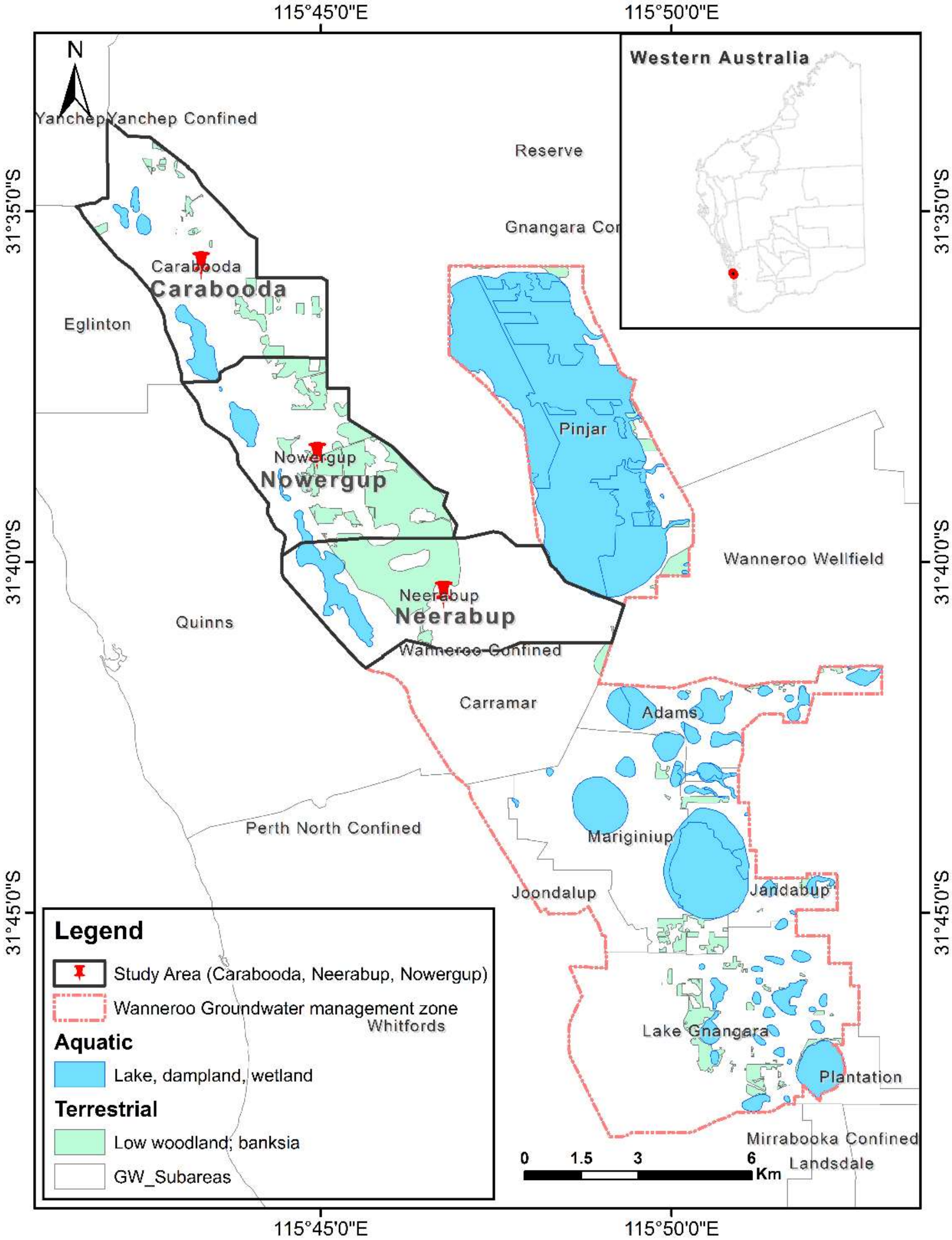

Figure 1 Map of Study Sites

The three sub-areas were identified by DWER as important and relevant for the purpose of the study. Each sub-area is characterized by different hydrological features including depths of water table, hydraulic conductivity, sub-aquifer area, and natural recharge. The groundwater allocation limits, available farming area, size of GDEs and the average distance from extraction points to GDEs also vary with each sub-area. As last documented in 2009, the total annual groundwater allocation in the three sub-areas is 11.05 GL, which is about 40% of total groundwater allocation in Wanneroo (DoW, 2009). In 2017, following the overall reduction in allocation, the total groundwater allocation in the three sub-areas is estimated at about 8 GL. There are about 60 groundwater licenses for horticulture purposes. The DWER is currently considering reductions in water allocations in the three sub-areas to sustain water requirements for the GDEs. However, both the extent of water allocation cuts and the method that will be used to reduce water extraction, has not yet been determined. Our study contributes to this policy gap.

3 Methodologies and data

We have adopted a simulation-optimization model to answer the main research questions. In agriculture economics and water management literature such an approach is very common (Koundouri, 2004). An empirically-based mixed-integer non-linear profits optimization model to investigate the aggregated impacts of reducing groundwater for agriculture on farmers' gross margin and ecosystem damages at sub-regional scale. These impacts are compared between two different approaches (Figure 2): (1) internalization of ecosystem damage and (2) uniform proportional cut in water allocation. Detailed descriptions of the different parts of the model are provided below.

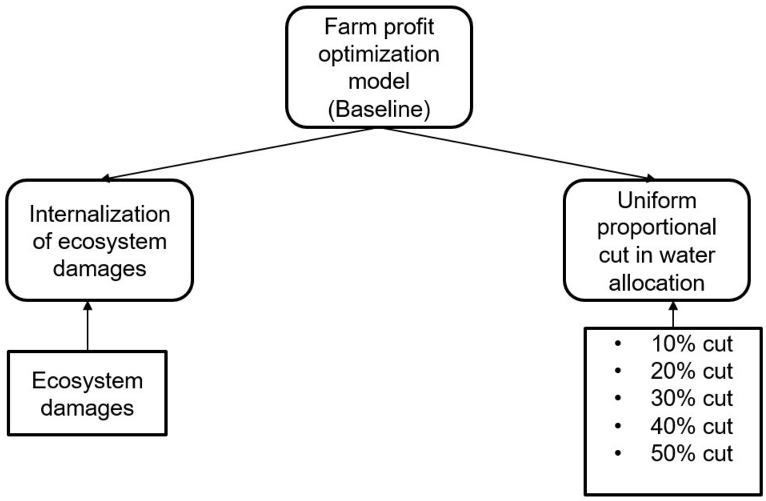

Figure 2 Modelling approaches

3.1 Farm profit optimization model

In the model, farm j can grow i vegetables, where i indexes summer vegetables and winter vegetables. It is assumed that within any given year a farm can produce at most one winter crop and one summer crop, and the area, in hectares, allocated to the cultivation of vegetable

i in year t by farm j is denoted a_{it}^j. Depending on water availability, a farm may not plant out all the available cropping areas. The maximum irrigable area available to farm j denoted AR^j does not vary through time, nor does the annual groundwater extraction limit, denoted gw^j which is set by the water authority. Each vegetable is characterized by a per hectare water requirement, expressed in megalitres (ML) and denoted by w_i. With the specified water application rate, the yield for vegetable i is y_i tonnes per hectare, and this product is sold at an average price p_i per tonne. The annual gross revenue of farm j at time t, denoted B_t^j, is then:

$$B_t^j = \sum_i p_i . y_i . a_{it}^j \quad (1)$$

The production cost function has three main components: water extraction costs, farm input costs, and irrigation costs. The water extraction cost is a summation of two terms $C_o.W_t^j$ and $C_1.(SL^j - H_t^j).W_t^j$. The first term is the product of pumping cost per ML, C_o and the total ML of water extracted to produce vegetables, W_t^j, which is calculated as $W_t^j = \sum_i a_{it}^j . w_i$. This is the initial cost of pumping and does not vary with the pumping height. The term C_1 is the marginal cost per ML pumped which is multiplied by the pumping height $(SL^j - H_t^j)$ and by the total water extraction, W_t^j. The pumping height is derived from the difference between the natural surface water level SL^j and the water table level H_t^j. The third component $\sum_i a_{it}^j.IC_i$ is the total cost of inputs per hectare IC_i including the cost of fertilizers, seedlings, and labour to produce vegetable i, which is multiplied by the total irrigated hectares for vegetables. The final component, irrigation cost, is represented by $\sum_i a_{it}^j.IR$, where IR is the annualized cost per hectare of the current irrigation technology used for vegetables production.

$$C_t^j = C_o.W_t^j + C_1.(SL^j - H_t^j).W_t^j + \sum_i a_{it}^j.IC_i + \sum_i a_{it}^j.IR \quad (2)$$

The annual gross margin of farm j at time t, denoted π_t^j, is defined as the difference between annual gross revenue and annual production costs:

$$\pi_t^j = B_t^j - C_t^j \qquad (3)$$

We assume the objective of farm j is to maximize the net present value of gross margin where future gross margin is discounted at the rate r , and the time horizon is ten years. To maximize gross margin, farm j decides on how much land area allocated to each vegetable.

There are four constraints to the optimization problem. The first two constraints are: (i) that the total area allocated to production by farm j at time t must be less than or equal to the total irrigated area available to farm j ; and (ii) that the total volume of water extracted by farm j for production at time t cannot exceed the farm's groundwater extraction limit. These two constraints can be written as $\sum_i a_{it}^j \leq AR^j$ and $W_t^j \leq gw^j$, respectively. The second two constraints are hydrology-based constraints. The first hydrology constraint is that the water table height at any given point in time cannot be greater than the surface level, and if the water table height at farm j in year t is denoted H_t^j , and the associated natural surface level at this point is denoted SL^j , this constraint can be written as: $H_t^j \leq SL^j$. The second hydrology constraint is the change in water table height, and the change in water table height at farm j is defined as: $\Delta H_t^j = H_{t+1}^j - H_t^j$. The change in the water table is in turn governed by the change in the stock of water in the aquifer around farm j, which depends on the volume of water flowing in and out of the land associated with farm j .

The total water flowing out includes the volume extracted by the farm W_t^j and the drawdown by all other licensed sectors in the area O_t^j . The first component of water flowing into farm j is the recharge due to irrigation inefficiency, which we denote λW_t^j , where $(0 < \lambda < 1)$. The second component is the natural recharge at farm j , denoted RC_t^j . The total flow of water onto farm j is then scaled by the product of the groundwater area that flows within the sub-area, here called sub-aquifer area AS^j and the specific yield of the aquifer S.

The third component represents the impact of groundwater extraction from neighbouring sub-areas on the drawdown of water table level at farm j , denoted x_t^j , $(x_t^j < 0)$. Economists define this impact as extraction externality. To describe x_t^j we follow Darcy's law and use a relatively simple parametrization that nevertheless captures the essential features of the system. In the model, this impact x_t^j is a function of the volume of water extracted at neighbouring farm j', $W^{j'}$, where $j' \neq j$, the physical distance between farm j' and j, $d^{j'j}$, and the hydraulic conductivity at farm j', denoted by $k^{j'}$, multiplied by a coefficient γ which represents the spatially weighted impact of water extraction of neighbouring farms on the drawdown of water table level at farm j. This coefficient is estimated based on Pfeiffer and Lin (2012), adjusted to hydrological parameters in the study sites. The impact x_t^j can be written as: $x_t^j = \gamma . \frac{\sum_{j',j'\neq j} k^{j'}.W^{j'}}{d^{j'j}}$

The objective function of farm j, $\hat{\pi}^j$, and associated constraints can now be written as:

$$\hat{\pi}^j = Max\ \pi^j = \sum_{t=1}^{T} \frac{1}{(1+r)^t} . \pi_t^j \quad (4)$$

subject to:

- Area constraint: $\sum_i a_{it}^j \leq AR^j$ (5)
- Water availability: $\sum_t W_t^j \leq gw^j$ (6)
- Change in water table level constraint:

  $\Delta H_t^j = H_{t+1}^j - H_t^j = \frac{-W_t^j - O_t^j + \lambda . W_t^j + RC_t^j}{AS^j . S} + x_t^j$ (7) and
- Maximum water table height constraint $H_t^j \leq SL^j$ (8)

This objective function does not consider the damage cost to GDEs that is associated with the groundwater extraction at farm j, so solving this problem provides information on the total value of agricultural production. This value can be interpreted as our estimate of the current value of agricultural production from the sub-area.

3.2 Ecosystem damages cost function

Several studies have formulated functions describing the economic value of ecosystem damage due to groundwater extraction. For example, Esteban and Dinar (2012) assumed a linear relationship between the cost of environmental damage and the reduction of groundwater table level height and internalized this cost in farm' economic optimization model to study the feasibility of cooperative management of groundwater in the Western La Mancha Aquifer and Tablas de Damiel wetland in Spain. Using similar approach, Ghadimi and Ketabchi (2019) investigated the possibility of cooperative management in groundwater in Iran. Kahil et al. (2015) assumed a water inflow-specific relation with three levels of ecosystem health status and tested the impact of drought on environmental and economic outcomes of an arid and semiarid basin in Southeastern Spain. Esteban and Dinar (2016) advanced the model of Esteban and Dinar (2012) by developing three different response functions of GDEs to the reduction of water table: linear negative slope, step-wise declining and backward folded. By internalizing the environmental damage costs derived from the three different functions, the authors indicate that the optimal extraction rates and the optimal water table level vary with the economic value of ecosystems as well as the ecosystem health function slopes. Another study that considered the cost of groundwater's environmental externalities in the hydro-economic model is of Pereau and Pryet (2018). In this study, the authors introduced a new environmental damages cost which is measured by the difference between the natural recharge and the natural discharge of the aquifer. The authors highlight that omitting the natural discharge may lead to excessive allocation of pumping quotas. Baniasadi et al. (2019) used replacement cost approaches to estimate the monetary value of the environmental externalities and examined appropriate policy recommendations for groundwater management in Iran. However, as mentioned above none of them evaluated the role of internalization against an uniform reduction of allocation.

To estimate the change in groundwater table at the ecosystem site, we assume that ecosystem is one of groundwater's user. In Australia, GDEs are classified into four classes: terrestrial vegetation; river systems; aquifer and caves systems; and wetlands (Hatton et al., 1997). This paper focuses on two types of GDEs: terrestrial vegetation and wetlands because these systems are the most widespread across the study areas (see Figure 1).

Following Darcy's law, the change in groundwater table level at the ecosystem site at time t, denoted ΔH_t^{ej} where e is the set of two GDEs: wetland and terrestrial vegetation, is impacted by the water table level at farm j at time t, denoted H_t^j and can be written as: $\Delta H_t^{ej} = k^j (H_t^j - H_o^{ej}) / d^{ej}$, with k^j is a hydraulic conductivity at farm j, H_o^{ej} denotes the initial water table level at the ecosystem site, and d^{ej} is a measure of the distance between farm j and the ecosystem site.

We assume a linear relationship between the damages cost to ecosystem and the change in water table level following Esteban and Dinar (2012). This damages cost, denoted ED_t^j is a function of change in water table level ΔH_t^{ej}, the economic value of ecosystem loss β^e which is measured in per hectare per meter of water depleted per year, and the per hectare area of wetlands and terrestrial vegetation that shares the groundwater use with farm j. The ecosystem damages cost can be written as:

$$ED_t^j = \sum_e \beta^e.\Delta H_t^{ej}.a^{ej}$$

We assume that when $\Delta H_t^{ej} \geq 0$ there is no damage to the ecosystem and thus the damages cost ED_t^j is equal to 0. Because the economic value of ecosystem loss due to the decline of groundwater table level β^e has not been officially reported in the study area as well as in Australia, we assign an implicit value of β^e based on two steps: (1) estimation and (2) validation.

In the first step, given that there is limited literature on estimating the economic value of physical changes in GDEs that links to the fall in groundwater table level, we must combine studies on physical behaviour of ecosystems under declining groundwater and non-market valuation studies of such ecosystems. Given that ecosystem responses to change in groundwater conditions vary depending on the ecosystem types (Esteban & Dinar, 2016; B. B. R. Murray et al., 2008), we compute different economic values for wetlands and terrestrial vegetation. For wetlands, we follow a hedonic price study of wetlands in Perth of Tapsuwan et al. (2009) where the authors estimated the value of wetlands around \$7 million per hectare. To link the fall in groundwater table level to wetland damage we made similar assumption to Ranjan et al. (2009) which stated that if the water table level in the wetland site falls by 1 meter, compared to the initial level at the start of the study period, and does not recover in 23 years, the site is considered to be dried out. A discount rate of 3% is used to calculate the annualized value. From these assumptions, the implied economic loss of one hectare of wetland per meter decline in the water table is \$ 425,697. For terrestrial vegetation, we combined the non-market valuation study of Roberson (2005) and the study on physical responses of vegetation in Western Australia to the fall in groundwater table level of Sommer and Froend (2011). In the study of Roberson (2005), the willingness to pay (WTP) for change from pine plantation to nature conservation in Gnangara Park is estimated between \$2.75 per person (lower bound) and $11.92 (upper bound) per year for every extra one percent of the land used for nature conservation in Gnangara Park Zone 1. Given that the total area of Gnangara park zone 1 is about 7,500 hectares, this one percent equals to about 75 hectares of land (Ranjan et al., 2009). Assuming that the number of visitors per year is fixed at 250,000 (Perriam et al., 2008), the implied value of maintaining vegetation in the region is between $9,167 and $39,733 per hectare. In the study of Sommer and Froend (2011), the authors reported that around 44% of the floristic change in the site could be associated with a decline of 5.4 m in groundwater table level. We then compute these information and set the economic value of terrestrial vegetation at $3,004 (lower bound) and $13,025 (upper bound) per meter of declining water per year. All of these monetary values of wetlands and terrestrial vegetation are then adjusted to 2017 prices using the Reserve Bank of Australia inflation calculator (Table 1).

Table 1 Economic value of ecosystem damages

GDEs	**Value ($ per hectares per 1 m decline of groundwater table level)[1]**	**References**
Wetlands	$506,847	Tapsuwan et al. (2009) Ranjan et al. (2009)
Terrestrial vegetation	Lower bound: $4,025 Upper bound: $17,452	Roberson (2005) Sommer and Froend (2011) Ranjan et al. (2009) Perriam et al. (2008)

[1] Adjusted to 2017 using inflation calculator of Reserve Bank of Australia (https://www.rba.gov.au/calculator/annualDecimal.html)

In the validation step (second step), we ran simulations using the value of β^e that was estimated in the first step. We then compare the ecosystem damages cost per kL of water extraction generated from our simulations to several sources including academic and non-academic such as reports, and policies as part of the validation process. The first source that we use for validation is a report of economic cost of groundwater (Deloitte Access Economics Pty Ltd, 2013). The average cost of groundwater for gardens and parks after adjusting to 2017, which we found the most relevant to the environmental cost among other economic costs, is estimated from $0.13 to $1.98/kl of water extracted. The second source is Ranjan et al. (2009), which provides an estimate of the augmentation cost of wetlands in Gnangara using an opportunity cost approach. The cost, after adjusting for inflation, is

\$3.05/kl. The third source is based on the Natural Resources Management Act 2004 published by The South Australian Government (2019). In the Act, a levy of \$4.31/KL of water allocated for the operation of tourist parks. We found that our simulation results is a bit low compared to the validation sources. This is not surprising due to lack of comprehensive evaluation studies capturing the wide range of the environmental values of groundwater in Gnangara. For example, the ecosystem damage cost can be influenced by the environmental externalities related to aquifers (e.g. a lower water table can reduce water quality). However, due to lack of data we could not incorporate this information. Therefore, after running multiple simulations, we decided to multiply β^e with a scaling factor σ to capture the variation in real-world observations. We found that a scaling factor σ uniformly distributed from 1 to 3 gave us an average environmental cost of \$1.3/KL of water, with almost 100% of our simulation results falling within the reported range of \$0.18/kl to about \$4.5/KL (Figure 3). On this basis, we are satisfied that the environmental cost values generated by parameters in our model are reasonable.

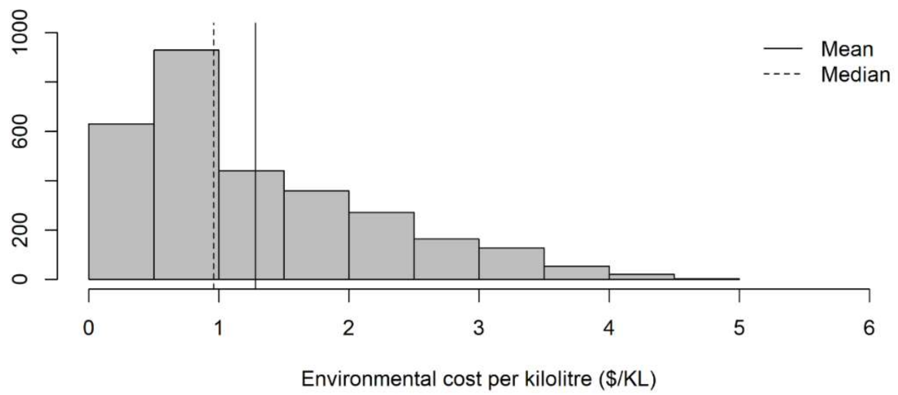

Figure 3 Distribution of environmental cost per kilolitre of groundwater extraction

3.3 Internalization of ecosystem damage

Environmental economic theory predicts that if a farm is forced to internalize the ecosystem damages associated with farm's production activities, a proper level of water is extracted to minimize the damages cost and the loss to farm profit.

To model the internalization approach, we incorporate the ecosystem damages cost ED_t^j to the farm profit optimization in equation (4). The objective function of farm j when internalizing ecosystem damage costs is denoted $\hat{\pi}_{ED}^j$, and can be written as:

$$\hat{\pi}_{ED}^{j} = Max\ \pi_{ED}^{j} = \sum_{t=1}^{T} \frac{1}{(1+r)^{t}}.\left(\pi_{t}^{j} - ED_{t}^{j}\right) \quad (12)$$

where the constraints are similar to previous constraints which are presented in equation (5), (6), (7), (8). By comparing the solution to this optimization problem to the solution for the no ecosystem damages cost internalization solution, we can derive the level of reduced water extraction and the associated impact on farm gross margin and ecosystem damages.

3.4 Uniform proportional 'cut' in water allocation

To model the 'cut' in groundwater allocation, we use the farm optimization model in equation (4) and impose fixed percentage reductions in the constraint of groundwater allocation limits in equation (5) for all three sub-areas. We assess five levels of cut, starting from 10% to 50%. The change in gross margin and ecosystem damages of each level in relation to the current allocation represents the economic and environmental impacts of using uniform cut policy mechanism to reduce groundwater extraction in agriculture. A comparison of impacts on farms gross margin and ecosystem damages between the uniform cut and the internalization approach is then discussed.

3.5 Data and Model calibration

Complete details for each model parameter, including the source of information used, are summarized in Table 2. We collected information on parameters from various sources including literature review, personal communication with relevant governmental agencies and organization, and formal governmental online portals. Specifically, parameters were estimated based on the literature review include: water requirements for vegetables and vegetables input costs and irrigation costs (Iftekhar and Fogarty, 2017); initial pumping cost and marginal pumping cost (Department of Agriculture and Food, 2008); return coefficient of irrigated water (Esteban and Dinar 2012); economic value of wetlands (Sommer and Froend, 2011; Tapsuwan et al. 2009); spatial externality coefficient (Pfeiffer and Lin 2012); and discount rate (Ranjan et al 2009). Context-specific hydrological parameters such as groundwater allocation, total farming area, sub-aquifer area, recharge rate, specific groundwater yield, and groundwater use of other sectors were collected from personal communication with the DWER. Vegetables price was retrieved from personal communication with VegetablesWA. Several online portal sources were used to collect parameters of vegetables yield (ABS, 2016-2017), initial water table level and natural surface level (DWER's groundwater online map); area of ecosystem sites and the average distance from farm to ecosystem site (Australian GDE's atlas map); and distance from the central point of each sub-area (Google map).

Depending on the availability of information, the min and max value of parameters were either generated as plus and minus 20% of the mean or based on other information as provided in the Table. For stochastic parameters, draws are from a uniform distribution.

To account for the heterogeneity in sub-areas and hydrological conditions, we treat each sub-area as a single production unit, where the total horticulture area in the sub-area is the sum of all horticulture farms within that sub-area and all associated groundwater allocation licenses. The distance between sub-areas $d^{j'j}$ is measured from the central point of each sub-area. Because there are many GDEs sites and many extraction bores within a sub-area, we measure the distance to GDEs in each sub-area, d^{ej}, as the average distance of extraction bores for horticulture within that area to the nearest wetlands or terrestrial vegetation. The area of ecosystem sites a^{ej} is the aggregated area of multiple ecosystem sites within a sub-area. From the individual water license data provided by DWER, we use only licenses with a water allocation used for horticulture in 2017. For multiple licenses with a similar name and amount of water and land allocation, we picked only one license as the representative license. The total area for horticulture is 581.4 hectares in the three sub-areas. The available area for farming, groundwater allocation limits, average distance to GDEs, and area of GDEs of each sub-area are reported in Table 2.

Due to declining rainfall, the natural recharge rate for the GGS has been falling over time. Base case climate scenario projections are that rainfall, and hence natural recharge in the southwest of Western Australia, will continue to decrease (Charles et al., 2010). Consistent with this scenario, in the model the recharge rate is projected to continue to fall at the annual average rate observed over the past decade (Figure 4).

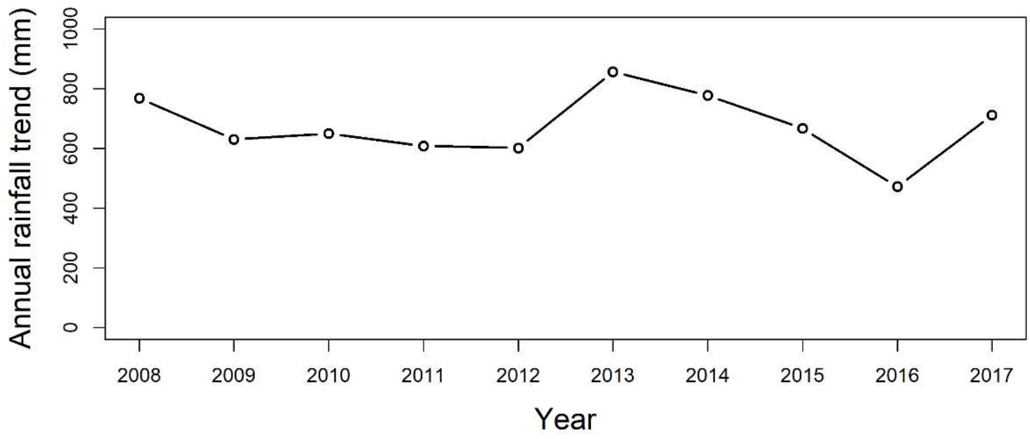

Figure 4 Rainfall trend in 10 years

(Data was retrieved from Bureau of Meteorology, http://www.bom.gov.au/)

The vegetables used in the model are tomatoes (water-intensive summer crop) and broccoli (less water-intensive winter crop). These are profitable vegetables that are widely grown in the region as a regular cropping rotation combination. The vegetable production costs parameters are adjusted for inflation. Yield and price parameters were updated in 2017 data.

The assumed irrigation technology in the model across all farms is drip and micro-irrigation (DM). This technology consumes the least amount of water among available technologies in the region but is not the only irrigation technology observed in the region. Sensitivity analysis using different irrigation technologies was conducted, and the results, which are presented in Appendix B, show that alternative assumptions about irrigation technology have little impact on the results, except for Furrow which is the least efficient technology.

The parameters related to the economic value of GDEs were derived from the literature as discussed previously in section 3.3.

For the base case, we use a real social discount rate of 3%. We use this discount rate as it allows us to compare our results to Ranjan et al. (2009), which is an earlier cost-benefit analysis of land and water uses in Gnangara. The sensitivity analysis results from the use of different social discount rates are presented in Appendix C, and these results show that alternative social discount rate assumptions have little impact on the main results and conclusions.

Table 2

Parameters in the Model

Parameters	Explanation	Unit	Value[1] (min-max)[2]	Reference
y_i	Yield of vegetable i	Ton per hectare	Tomatoes: 25.7 – 38.6 Broccoli: 15.2 – 22.8	ABS (2016-2017)
p_i	Price of vegetable i	AU$ per ton	Tomatoes: 1,700 – 2,500 Broccoli: 1,400 – 2,100	VegetablesWA. Per.Com.2018
w_i	Water requirement for vegetable i using DM irrigation technology	Megalitre per hectare	Tomatoes: 14.1 Broccoli: 4.5	Iftekhar and Fogarty (2017)
IC_i	Fixed cost of inputs for producing vegetable i	AU$ per hectare per year	Tomatoes: 3,600 – 15,000 Broccoli: 7,000– 28,900	Iftekhar and Fogarty (2017)
IR_i	Annualized capital cost of irrigation technology used for vegetable i	AU$ per hectare per year	Drip and Micro: 500– 700	Iftekhar and Fogarty (2017)
λ	Return coefficient of total water pumped for vegetable	None	0.15	Esteban and Dinar (2012)
C_o	Fixed initial cost of pumping water	$AU per Megalitre	80– 120	Department of Agriculture and Food (2008)
C_1	Marginal cost of pumping	AU$ per Megalitre per meter	Carabooda: 3 - 5 Neerabup: 7 - 10 Nowergup: 3 - 5	Current cost of pumping divided by the current average depth to water
AR^j	Total area available for farming of farm j[3]	Hectare	Carabooda: 384.2 Neerabup: 98.6 Nowergup: 98.6	DWER. Per.Com. 28/09/2018

Parameters	Explanation	Unit	Value[1] (min-max)[2]	Reference
RC_o^j	Initial natural recharge of groundwater at farm j (e.g. rainfall)	Gigalitre per year	Carabooda: 4.4 Neerabup: 3.2 Nowergup: 2.7	DWER. Per.Com. 28/09/2018
k^j	Hydraulic conductivity	Metre per day	Carabooda: 40 Neerabup: 22 Nowergup: 35	DWER. Per.Com. 28/09/2018
O^j	Groundwater uses of other sectors	Gigalitre per year	Carabooda: 1.24 Neerabup: 1.26 Nowergup: 0.7	Authors' estimation based on DoW (2009) and DWER. Per.Com. 28/09/2018
AS^j	Area of the sub-aquifers	Km^2	Carabooda: 18,200 Neerabup: 18,200 Nowergup: 17,000	DWER. Per.Com. 28/09/2018
S	Specific yield of the aquifer	None	0.2	DWER. Per.Com. 28/09/2018
SL^j	Natural surface level of groundwater at farm j	Metre	Carabooda: 37 Neerabup: 29.1 Nowergup: 39.3	DWER's groundwater online map accessed on 20/10/2018
H_o^j	Initial water table level at farm j	Metre	Carabooda: 13.6 Neerabup: 17.7 Nowergup: 17.3	DWER's groundwater online map accessed on 20/10/2018
γ	Spatially weighted impact coefficient[4]	None	0.000018	Author calculation based on Pfeiffer and Lin (2012)
β^e	Economic value of ecosystem damage or depletion	AU$ per metre per year per hectare	Wetland: 405,478 – 608,216 Bushland: 4,025 – 17,452	Author calculation based on Sommer and Froend (2011); Tapsuwan et al. (2009); Roberson (2005)
$d^{j'j}$	Distance between sub-areas[5]	Metre	Carabooda to Neerabup: 10,800	Google map

Parameters	Explanation	Unit	Value[1] (min-max)[2]	Reference
			Carabooda to Nowergup: 8,100 Nowergup to Neerabup 5,700	
d^{ej}	Distance from sub-aquifers to ecosystem sites[6]	Metre	Carabooda: 492 Neerabup: 614 Nowergup: 477	Australian groundwater-dependent ecosystems GDE atlas online map
a^{ej}	Total area of ecosystem sites within sub-area	Hectare	*Wetland* Carabooda: 150 Neerabup: 129 Nowergup: 149 *Terrestrial vegetation* Carabooda: 653 Neerabup: 150 Nowergup: 1,301	Australian groundwater dependent ecosystems GDE atlas online map
H_o^e	Water table level at ecosystem sites[7]	Metre	Carabooda: 13.6 Neerabup: 17.7 Nowergup: 17.3	Assumed to be similar to H_o^j
r	Discount rate	%	3	Ranjan et al. (2009)

Note. [1]Value of cost and price have been adjusted to 2017 using the Consumer Price Index available in the inflation calculator of Reserve Bank of Australia (https://www.rba.gov.au/calculator/annualDecimal.html). [2]Min and max range are generated by plus and minus 20% of the average values except for IC_i which is based on the min and max values reported in Iftekhar and Fogarty (2017). [3]In WA, an irrigated farm can grow two crops per year on the same land and 15% of the total land is left as fallow.[4] Result from Pfeiffer and Lin (2012) indicates that extracting 1,000-acre ft (1.2 GL) within one mile (1,609 meters) leads to 0.9 ft (0.27 meter) change in height of water table, under hydraulic conductivity of 20 meters per day. [5]Distance between sub-areas is estimated from the central point of each sub-area. [6] Distance from sub-aquifers to ecosystem site is the average distance of extraction bores for horticulture to the nearest wetlands or bushlands. [7] We assume that the water table level at ecosystem sites is equivalent to the current water table level at each sub-area given many ecosystem sites and the variation of their water level.

Three main scenarios are considered in the analysis: the baseline scenario, where farmers do not internalize the cost of environmental externalities; the internalization scenario, where farmers do internalize the impacts of the environmental externalities; and the uniform proportional cut scenario with five sub-scenarios corresponding to five levels of reduction in the current groundwater allocation limits of each sub-area. For each scenario and sub-scenario, 1,000 simulations were run for each sub-area over 10 years. The model was solved using the DICOPT mixed-integer non-linear optimization solver in the General Algebraic Modelling System (GAMS). Details on model benchmarking and validation that allow us to be confident the model performed reliably are presented in Appendix A.

4. Results

The result section has been divided into two parts. In the first part we discuss the economic and environmental outcomes under internalization scenario. This is followed by a discussion of the economic and environmental outcomes under uniform proportional cuts.

4.1. Economic and environmental outcomes under internalization scenario

Table 3 presents a comparison of economic and environmental outcomes predicted by our model between baseline and internalization scenario. In the baseline scenario, farmers do not internalize the effect of their groundwater extraction on the ecosystem. In the internalization scenario, farmers consider the ecosystem damages cost that links to their production activities.

Table 3

10-years aggregated groundwater extraction, gross margin, and ecosystem damages in three sub-areas with and without internalizing damages cost on GDEs

Sub-area	Scenario	Groundwater allocation limit (GL)	Groundwater extraction (GL)		Gross margin ($m)		Ecosystem damages ($m)	
			Mean	SD	Mean	SD	Mean	SD
Total	Baseline	50.53	45.09	4.89	95.33	36.40	61.33	39.84
	Internalization	50.53	36.51	0.44	75.44	30.59	1.43	0.74
Carabooda	Baseline	33.23	29.73	3.15	62.88	23.99	41.33	29.61
	Internalization	33.23	24.70	0.04	52.07	20.73	0.01	0.09
Neerabup	Baseline	8.65	7.68	0.87	16.19	6.21	4.36	2.22
	Internalization	8.65	6.41	0.44	12.03	5.45	1.39	0.71
Nowergup	Baseline	8.65	7.68	0.87	16.26	6.21	15.64	8.17
	Internalization	8.65	5.40	0.07	11.34	4.51	0.03	0.18

The first thing to note about the results in Table 3 is that for production under a drip and micro-irrigation system, in all three sub-areas, current groundwater allocation limits are not a binding constraint. Overall, on average, the amount of water used is about 89% of the total allocation. This result is especially noteworthy as the assumed summer crop, which is tomatoes, is the most water-intensive crop grown in the region. Historically, groundwater extraction limits were set based on less efficient irrigation technologies, but one interpretation of this result is that for efficient and cost-effective irrigation technologies, current water allocation limits for horticulture could be considered generous. We argue that it is reasonable for the water management authority to redefine water allocation limits for the horticulture sector based on the water requirement determined using a modern efficient irrigation technology (see Appendix B for details on the positive profit benefits associated with drip and micro-irrigation, relative to a less efficient older style furrow irrigation system).

For the internalization scenario, a reduction in groundwater extraction is observed in all three sub-areas, leading to substantial reduction in ecosystem damages compared to the baseline. The gross margin of farmers under the internalization scenario also falls, since the cost of ecosystem damages appears in the objective function. However, the loss in gross margins in all three sub-areas is significantly lower than the gain in ecosystem damages cost reduction. Overall, groundwater extraction falls 19% relative to the current water extraction, gross margin falls by 21%, and ecosystem damages cost falls by 98%. The large difference of high gains from ecosystem damages reduction and the relatively low loss in farm gross margin could be explained by the flexibility in farmers' water extraction behaviour considering the ecosystem damages cost (Figure 5). In all three sub-areas, under internalization the level of water extraction is adjusted accordingly to sustain the initial level of water table which fluctuates with the rainfall recharge patterns (Figure 4). When the groundwater table level declines because of lower rainfall recharge, for example from year 2 to year 6 and from year 9 to 10 (Figure 6), the level of groundwater extraction also declines accordingly (Figure 5). Similarly, when the groundwater table level increases because of higher rainfall, the level of groundwater extraction also increases. Since the ecosystem damages cost is directly linked to the change in water table level, keeping this level unchanged can avoid significant impact to the GDEs.

When no ecosystem damages cost is considered, almost all available water is extracted every year, leading to huge damages cost to the ecosystem due to accumulative decline of the water table level.

Another important result to note here is the heterogeneous proportional reduction in water extraction across the three sub-areas. This water reduction varies depending on the economic and environmental value of per unit of water extracted. The distribution of water reduction in each sub-area, under both scenarios, is summarised via a series of violin plots in Figure 7, where the mean (not median) values are also identified. The level of water reduction is the highest in Nowergup (38%) where the largest areas of both wetlands and terrestrial vegetation are located. The water extracted in this sub-area, therefore, generates more environmental loss than profit gain. In Neerabup, the distribution of water reduction level is wide, ranging from 2% to 38%. The reduced cost of ecosystem damages in this sub-area, under internalization scenario, is the lowest among all three (Table 3). This is because the water extraction in Neerabup has lower environmental value compared to the other two ($0.6/KL compared to $1.3/KL in Carabooda and $2.0/KL in Nowergup). Hence, the economic return of water extraction in this sub-area is able to compensate the loss that it causes to the GDEs.

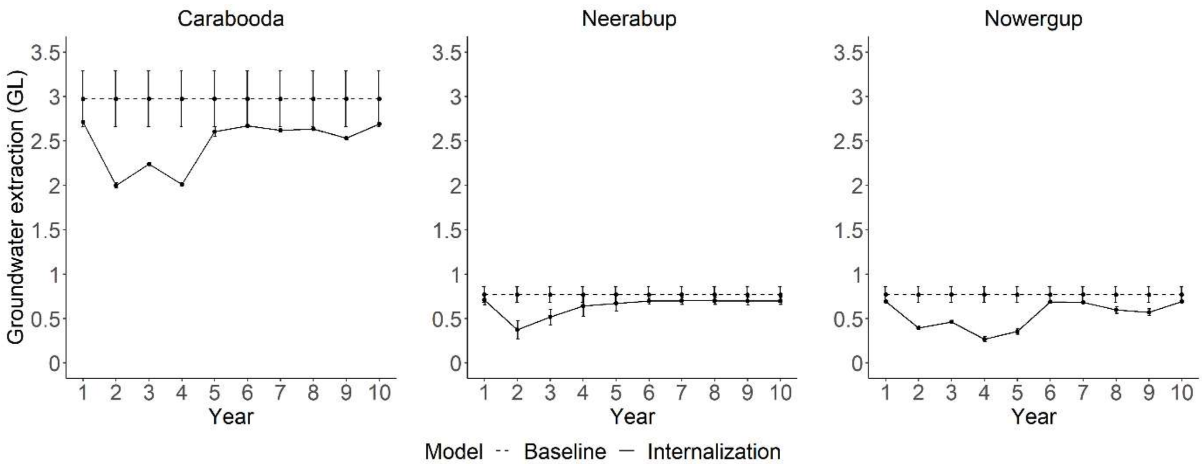

Figure 5 Farmers' water extraction over 10 years under baseline and internalization scenario

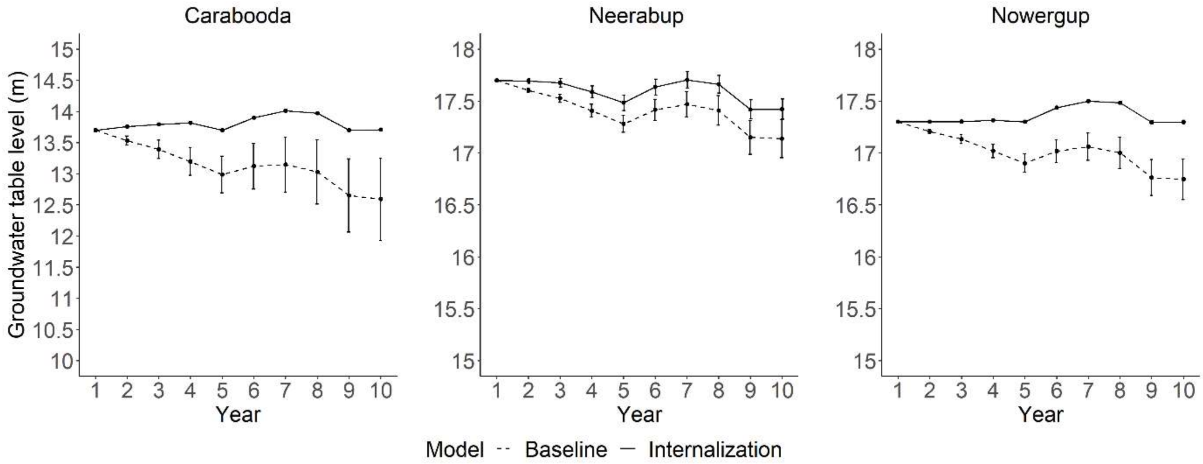

Figure 6 Hydrological change over 10 years under baseline and internalization scenario

The heterogeneous hydrological conditions across the three sub-areas might also contribute to explain the difference in water extraction level. However, given that hydrological parameters, including hydraulic conductivity and spatial distance from the extraction bore to GDEs, of the three sub-areas are relatively similar, we cannot confidently confirm this conclusion.

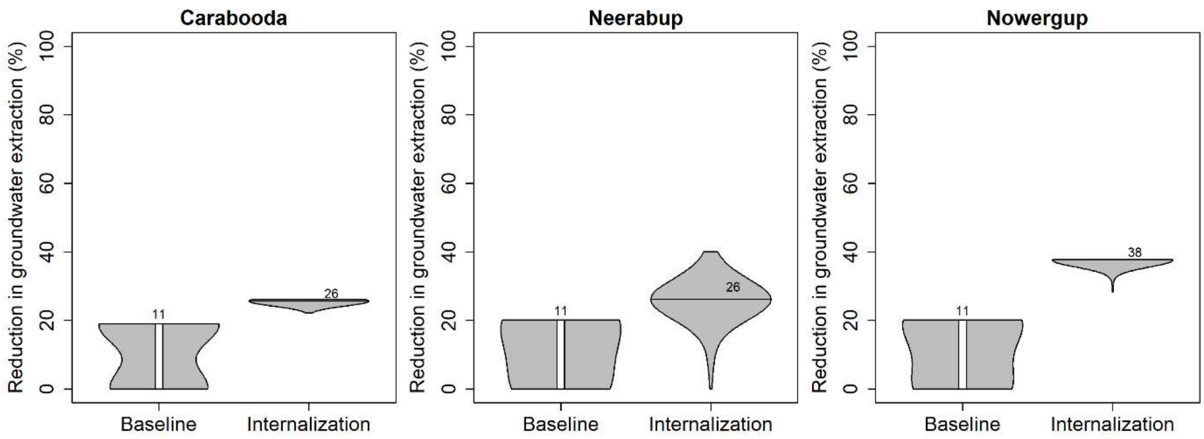

Figure 7 Reduction in groundwater extraction in the three sub-areas

4.2. Economic and environmental outcomes under uniform proportional cut

In the previous section, we found that the internalization approach can reduce ecosystem damages cost to almost zero with a small loss in gross margin. However, this approach might not be as practical as the uniform proportional cut to water extraction , which is widely adopted by many water authorities (OECD, 2015). The uniform cut approach has also been proposed to reduce agricultural water extraction in the study areas. This section provides the economic and environmental impacts using uniform proportional reduction of water extraction and compares to the outcome of the internalization scenario.

Gross margins and ecosystem damages in the three sub-areas under baseline and five levels of cut in current water allocation, from 10% to 50%, are presented in Figure 8. As expected, the results indicate that the performance of the same proportional cut in water extraction varies across sub-areas. A 30% cut results in nearly no damage to the ecosystem in Carabooda while in the other two sub-areas, to achieve similar ecosystem damages outcome, the cut must be from 40% to 50%. However, if a decision of 40% to 50% uniform cut is made, unnecessary loss in gross margin will be experienced in Carabooda without any further improvement in the ecosystem.

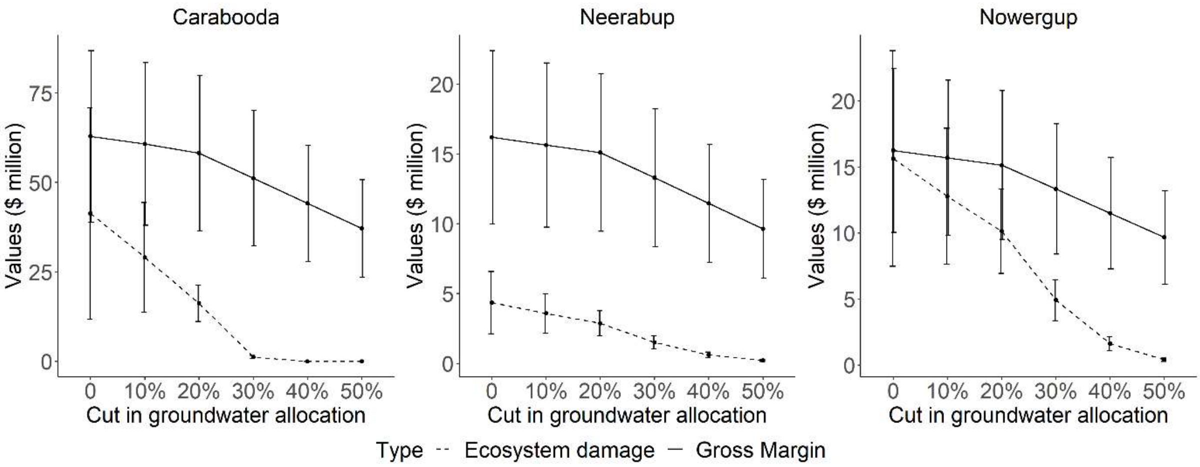

Figure 8 10-years aggregated ecosystem damages costs and gross margin of different levels of uniform cut in groundwater allocation

Note: The bar presents mean +/- standard deviation

Table 4

10-years aggregated total gross margin, total ecosystem damages cost under uniform cut and internalization approach

Water reduction approach	Level of reduction in water allocation (%)	Total gross margin ($m)		Total ecosystem damage ($m)	
		Mean	SD	Mean	SD
Uniform	0 (baseline)	95.33	36.40	61.33	39.84
	10	92.12	34.52	45.42	21.71
	20	88.46	32.94	29.28	9.18
	30	77.84	28.82	7.71	2.42
	40	67.16	24.72	2.24	0.70
	50	56.46	20.69	0.62	0.19
Internalization	Carabooda: 26% Neerabup: 26% Nowergup: 38%	75.44	30.59	1.43	0.74

Table 4 provides a comparison of aggregated gross margin and ecosystem damages cost of all three sub-areas under uniform proportional cut and internalization scenario. The results show that the internalization scenario is more economically efficient than the uniform proportional cut. The estimated ecosystem damages cost under internalization scenario is $1.4 million with a reduction in water extraction from 26% to 38%. For the uniform cut to achieve relatively the same level of environmental damages cost as the internalization approach, the reduction in water extraction has to be within the range from 40% to 50%. Following this cut level, the expected loss in gross margins is from $28 million (29%) to $39 million (41%), rather than $20 million (21 %) with respect to the baseline.

The significant low ecosystem damages cost under internalization scenario compared to uniform proportional cut is derived from the flexibility in the level of groundwater extraction as explained in the previous section. This flexibility under the internalization scenario allows the level of groundwater extraction fluctuates with the weather outcome (rainfall) in order to maintain the initial level of groundwater table level at the ecosystem site to minimize the ecosystem damages. For example, the 10-years aggregated total groundwater extraction in Carabooda under the internalization reduced by 26%, but it declines the highest in year 2 (1.99 GL or 38%, Figure 5) and slowly increases in other years. Similarly, the highest reduction level of water extraction in Neerabup is about 60% (0.37 GL in year 2) and

Nowergup about 69% (0.27 GL in year 6) while the 10-years aggregated total groundwater reduction levels in these two sub-areas are 26% and 38% respectively. With the uniform reduction, the level of water extraction reduces by the same percentage every year regardless of the rainfall. This produces significantly higher ecosystem damages cost since the water table level at the ecosystem sites drops low when the rainfall recharge is low.

5. Conclusions and policy discussion

Reducing groundwater extraction to balance economic and environmental trade-offs is of continuing concern to economists, resource managers and policy makers. In the arid and semi-arid regions such as Australia, the competing demand of water between agriculture and environment is more intense, and the choice of policy approaches to achieve this balance becomes more crucial. This paper discusses the economic and environmental impacts of reducing agricultural water under internalization of environmental externalities and uniform proportional cut in water allocation approach. While the first is more likely to be the most economic efficient instrument according to the environmental economic theory, the latter is more commonly implemented due to its practicability and convenience. Our model results show that internalizing environmental externalities generates substantial reduction in ecosystem damages cost (98%) with relatively low costs to farmers (21 %) compared to the current water allocation. If using uniform proportional water cut to achieve similar level of environmental outcome to the internalization, the loss in gross margin is from 29% to 39%.

Our study highlights numerous advantages of implementing internalization approach to reallocate groundwater in agriculture for ecological conservation. Introducing additional environmental damages cost to farmers' objective function, water extraction achieves the most efficient level which equates marginal environmental cost and marginal farm profits. The internalization approach can capture the heterogeneity in agriculture and ecosystem conditions across locations. For example, in our case study, the level of water reduction under internalization approach varies across the three sub-areas with the extent of marginal environmental and economic value of water. Where water has higher environmental value, the reduction is higher and where water has higher economic value, the reduction is lower. The internalization approach also allows flexibility in water extraction behaviours of farmers depending on rainfall patterns. In all three sub-areas, we observe the trend in water extraction that decreases in dry years and increases in wet years in the internalization scenario.

A relevant policy instrument to internalize environmental externalities is environmental taxes (e.g. pollution tax, emission tax, effluent charges). The tax policy instrument is believed to outperform other regulatory instruments (Boyd, 2003; Kneese & Bower, 1984; Noel et al., 1980; Oates & Baumol, 1975), because it allows flexibility in water extraction decisions for the irrigators and generates revenue for the government which could be used to subsidize public goods such as irrigation infrastructures or agricultural extension services (Pigou, 1920). Ideally, tax should be set at a level which equates marginal costs of reducing environmental damages with marginal benefits of such reduction. However, in reality, implementing an effective tax policy to manage groundwater is challenging due to lack of information on environmental costs of groundwater extraction. In addition, the implementation costs associated with environmental taxes could be very costly. In the case of groundwater, it is necessary to implement meter at all wells to monitor groundwater use if taxes are charged.

In practice, simple approaches such as a uniform proportional cut in current allocation is still a popular management approach even if they are not economically viable. Our results contribute to the empirical evidences of the inefficiency of uniform water reduction policy in WA. The loss in agriculture profits in this study can be used to provide some indication of the potential cost of using a uniform proportional cut policy. Our case study site represents 24% of the total groundwater allocation for horticulture and total horticulture land area in the Wanneroo region. That in turn suggests that the size of the potential benefit from using alternative mechanisms, that are more flexible and allow targeted reductions, rather than a uniform cut that achieves the same level of environmental protection, could be very large (in the order of $49 million to $68 in 10 years). The scale of the benefit suggests that despite the complexity of introducing alternative mechanisms, such a policy would be worthwhile.

The modelling approach and the assumptions used in this paper could be expanded further. For example, we assumed that groundwater users are completely rational, forward thinking and consider the cost of extraction on themselves in the future. We also assume that all users within a given sub-area coordinate at 'zero' cost. In addition, the impact of extraction externalities in our model was reduced due to the assumption of the distance between farm's extraction points. We considered three representative farms located at the central of each sub-area and therefore the distance between farms is large, leading to a marginal impact of extraction externalities. These assumptions were adopted to understand the impact of allocation reduction and the role of environmental damages cost internalization at sub-area scale as clearly as possible. Relaxation of these assumptions would make the difference between the two scenarios (without and with internalization) smaller. However, we did not have adequate and models from WA to formalize such behaviour. In future, surveys could be undertaken to identify individual irrigators' locations for incorporating individual's extraction externalities and to understand the groundwater use decision making models better.

While we introduced the heterogeneity in spatial and economic value of ecosystems between sub-areas in the environmental damages function; we assumed a linear response function of GDEs to the groundwater depletion. In reality, the change in physical conditions of GDEs caused by declining groundwater may be more complex and thus require further investigation. In addition, our model considered the monetary value of two types of environmental damages cost (wetlands and terrestrial vegetation), which are only a fraction of the total environmental values. The farm optimization model only allow selection of areas and ignore choices related to water application rates and irrigation technologies. Further, we do not explicitly consider interactions between different types of groundwater users (such as plantations, water utilities, etc.) and dynamic updating of behaviours of individual irrigators. Finally, we focus only on horticultural sector. Future studies could include the impact of groundwater extraction by other sectors.

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FAIR data policy statement:

Since simulation-optimization modelling was used to generate the results in this paper, no primary data is required. All parameters used in the model were included in Table 2. GAMs codes are available upon request.

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Appendix A: Validation and benchmarking

Validation and benchmarking

For validation purposes, we compare the estimates for gross margin per hectare, gross margin per mega litre (ML) of groundwater extracted, and the environmental cost per kilolitre (KL) of groundwater extracted under the baseline scenario to values from other sources (Figure A1).

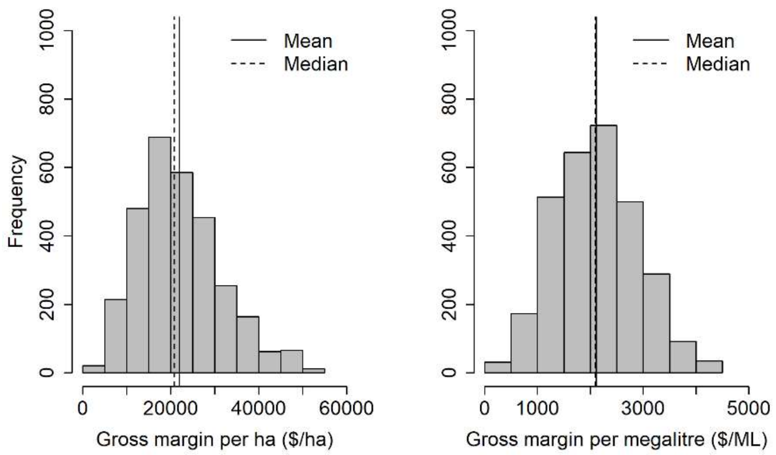

Figure A1 Gross margin per hectare, per megalitre (ML) of water

Gross margin per hectare. Initially, we ran the simulations where we allowed the vegetable price to be drawn from a range set to the highest and lowest monthly price report for 2017. With this range for prices, in 25% of the simulations there was zero profit. This result overlaps with the financial report of (VegetablesWA and Plantfarm Pty Ltd., 2018) where the bottom 25% of growers receive negative profit. However, this study aims to estimate the impact on farm's profit of the internalization of the environmental externality. In order to internalize the cost of externalities, gross margin is expected to be higher than zero. Thus, we narrowed the range for vegetable prices to be plus and minus 20% of the average price. With this assumption the average gross margin per hectare is \$22,003, and this level of return is consistent with the average gross margin per hectare of the top 25% growers reported (VegetablesWA and Plantfarm Pty Ltd., 2018).

Gross margin per megalitre of groundwater. The average gross margin per ML of water extracted is \$2,110, and this falls within the range of the inflation adjusted individual vegetable gross margins per ML of water reported in Hickey et al. (2006) of \$1,163 to \$4,098. The inflation adjusted estimate of Hoffmann et al. (2005) for tomatoes is slightly lower (\$1,919) than our estimate, but this is also consistent with expectations given our assumption for the price distribution. On this basis we see no evidence that the model values are materially different to other values reported in the literature.

Appendix B: Impact of irrigation technologies on groundwater extraction, gross margin, and ecosystem damage

The performance of four different irrigation technologies: Furrow, Centre Pivot (CP), Impact and Rotating (IR) and Drip and Micro (DM) are evaluated to measure the impact of irrigation efficiency on groundwater extraction behaviour of farmers, farm gross margin, and environmental damage under the baseline scenario (no internalization). Figure B1 shows total water extraction, gross margin, and ecosystem damage cost of all three sub-areas under four different technologies. From Figure B1, it can be seen that farming under DM irrigation technology requires the least amount of water, and hence imposes the least damage on the

GDEs while producing a level of profit almost identical to that of Centre Pivot and Impact and Rotating systems.

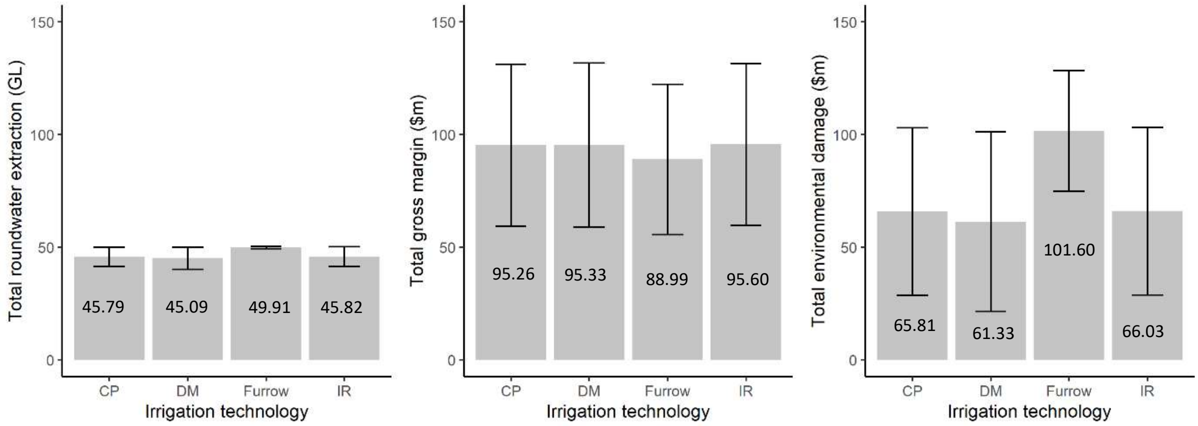

Figure B1 Total Water Extraction, Gross Margin and Environmental Damage of Different Irrigation Technologies

Note: The bars in the figure represents Mean ± SD

Appendix C: Sensitivity analysis of the discount rate

Table C1 compares the impact of using both a higher (5%) and lower (1.5%) discount rate relative to the 3% that is used in the base case. Details are reported for water extracted, gross margin, and environmental damage under the internalization scenario. The results show that farmers' water extraction is not sensitive to the discount rate assumption. Gross margin and environmental damage follow the expected pattern and are lower with higher discount rates, but the extent of the variation is not substantial. On this basis, we conclude that the core results are not sensitive to the discount rate assumption.

Table C1

Water Extraction, Gross Margin, and Ecosystem Damage under Different Social Discount Rate

Sub-area	Social discount rate	Groundwater extraction (GL)		Gross margin ($m)		Ecosystem damage ($m)	
		Mean	SD	Mean	SD	Mean	SD
Carabooda	5%	36.49	3.15	68.13	27.80	1.31	0.65
	3% (baseline)	36.51	0.44	75.44	30.59	1.43	0.74
	1.5%	36.51	0.39	81.82	33.19	1.51	0.79
Neerabup	5%	6.46	0.45	10.89	4.94	1.40	0.72
	3% (baseline)	6.41	0.44	12.03	5.45	1.39	0.71
	1.5%	6.35	0.43	13.42	5.99	1.38	0.70

Sub-area	Social discount rate	Groundwater extraction (GL)		Gross margin ($m)		Ecosystem damage ($m)	
		Mean	SD	Mean	SD	Mean	SD
Nowergup	5%	5.43	0.07	10.19	4.07	0.03	0.16
	3% (baseline)	5.4	0.07	11.34	4.51	0.03	0.18
	1.5%	5.39	0.06	12.26	4.86	0.03	0.19