Irrigating the Environment

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Abstract

Internationally, countries are adopting alternative approaches to redistribute water between users to mitigate economic loss and sustain ecosystems for current and future generations. A distinction is often made between the welfare derived from economic uses of water compared to environmental uses of water. This distinction provides opportunities for those seeking to challenge the economic benefit of restoring environmental flows, and to increase their private individual gains from water use, at the expense of national welfare benefits. However, we can think of a production system's demand for water, whether economic or environmental, as a combination of inputs used to preserve capital (private or natural) and to generate welfare benefits. Modelling can provide insights into our understanding of national welfare benefits from water reform. For a model to understand the trade-offs between users it needs to incorporate: profit maximising objectives for economic water uses; environmental objectives through institutional constraints; the difference between water inputs used to preserve capital or generate welfare; how both users/uses of water are impacted by biophysical system limitations; and how water supply risk and uncertainty affect welfare outcomes. This paper outlines a modelling process designed to incorporate these factors into a framework aimed at: i) justifying that the environment can be treated as just another consumptive water user; ii) providing an alternative approach to the treatment of water as a production input; and iii) examining three policy mechanisms that may be applied to prevent irreversible welfare losses under any misallocation of resources by state of nature.

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Highlights:

- Water is classified as an input to preserve capital (financial or natural) or generate output
- Modelling framework developed to evaluate benefits of restoring water to the environment
- Behavioural (human and nature) responses to realised water supply
- Policy mechanisms: trade, carryover and water-use efficiency, are explored

Introduction

Worldwide, water resources are inequitably distributed and over-allocated to economic consumption, creating negative externalities that reduce welfare (Gómez Gómez et al., 2018). The reallocation of water to the environment can provide welfare gains (Quiggin, 2001) for all water users (cultural, economic, social and environment), if that reallocation is managed in the national benefit (Ciriacy-Wantrup and Bishop, 1975). However, in the absence of clear, fixed and well-defined environment objectives, the environmental water reallocation process may be compromised, as rent seeking opportunities emerge and public (national) welfare benefits are reduced (Huppert, 2013). Better understanding of total national welfare benefits and the trade-offs associated with alternative water resource uses can reduce the opportunities for, and negative effects from, rent seeking behaviour by private individuals or groups. Modelling approaches (e.g. partial equilibrium models) provide a platform to frame and test welfare trade-offs. However, if water resources are perceived and/or treated as providing distinctly different economic and environmental welfare gains, then the outputs from models may be summarily dismissed by rent seeking individuals.

The process of water reallocation in Australia has combined institutions, regulations, and market-based instruments to reallocate water resources toward the environment (Crase et al., 2011). In this process, consumptive (economic) water rights have been transferred (i.e. no change in the type, purpose, reliability or definition of the property right) to an environmental manager that is responsible for allocating the available water between a set of competing ecological objectives that, preserve natural capital and/or produce ecological outputs. Within uncertain and variable water supply constraints similar to that of consumptive users, this approach is aimed at protecting biodiversity and restoring previously degraded ecosystem services, allowing for a process of learning and adapting to new institutional arrangements, and optimising economic, social and environmental welfare (Commonwealth of Australia, 2008).

However, prior modelling approaches and their analyses generally: i) treated the environment's supply, demand or welfare gains from water as constant; ii) assumed that the environmental manager is passive to water supply uncertainty; iii) considered water use decisions by environmental or consumptive users as completely independent; and iv) assumed that use decisions have no corresponding impact to the other party, thereby ignoring the hydrological realities of a river system (Gómez Gómez et al., 2018; Horne et al., 2018).

This article proposes that we can model both the management and use of environmental water in exactly the same way as that of a consumptive user (e.g. a farmer). By treating all water users as identical, we can identify useful insights into joint behavioural responses to managing the risk and uncertainty associated with water resources. Central to this objective is an assumption that a production system's demand for water, whether economic or environmental, involves a combination of water inputs used to preserve capital (private or natural), and water inputs that generate welfare benefits. To assess the trade-offs between, and maximise our understanding of, national welfare benefits we must therefore incorporate: profit maximising objectives for economic water uses; environmental objectives through institutional constraints; the difference between water inputs used to preserve capital or generate welfare; how both users/uses of water are impacted by biophysical system limitations; and how water supply risk and uncertainty affect welfare outcomes.

To justify this argument the following article is divided into the following sections. First, a literature review outlines the key features of water resource economic modelling and argues why we can consider the environmental manager as just another water user. Second, the methodological framework for the model is detailed together with recent findings in state-contingent analysis to help explore behavioural responses to realised water supply (drought, wet and normal). This work focuses on the separation of water into inputs that preserve capital (natural or financial) and/or generate outputs, to help explain consumptive and environmental water use decision-making behaviour (Adamson et al., 2017). Third, the model is described with a focus on highlighting the benefits from modelling all water users' behaviour with respect to desired (constrained) environmental objectives. The article then explores how decision-makers can adapt using three policy or institutional mechanisms: trade, carryover and water-use efficiency with respect to any separation of water's role in production. Finally, a wider discussion is presented, before final comments are given.

Water Resource Modelling and the Environmental Manager

The conceptual framework outlined in this article draws inspiration from Australia's Murray-Darling Basin (MDB) Plan (the Plan). The Plan embraces the concepts of common property via seeking to recover a portfolio of excludable and enforceable environmental water rights to meet a set of environmental objectives (Adamson and Loch, 2018). These water rights are recovered from a reallocation of a fixed number of rights to access water (Quiggin, 1986) between groups, and not the introduction of a new set of rights for the environment. Reallocating rights to the environment means that society may gain additional rents from subsequent environmental restoration actions (i.e. differential rents with and without an environmental manager), where increased environmental flows improve ecosystem functions and quality, benefiting all water resource users (Ciriacy-Wantrup and Bishop, 1975). Additional welfare benefits are gained in perpetuity as the preservation of environmental goods prevents further public investments to reverse harm, and/or minimises the possibility of irreversible losses to defined environmental objectives. Irreversibility, is defined as the combination of any positive social discount rate, and the time to regenerate the environmental asset to its original (or desired) state that essentially makes the losses irreversible (Arrow and Fisher, 1974).

It should be noted that the exploration of welfare benefits in this article via a reallocation of environmental flows is limited to river basins where extensive modification has occurred. Modification of river basin water resources includes any intensive and extensive development that has altered: river inflows (landscape modification dams, removal of trees, etc.); stream flow patterns (irrigation demand, weirs, locks, irrigation systems, etc.); the river's morphology; water quality; and water temperature (Sherrick et al., 2004; Walker, 1985). The definition of environment in this context is therefore important, as extensively modified river systems are no longer natural basins, nor will the environment be returned to pristine condition as other water users are still present, and trade-offs still occur. In other words, what is the level of welfare harm that society deems acceptable (Coase, 1960).

This article defines the 'environment' as the targeted ecological objectives and assets that public institutions seek to preserve, maintain or improve (Levin et al., 2013). Consequently, these environmental objectives determine the environment's demand for water that must be met via the environmental manager's portfolio of water property rights[1]. However, both the environmental objectives and the reliability of the portfolio of rights will alter in respect to the realised supply of water by each state of nature (drought, normal and wet). Additional non-targeted environmental welfare benefits or positive externalities may also occur from the application of the environments water. However, positive externalities could be considered as a sub-optimal resource allocation of water in the short-run, if the environments objectives could be achieved with less water. Further, these public institutions need not be passive in their arrangements, utilising existing and new infrastructure to (re)allocate water throughout their portfolio. However, care is needed as these investments may lock in management goals and strategies that could be inflexible to future adaptation requirements (Pittock et al., 2013).

[1]A water portfolio is a set of water rights (high, general and supplementary) where the supply reliability of each water right is determined by location and state of nature (drought, average, wet).

Economic models to help assess trade-offs

To maximise the value of a model's relevance for institutional policy analysis, its developer/s must consider how the models design, scale and scope (Loch et al., 2014) will influence the relationship between water allocations, uncertainty, non-convexity and irreversibility (Howitt, 1995). A model's scale details how: individual/s and geographical region/s are grouped; and the time frame for the policy review; and scope defines the issues explored by alternative policy designs. As the scale and scope of any economic model increases, it allows for greater understanding of the opportunity costs and trade-offs between groups and outcomes. Scaling issues allow for: determining the value of exploring a single of conjunctive water resources (Pulido-Velazquez et al., 2011; Stahn and Tomini, 2017); and the trade-offs that occur between individuals, alternative water users in different hydrological or political boundaries (Berck and Lipow, 1994). While scoping issues provide the capacity to see: how the value and quality of water changes in a landscape in response to a policy setting (Quiggin, 1988; Schwabe, 2000); exploring the binding institutional and hydrological constraints; and scaling up from first-round optimisation to analysing second round economic impacts (Brouwer and Hofkes, 2008). Understanding the scale and scope of the issue then may prevent sub-optimal outcomes (Connor et al., 2013; Watanabe et al., 2006).

Further economic clarity can be incorporated into modelling approaches by: including fixed costs to prevent bias towards existing water owners (Randall, 1975); articulating the difference between water prices and charges (Schuck and Green, 2002; Ward and Michelsen, 2002); and understanding the limitations of basin wide and in field water-use efficiency in terms of reduced resilience, increased water use and reduced river flow (Gómez and Pérez-Blanco, 2014). Although clearly complex, these additions increase methodological rigour in the model analysis, and may prevent unintended policy consequences (Adamson and Loch, 2014). Therefore, a critical research question is, how can we best represent the inherent risk and uncertainty of water supply, as well as any nonconvex behavioural responses to a realised supply, so that policy solutions may be identified that help prevent irreversible losses for water users?

In the MDB, annual drought and flood events define the realised water supply (Khan, 2008). In response farmers, rural communities and ecosystems have learnt to actively adapt. Farmer adaption may take the form of reallocating inputs, changing production systems, choosing to opportunistically irrigate; and using water markets to reduce risk (Mallawaarachchi et al., 2017). Rural communities may adapt by reducing their exposure to irrigated-farming, diversify toward alternative sources of community income, and/or invest in training and adjustment packages to ease structural change (Edwards et al., 2008). Ecosystem adaptation revolves around capitalising on flood events to increase total environmental goods ((e.g. increase the size (area) and/or density of a native species; Kingsford et al., 2010) and improved biodiversity quality by preventing genetic fragmentation derived from poor river connectivity (Kelley et al., 2017). During drought states, key *refugia* sites provide critical habitat for species to emerge from and repopulate during normal or wet states of nature.

Importantly, both this variability in realised water supply, and the breadth of total decision-maker choice-options in response to realised water supply, must be accounted for by policy makers (Just and Pope, 2003).[2] While significant uncertainty about future climate and realised water supply outcomes prevail in many contexts including the MDB. However, climate change predictions for the MDB suggest that water supply will experience both a mean reduction to inflows, and an increased frequency of extreme events (i.e. droughts and floods) (Chiew et al., 2008; Garnaut, 2011). This combination of climatic events will pose future

[2] This paper simplifies the relationship between the environmental decision manager and the environment to a production system where given a combination of inputs (i.e. water or infrastructure) stimulates an environmental response (output). Logically, arguments about the quantity of inputs and the timing of those inputs is required, and as such, key responses are not well reflected in an annual model framework.

complexities (Chugh and Bazerman, 2007) for policy-makers, and require water resource decision-makers to learn and adjust existing/explore new adaptations in response (Goldstein and Gigerenzer, 2002; Marangos and Williams, 2005). Three key adaptation strategies to water supply risk and uncertainty include: how these decision-makers will utilise permanent (entitlement) versus annual (allocation) water trade strategies; the use of carryover provisions to store water in a dam or other structure for use in subsequent seasons; and/or investing in water-efficient technology.

Central to the effectiveness of these adaption strategies will be an appreciation that the value of water resources can transition from elastic to inelastic in response to realised supply (Randall, 1981). Adamson et al. (2017) suggested that farmers may be willing to purchase water at high prices—and even incur a net financial loss in the short-run—to preserve their capital investments and long-run returns from perennial crops (e.g. almonds). Thus, while the initial value of water can be linked back to water quality characteristics, potentially the real value of water to a producer can be separated into two key features: i) the volume required to preserve capital investments, and ii) the water required to generate productive outputs from that investment. While the social value for improvements in the quantity or quality of water can be determined (Bergstrom and Loomis, 2017; Brouwer and Sheremet, 2017; MacDonald et al., 2011), these studies do not quantify changes to social values by realised water supply; particularly drought states. However, it is logical to assume that water's value to society and the environment is not constant by state of nature.

The contribution of this paper

This paper proposes that any public institution decision-maker responsible for managing the environment fundamentally faces the same questions as those of a farmer when allocating water between competing uses. The environmental manager has: the same set of water right structures as a farmer; access to state-allocated water inputs to produce state described outputs (i.e. a set of choices between alternative sets of environmental assets) over space and time; some capacity to alter their delivery and demand schedule. In addition, the use of water by the environmental manager in one area, will create downstream impacts on the quantity and quality of water available to other users. However, unlike a farmer, the environmental manager is also responsible for achieving broader environmental objectives (e.g. fixed water quality and river flow targets).[3] In both cases though, these objectives reduce down to a question of how much water do they have, and how can they best allocate it to maximise or satisfy their objective function(s) (Simon, 1947). Consequently, all environmental water can be treated as an input for three production choices: environmental assets, water quality and water flow targets, to achieve the institutional (the Plan) objectives.

Our conceptual economic model must therefore: account for the portfolio of water rights; efficiently manage the environment's share of water; and determine the trade-offs for multiple water users in a closed basin. Central to such a model will be a directed water flow network that highlights the opportunity costs of water use in a location, as well as Randall's (1975) suggestion that any constrained optimisation framework must allow for testing how alternative institutional goals (e.g. water quality, water shares, environmental objectives) and alternative policy mechanisms (e.g. water market trading, water-use efficiency, and carryover) impact on private economic returns from water use (i.e. income minus fixed and variable costs of production). Consequently, the economic model described herein does not place a value on environmental assets. Rather, it explores how environmental decision-makers may utilise water, the implications from changed institutional goals (reflecting social expectations) when

[3]A farmer's water use may be constrained by water quality targets and a lack of flow in a river, but they are not actively required to allocate water to those uses, unlike any environmental manager. Logically, in some cases by actively allocating inputs, the environmental manager may allow private users to consume more water that before. This is akin to salinity interception schemes in the MDB that have allowed farmers to use more water locally and pass the salt-problem onto downstream users to deal with (Quiggin 1991).

the supply of water is realised, and any impact that may occur under known and future climate settings. To achieve these outcomes, the model adopts a stochastic state-contingent approach (SCA) that is able to deal with water supply uncertainty; that is, it has the capacity to separate water supply uncertainty signals (i.e. water scarcity) from the decision-makers' response to that uncertainty (Chambers and Quiggin, 2002). SCA modelling approaches have to-date provided greater clarity around user adaption to alternative periods of water scarcity, and highlighted users' willingness to pay for water between different production and users types, in both theoretical settings (Adamson et al., 2017) and applied analysis (Mallawaarachchi et al., 2017). We detail the SCA approach in the following sections.

State-Contingent Analysis & Modelling Decision Making Responses

This section of the paper draws heavily from work published by Chambers and Quiggin (2002) who specified the state-contingent properties of stochastic production functions, including the Just-Pope stochastic production model.

SCA foundations are derived from the work by Arrow (1953), Arrow and Debreu (1954) and Debreu (1959), who pioneered the state-space approach to transcribe all possible outcomes $(s\epsilon\Omega)$ from uncertain events across alternative states s. This insight provided the rationale that, once s was revealed, decision makers actively respond to that s by altering inputs (x) to influence the final output (z), based on their past experiences to managing risk. Therefore, the objective function of the producer does not solely concentrate on the production of a single commodity, but rather the net return (y) from all commodities contingent upon both the commodity's payout by state of nature, and the probability (π) of those states occurring $(s\epsilon\Omega)$. SCA builds on these foundations by merging the state-space with dual optimisation, allowing resource use to be optimised by each state of nature, time, place, type[4] (Rasmussen, 2003). This article suggests that the role of resources also needs to be optimised by each state of nature, where role is separated as an input for capital investments and production.

Therefore state-specific inputs are optimised in the traditional way where once s is revealed, inputs are applied up to the point where marginal cost equals marginal value product. Assuming risk neutrality, state-general input use is optimised by ensuring that the expected value of marginal product remains greater than the marginal costs of each extra unit of input. State-allocable inputs (and all inputs allocated across states) are applied up until the point that the input price is equal to the marginal value of products produced across all states of nature. See Rasmussen (2011) for more information on the conditions of optimality. The separation of inputs into those that generate output and those that preserve capital must follow these rules.

SCA production systems and input usage

Chambers and Quiggin (2002) provided a SCA representation of a Just-Pope formulation of production (equation 1) as:

$$z_s = f(x_s, \varepsilon_s) = g(\boldsymbol{x}) + h(\boldsymbol{x}_s)\varepsilon_s. \tag{1}$$

where the inherent variability (i.e. the error term ε) in z is an additive combination of a non-stochastic technology (g) and the multiplicative uncertainty $(h(x_s)\varepsilon_s)$ associated with the use of a vector of inputs $(\boldsymbol{x})$. Two recent advances in describing SCA production systems Adamson et al (2017) and Mallawaarachchi et al. (2017) have both have adapted this state-contingent representation of the Just-Pope production function (see Figure 1 for the differences in interpretation).

[4] Refers to the three input types: ***non-state-specific (or state-general)*** inputs and these must be allocated *ex-ante* to the s being realised and they influence z in all s; ***state-specific inputs*** are applied *ex-post* the realisation of the s as they influence z in only that s; and ***state allocable (flexible) inputs*** that are applied *ex-ante* to s being realised and benefits accrue once to s is realised.

Mallawaarachchi et al. (2017) proposed that z is a function of both the natural resource endowments ζ (e.g. soil fertility, rainfall, etc), and natural variability in output h derived from the use of an input, (equation 2). ζ then represents the differential (Ricardian) rent of land:

$$z_s = \zeta_s + h(\boldsymbol{x}_s)\varepsilon_s. \tag{2}$$

This formulation assumes that $\boldsymbol{x}$ is not allocated by ζ to produce z, but rather that the natural resources a decision maker cannot control by s will influence $\zeta's$ contribution to z. If we assume z is fixed (both in terms of output and commodity choice), and we reduce the vector of natural resources to rainfall, then in good rainfall years the contribution of ζ increases allowing the decision-maker to reduce $\boldsymbol{x}$. In bad rainfall years, the decision-maker increases or changes the set of $\boldsymbol{x}$ to offset a lack of z from ζ (see Figure 1). By highlighting the nature of adaptability in a dairy production system, Mallawaarachchi et al. (2017) highlighted the ability of decion makers to swap inputs (i.e. rather than grow pasture with water, sell water and buy pasture to generate milk) to maximise their objective function.

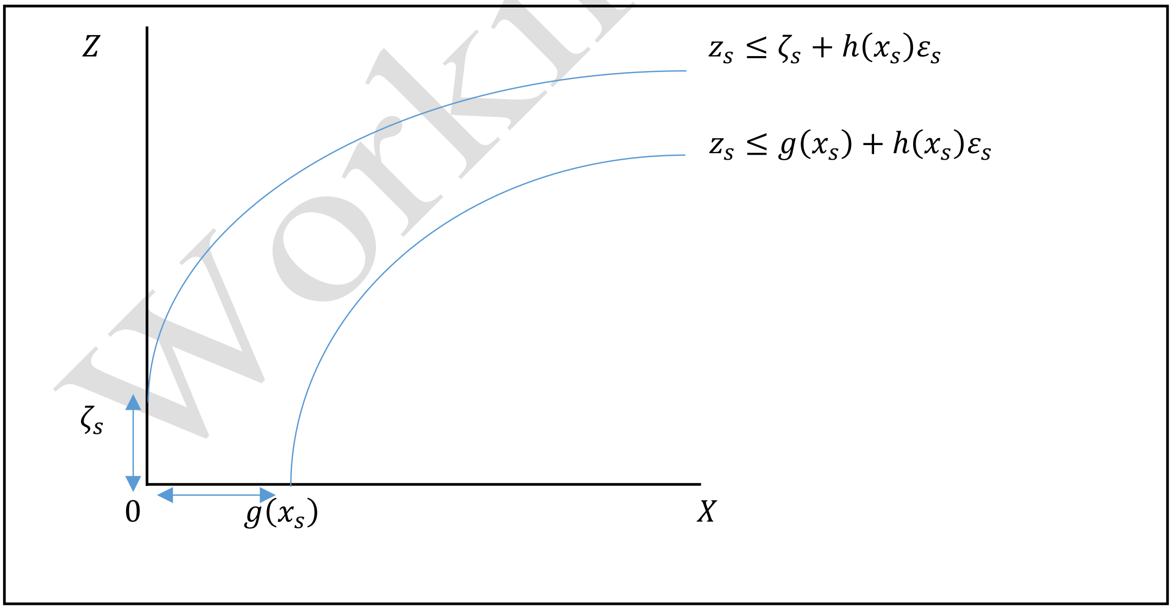

Figure 1. Two alternative forms of a state-space production function, where $\boldsymbol{x}$ is water inputs.

Adamson et al. (2017) suggested that z_s from a production system is derived from a combination of state-general water inputs (before s is realised), and state-allocable inputs (benefits accrue one s is realised). For this paper we define water inputs into state-general and state-specific inputs. From this starting point, by rewriting Adamson et al. (2017) into the SCA formulation of the Just-Pope equation, and assuming that: g represents state-general inputs; state-specific inputs are represented by h; and x represents water used, we get Equation 3:

$$z_s \leq g(x_s) + h(x_s)\varepsilon_s. \tag{3}$$

Equation 3 states that, the same input can have different roles in the production function. By suggesting that g could be considered as the quantity of inputs required to keep a commodity alive by s, and h as the volume to generate output, the equation can highlight differences in management strategies between perennial and annual producers as they respond to realised water supply in each state of nature. In this case, $g(x_s)$ must be allocated to maintaining the underlying capital, before any water is allocated to production from the perennial crop. By noting that annual producers[5] are not constrained by g (i.e. do not experience capital losses associated with root stock), that set of decision-makers could adopt more flexible adaptation strategies (e.g. plant an annual crop or sell water) in response to water supply and water price signals, once s was realised.

Perennial producers face irreversible loss consequences if they mis-specify future water supply, are unable to source additional water, and/or are unable to pay the market price for water (in a drought at the short-run choke price)[6]. Consequently, $g(x_s)$ becomes a binding constraint and a risk-averse decision-maker would ensure that their combination of water rights

[5] For an annual producer Equation 3 simplifies to $z_s = h(x_s)\varepsilon_s$, or simply the multiplicative uncertainty associated with production (Chambers and Quiggin 2002).
[6] The short-run choke price is where price of water is greater than the return from its use as farmers are willing to spend the money on a multi-year commodity to preserve future earnings. Note, this management strategy can only occur periodically if in the long run, profit is greater than or equal to zero, otherwise the commodity choice must change.

provides a minimum supply of water to meet that demand. By relaxing the assumption that g is purely a state-general input (i.e. all g must be allocated before s is realised), it can be thought of as a combination of both state-general and state-specific inputs, (i.e. to keep the rootstock alive, water is allocated over the entire year) where the total inputs of g required are revealed as the state is released. Equations 2 and 3 are merged taking advantage of their respective benefits in understating the allocation of inputs by s (Equation 4):

$$z_s \leq \zeta_s + g(\boldsymbol{x_s})\varepsilon_s + h(\boldsymbol{x_s})\varepsilon_s. \tag{4}$$

Equation 4 then helps illustrate not only the value of different areas ζ, but the risks associated to capital (financial or natural) if the production system requires a large fixed quantity of g. By defining the environmental objectives requirements of water by g and h by s, the risks to meeting those objectives can be explored across a basin. Consequently, events measurable with respect to S define how the water user responds to information, but events measurable by ε account for the uncertainty associated with the true quantity of water that will be required by s. As experience increases the variance of ε should decrease, unless the future changes either/or the frequency of s or the description of S (Goldstein and Gigerenzer, 2002).

Formal Model Description

As discussed, this model includes: a directed river flow network to account for opportunity costs of water use (income forgone, net water used and changes in water quality); capital costs to prevent bias towards existing water right owners; and signals as to how decision-makers adapt to risk and uncertainty. The modelling is divided into four sections. Initially the changing nature of water and the delivery network that supplies all users with their volume of water is presented. Next, the role of inputs in the SCA production systems is explored. Then the objective function of the model is provided, and lastly the institutional goals and management options are discussed. Only where required, will the equations and discussion be separated between farmers and the environmental manager.

Modelling water supply and its delivery

A basin's river system can be considered as a delivery network of water inputs for all water users. By dividing a river network into a series of river segments and environmental assets to deliver water to, hereafter catchments $(k = 1, \dots, K)$, a directed flow network can help illustrate the opportunity costs of water used in one catchment on subsequent catchments in the basin (i.e. exclusion of subsequent use, changes to river levels, and water quality). The water resource available in each k is a combination of its conjunctive water resources and, if not at the start of a river, inflows from the previous catchment.

The total conjunctive supply of water θ[7] of each k by s is defined by:

$$\theta_{ks} = sw_{k,s} + gw_{k,s} + tw_{k,s}, \tag{5}$$

the sum of surface (sw) water, ground water (gw) and inter-basin water transfers (tw) into the basin of interest (Brouwer and Hofkes, 2008; Harou et al., 2009). The separation of θ into separate components allows the model to individually or collectively explore these issues. For example, on-going landscape change (Pittock et al., 2013) and infrastructure modifications (Wilson, 2015) will alter surface flows. The inclusion of gw helps account for total return flows into the river system (Scheierling et al., 2006), and tw highlights the change in welfare benefits gained from meeting multiple objectives across the basin (Gómez Gómez et al., 2018; Rey et al., 2016). The current model formulation ignores the externalities created from transferring water out of any catchment, but tw can be altered to explore changes in welfare. By further altering the frequency of s occurring and the parameters of θ, issues such as climate change

[7] Where θ is a megalitre (ML) of water, 1 ML = 0.8107 acre-foot.

and policy decisions concerning access to additional sources of water sw, gw or tw can all be represented (Adamson and Loch, 2018; Adamson et al., 2009).

The quantity of water (wf) within each k by s, is a combination of θ, plus, if applicable (i.e. k is not the source of a river), wf from the immediate upstream catchment $(k-1)$. From this we subtract water used for irrigation (wu), add any return flows from that water use (rf), and adjust for conveyance losses (v):

$$wf_{k,s} = \left[\left(\theta_{k,s} + wf_{k-1,s}\right) - \left(wu_{k,s} - rf_{k,s}\right)\right] * (1 - v_{k,s}). \tag{6}$$

While v is represented as a percentage, the equation could be adapted to include a fixed (e.g. a quantity of water (vq) required to get the river flowing, so that $\left(\theta_{k,s} - vq_{k,s} + wf_{k-1,s}\right)$) and variable component, $v_{k,s}$, to account for changes in alternative river heights (e.g. over banking events, etc.).

Salinity (σ), expressed in electrical conductivity (EC), (Equation 7), is used to represent the quality of water within the river, as it is a fixed institutional objective in the Plan. EC is a determined by the total salt load (tg) (Equation 8), divided by the river flow:

$$\sigma_{k,s} = \left[\left(tg_{k,s}/wf_{k,s}\right) \times 1000\right]/0.64. \tag{7}$$

The directed flow network also helps illustrate how upstream water use degrades the quality of water downstream. In this case tg for each k by s is the combination of the natural mobilised salt g, plus, if applicable (i.e. k is not the source of a river), g from the immediate upstream $k-1$, less any exogenous removal of salt and the salt that re-enters the river system with return flows $[rf(1-gr)]$:

$$tg_{k,s} = \left(g_{k,s} + tg_{k-1,s} - ge_{k,s}\right) + rf_{k,s}\left(1 - gr_{k,s}\right). \tag{8}$$

Note, $tg, g\ and\ ge$ are measured in tonnes of salt, and $gr < 1$. Further, while σ is an oversimplification of water quality (i.e. a dilution equation), it highlights the positive actions that can be undertaken to reduce (or prevent) pollution.

The variability of θ_s and wf necessitates that the allocation of water for consumptive use must be a share of a common pool resource (OECD, 2015). Consequently, individuals have an 'access right' and not a 'property right' to their water (Quiggin, 1986). An individual's 'access right' (AR) by s is determined by the product of their portfolio of water entitlements (E), a matrix with dimensions $[E\ \times 1]$, that they own in each catchment $(k = 1..K)$, and the reliability of those entitlements *ER* with dimensions $[E\ \times S\ ]$, so that:

$$\sum AR_{ks} = (E_k\ \times ER_{ks}). \tag{9}$$

Logically $AR < wf < \theta$. Thus, the variability in an individual's AR by s necessitates understanding the risk posed to both natural and productive capital, across alternative production systems by g, as discussed above.

SCA production systems, management choices & the role of inputs by state

A central insight provided by the SCA is that water managers do not remain passive once s is revealed. In agricultural systems this has been represented by alternative irrigation technology choices (e.g. flood versus drop systems) and, for all non-perennial crops, farmers have the ability to mix crop choices by s, or opportunistically irrigate, or choose not to irrigate in response to realised s (Adamson et al., 2009). Due to a requirement of g in all s, perennial crops cannot be substituted with other crops once resources are committed, thus preventing unrealistic modelling outcomes.

Environmental managers logically have the same capacity to reallocate resources to achieve z_s. The Commonwealth Entitlement Water Holder (CEWO 2015) identifies $\Omega = 5$, with four management goals that adapt to the environment's AR, as illustrated in Figure 2. The CEWO's objectives are as follows. First, limit harm to the environment in 'very low' and 'low' states when AR is insufficient for all needs. Second, during 'moderate' states, AR must provide sufficient water to protect a greater number of environmental objectives and management aims to maintain the ecosystem, so that it has a greater capacity to respond to better future conditions. Third, maintain existing levels of environmental health to provide the ecosystem with greater resilience. Finally, the CEWO must strategically improve existing ecosystem health e by opportunistically irrigating as many environmental assets as possible to increase resilience to future bad states of nature.

If any of these above environmental objectives are fixed (i.e. must be achieved every year), then those objectives have a constant demand for water akin to a perennial production system (i.e. $g(x_s)$ must be allocated). If we therefore assume that $(\Omega = 1,..,5 = 'very\ low', 'low', 'moderate', 'high', and\ 'very\ high')$, and ignore the script k, the environment's share of water rights must satisfy $\sum g_1 \leq \sum AR_1$. Alternatively, if environmental objectives are flexible (i.e. not required to be achieved in every state of nature), then the environmental manager is provided greater capacity to deal with uncertainty (Kling et al., 2010).

All SCA production systems (M) are a combination of commodity choices and management options that require a vector of resource inputs $(\boldsymbol{x})$ by s to produce z (Equation 8). The six inputs required include: production area (a); water inputs by, g and h, described by stochastic requirement; the stochastic variable costs of production (vc); annualised fixed costs of production (fc); and operator labour (l), so that:

$$\boldsymbol{x} = (a, g, h, vc, fc, l). \qquad (10)$$

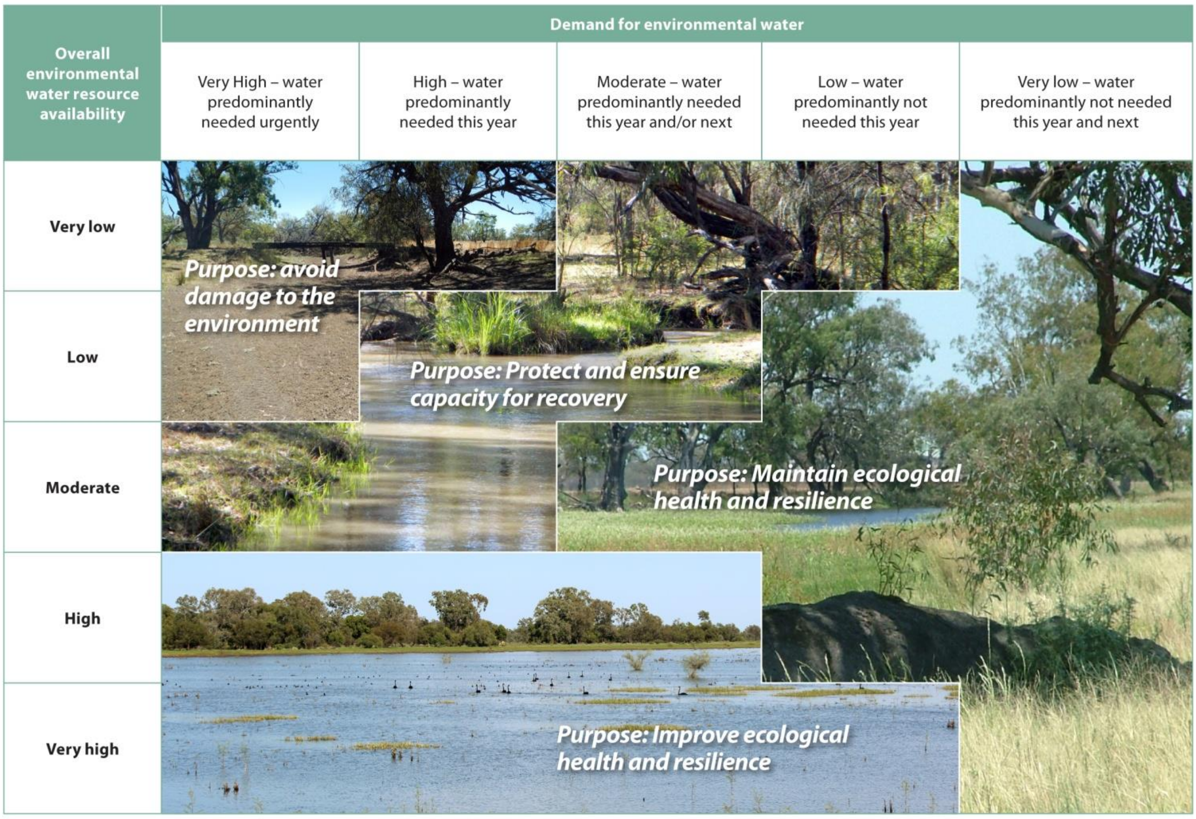

Figure 2. The SCA relationship between the supply of the environments share of water and the management response to the realized state of nature (Figure from CEWO 2015, p8).

where production area represents a single hectare (ha) for a farmer, while for the environmental manager it represents a single environmental objective. In this case the environmental objective may be to deliver water to irrigate a given k that may be greater than a single ha, or a volume of water needed to obtain a set objective (water flow or salinity). By tracking variable costs, the use of all variable inputs of production is incorporated into the model. Additionally, both the use of water and the cost of purchasing water can be modelled (see trade section below), while selling water is included as an output to track income from its sale. The inclusion of fc helps prevent bias towards existing water rights owners. This formulation allows the costs of the environmental manager to be included within the model.

Finally other relevant costs associated with the use of water resources, such as transaction costs from the adoption of new management strategies or trade, could also be included as vc or fc depending on their role and description, (Loch et al., 2018). This approach would then illustrate the outcomes between aiming to minimise transaction costs, or *maximise* the gains from transaction costs.

The constrained welfare framework

The model is solved from the national welfare perspective, where a single agent has control over both the resources utilised by farmers (AG) and the environmental manager (EM). This agent attempts to maximise welfare gains from water use (i.e. water used to irrigate, and water allocation trade between AG and EM) subject to a series of economic, hydrological and policy constraints. This then provides the theoretical maximum possible benefits. Where necessary, the equations have deliberately been written in this section to account for how water is utilised by AG and EM, to make the water accounting exercise easier. For simplicity, the following equations assume that z_s accounts for the output from the SCA production systems.

The objective function (Equation 11) of the farmer is to maximise income $(E[Y])$ across Ω by understanding the frequency π of payouts; comprising revenue (r) (Equation 12) minus costs (c), (Equation 13); where (p) is the price paid per unit of z, less the vector of inputs (Equation 8) multiplied by the vector of input prices $(\boldsymbol{d})$, and the price of $a = 0$.

$$MaxE[Y] \; = \sum_{K} \sum_{s \in \Omega} \pi_s \left(R_{s,k} - C_{s,k} \right) A_{s,k} \tag{11}$$

Revenue: $$r_{s,k} = z_{s,k} p_{s,k} \tag{12}$$

Costs: $$c_{s,k} \; = \boldsymbol{d}_{s,k} \boldsymbol{x}_{s,k} \tag{13}$$

Input constraints: $$\boldsymbol{d}_{s,k} \boldsymbol{x}_{s,k} \; \leq \; \boldsymbol{X}_{s,k} \tag{14}$$

Area: $$a \; \geq 0 \tag{15}$$

Water use:

$$wu_{s,k} = \sum \left[a_{s,k} \times z_{s,k} \times \left(g_{s,k} + h_{s,k}\right)\right] \tag{16}$$

Input usage is constrained by a vector of maximum input use $\boldsymbol{X}$, and a cannot be negative. Note, a can be divided into a maximum area allowable for perennials and annual commodities, which may be useful for calibration purposes or policy analysis (Adamson and Loch, 2018). The water used (wu) by s, is the summation of the total area of each production system, multiplied by the total water requirements of each production system (equation 16). The other major constraints associated with wu, and ensuring that $wf_{s,k} \geq wu_{s,k}$, are discussed below in context of the environmental manager.

The Environmental Manager & Institutional Constraints

The complete set of institutional objectives (EO) that the environmental manager needs to address by k for each s is a vector including: water flow targets $\left(\widehat{wf}\right)$, water quality targets $(\hat{\sigma})$, and the water required $(g$ and $h)$ to irrigate environmental assets (Equation 17)

$$EO_{k,s} = \left(\widehat{wf}, \hat{\sigma}, g, h\right) \tag{17}$$

The environmental manager needs to be proactive in managing water in response to: realised θ, and how all water-users utilise their water in response to θ, as these drivers have the capacity to force management solutions to meet the objectives. Even if both the environmental manager's supply of water AR, and the demand for water to irrigate specific wetlands by k and s are known with certainty, the actual quantity of water required to meet $\widehat{wf}$ and $\hat{\sigma}$ will depend on: i) the actions of other water users in response to the realised θ, and ii) the corresponding impact those choices have on Equations 6 (water flow) and 7 (salinity). Consequently, any model interested in understanding the joint behavioural responses to water resource availability under risk and uncertainty must explore the synergies between all water users. However, this can be further complicated by the binding hydrological realities of total water use (Equation 18). In addition, water uses in any catchment must be less than water flowing through that catchment (Equation 19), which is vital to understanding the multiple trade-offs involved.

$$\sum wu = \sum wu_{AG} + \sum wu_{EM} \leq \sum AR_{AG} + \sum AR_{EM} \tag{18}$$

$$\sum wu_{k,s} \leq wf_{k,s} \tag{19}$$

How EO constrain the allocation of water is also important. As Adamson and Loch (2018) found, when EO are constrained by each s it may be highly restrictive or even impossible to meet those objectives if $\sum AR_{EM,k,s}$ fails to provide sufficient water. However, by relaxing the constraint so that, on average, EO can be achieved ($\sum AR_{EM,k}\pi_s = EO\pi_s$), there can be greater flexibility in meeting all objectives. (i.e. institutional and environmental) Alternatively, the model could be set so that, 95% of the time, total objectives could be achieved $\mathrm{VaR}_{0.95}\left(\sum EO_{k,s} \leq \sum AR_{EM,k,s}\right)$, providing the EM with capacity to sacrifice some objectives to maximise national benefits. However, both relaxations imply that environmental and social welfare is traded-off against increased private economic returns. For example, where river base flows[8] may not be fully protected, and wu_{AG} could place significant downwards pressure on wf, then to meet $\widehat{wf}$ and $\hat{\sigma}$ gaols the EM would need to allocate a greater number of AR_{EM} to maintain $wf > 0$, and reduce σ.

[8] Base flows are the primary source of running water in a stream during dry periods and may be derived largely from groundwater discharges. Ecologically, base flows may provide sources of water for isolated instream habitat maintenance, and/or sustain some connectivity between those habitats. They can thus be considered a minimum ecological requirement in most systems.

If the EM has sufficient water to meet EO with their existing portfolio of rights and technology, then questions concerning efficiency and social expectations can be further explored. Efficiency can be encouraged by the adoption of three management options discussed below. However, social aspirations for the EO should not be assumed to remain constant, especially as climate change is expected to reduce θ. However, if the EM has insufficient water to achieve the EO, and assuming that the EO are not reduced below social aspirations, then the EM must adapt their strategies to cope (Horne et al., 2018). For this article, the EM adaptive management strategies are: carryover (i.e. save surplus water in one state to irrigate the environment in another state); trade to either reallocate resources permanently or opportunistically trade water on a spot (seasonal) market; and/or invest in new technology to change the vector of EO requirements (CEWO 2015). These three strategies are also available to, and utilised by, farmers.

The role of carryover

Carryover negates the 'use it or lose it' mentality often identified with seasonal water allocations, and provides individuals with a risk management strategy of leaving unutilised water in storage for use in subsequent seasons. A penalty function (α) can be applied to water left in a storage, and Equation 17 can be altered as follows $(\sum wu_k\pi_s \leq \sum AR_k\pi_s\,\alpha_s)$ to explore hat option. Then, water use on average will be less than or equal to the volume that the rights provide. While α acts as a disincentive to save water, the real value of this water is revealed, if the subsequent season's allocation fails to provide sufficient rights to meet any fixed water requirements (i.e. to avoid irreversible losses). See Adamson et al. (2017) for greater details on this outcome.

Water trade

Water trading is a key risk minimisation strategy to mitigate water supply deficiencies (Loch et al., 2012). Trade can be divided into both allocation trade (i.e. trade that occurs on an annual basis, which is akin to renting the resource from its owner), and entitlement trade (i.e. trade where the ownership of the right changes permanently). The model described here can be set to allow ($\sum wu_{AG} + \sum wu_{EM} \leq \sum AR$) or constrain trade ($\sum wu_{AG} \leq \sum AR_{AG}$, $\sum wu_{EM} \leq \sum AR_{EM}$). However, the price of water (wp) in any market is dependent on the demand associated with alternative commodities (perennials versus annuals), and the realised supply of water (Adamson et al., 2017). Note also that wp will also be impacted by relative elasticity of supply and demand (Randall, 1981). Therefore, water markets can provide both a source of income and transaction costs for the EM (recalling that these costs can be included in Equation 10, or set as a new equation providing a budgetary limit).

The introduction of trade provides capacity to encourage environmental watering efficiency, or a representation of EM market engagement to purchase/sell water to meet their objectives. If water is transferred from EM to AG, more water can be used to irrigate commodities and the income from that transfer can be utilised by the EM to meet other objectives and/or explore other positive externality activities. A simple adjustment to the variable cost equation helps track the costs of purchasing water from the EM. Conversely, any water transfers from AG to EM suggest the EM has insufficient water to meet their objectives, and that either the EM may need to engage in entitlement (additional permanent AR) trade or change their long-run environmental objectives. However, as the model incorporates conveyance losses in a directed flow network, synergies are possible. Thus, the direction of trade may switch as the model optimises water use between AG to EM and it flows through a basin.

Importantly, uncertainty associated with climate change necessities an assumption that, what is optimal now, may not hold in the future. Therefore, any reallocation of water needs to consider climate change impacts on the reliability of the water portfolio and the environment's future demand. See (Adamson, 2015) for a basic introduction of a how a budgetary constraint can be developed to model a permanent reallocation of the EM water portfolio with respect to a changing climate. The use of carryover and trade, then helps compare the returns and risk from trade versus having surplus water in a dam (or other storage).

The wp revealed via trade can provide a proxy of the social value of natural capital $\left(\sum g_{s,k} \times wp_{s,k}\right.$ [9]$\left.\right)$ and environmental output $\left(\sum h_{s,k} \times wp_{s,k}\right)$ can be approximated by state of nature. Consequently, this approach may overcome some of the limitations of studies not valuing water by state of nature discussed above.

Incorporating water saving technology

Water-use efficiency (saving) technology in farm production systems are the cornerstone of current world policy dealing with water reform, despite the fact that the limitations of 'saving' water for increased environmental benefit are well noted (Adamson and Loch, 2014, 2018; Molle and Tanouti, 2017). The potential for water savings from altered approaches to irrigating the environment however, are slightly different. As discussed, typical river systems are highly modified, and this includes some key environmental wetlands where extensive capital works have been undertaken to irrigate the environment with less water (MDBA 2017). By placing a strict constraint within the model to meet only those EO required in a given period, potential (unintended) positive externality gains from any unused water (return flows to the river system) will not feature in the results. However, the reduction in return flows is still a factor that must

[9] In this case $wp_{s,k}$ is the price of water in the permanent water market.

be accounted for by re-examining the impact that $v_{k,s}$ has on wf. Under this scenario, additional costs of achieving environmental production must be expected.

But it is the way this article has redescribed production as, $z_s \leq \zeta_s + g(\boldsymbol{x_s})\varepsilon_s + h(\boldsymbol{x_s})\varepsilon_s$, that raises questions associated with how environmental watering efficiency is achieved. With this expression, we can re-examine the error term, as well as water required by g and h. If g is the binding constraint, then the EM must consider whether water-use efficiency targeting this term would provide greater benefits than an overall reduction in water inputs? That question is for a subsequent article and provides a rich area of future research. Further, constructing (and relying on) infrastructure networks to irrigate the environment creates institutional and environmental lock-in transaction costs (Pittock et al., 2013), where we only irrigate those environmental assets that have pipes and channels. This is also a rich area of future research and discussion. Ultimately though, the combination of all management techniques above highlight a necessity for the EM to adopt flexible arrangements for dealing with future supply variability and uncertainty.

Wider discussion

The objective of this paper was to explore if we can make a better model to understand the relationship between water users in a basin who have the same set of water rights. By separating water into two separate roles, water for the preservation of capital (natural or financial); and water to produce outputs, insights into state descried management options and potential increased market competition can be explored. The use of a flow network helps understand the opportunity costs that occur within a basin and how changes to: water use patterns; future descriptions of states of nature; or frequency of states of nature, are likely to place a bigger burden on the environmental manager depending on how they are tasked to achieve salinity and flow targets.

Variable and new rainfall patterns (Chiew et al., 2011) will alter conjunctive water supplies (Loáiciga, 2003). This variation in the conjunctive water supply will stimulate new strategies to managing and allocating scarce water resources (Lopez-Gunn et al., 2012; Noel et al., 1980; Scheierling et al., 2006). However, these new water management strategies can have unintended consequences on supply (Rodríguez-Díaz et al., 2011; Ward and Pulido-Velazquez, 2008) and water quality (Brouwer and De Blois, 2008; Dellink et al., 2011). As these authors argue, this may create perverse outcomes where an EM only irrigate the environmental assets they control. This combination of management outcome and infrastructure investment may create institutional lock-in that may lead to reduced ecosystems resilience under a changing climate, and may be difficult to represent in any model given current measurement limitations in the transaction cost discipline (Loch et al. 2018).

Further, while the market price of water and the elasticity of water demand and supply by state of nature can be determined, those prices only provide a proxy for societies value of environment, as the true value of ecological water may be far greater than those revealed in the market (Baumol and Bradford, 1972). In particular, diminishing marginal returns need to be better understood and incorporated into the analysis outlined here. However, while this model doesn't quantify environmental welfare gains, those welfare gains are embodied once the environmental objectives are achieved. Provided that the environmental objectives are consistent with social expectations, welfare increases.

Finally, while this paper doesn't explore the optimal quantity of water to be returned to the environment, it does provide insights into the limitations in current thinking about environmental manager efficient decision-making. Efficiency works where uncertainty doesn't exist (Horne et al., 2018). When uncertainty is present, flexibility and underutilised resources help deal with adverse events. Rent seeking is evident when you are attempting to make one individual do more with less and change their risk profile without changing your own. In a coarse sense, this model has the capacity to provide some useful insight around these issues.

Limitations in the approach

This model, like all models, is designed to tell a story. It's designed to explore trade-offs, the sensitivity of variables, management options and policy shifts. The model in its present form has three major interrelated limitations: the assumption that the environmental manager has perfect knowledge; our knowledge of how future input and output sets under a changing climate is incomplete; and the model's annual design.

It is unrealistic to expect the environmental manager to have perfect knowledge. The long-term success of the EM will depend on the EM's ability to recognise each state of nature and apply a suitable management strategy to deal with water supply under uncertainty. But the ability of the EM to act and learn will depend on: realised water availability; the binding environmental objectives; when those objectives/goals must be met (e.g. by state or on average); any political limitations placed on their suite of management options; budgetary limitations; and hydrological realities. The environmental manager needs time to explore and learnt how to manage resources, but this leaning needs to be by state of nature (Goldstein and Gigerenzer, 2002). Asking for perfection and efficiency gains now, will be detrimental to long term national welfare benefits.

Complicating the efficiency story is the fact that the climate is changing. We can't expect an individual to have a complete understanding of all and future realised states of nature, and the outcomes from alternative responses to those states of nature. In other words, we are aware that the future may reveal states or descriptions of those states of water supply that individuals have not experienced. While learning from these events is possible, and adaptation can be swift, it needs to be ecologically rational (Goldstein and Gigerenzer, 2002). Any deviation away from these principles will likely mean that the EM will fail in meeting their objectives, and the requirements to meet fixed environmental objectives will exasperate the notion of failure. Further, even if we assume that some water in the river system is not diverted illegally, and that we have perfect knowledge about rf and v, water users may still choose to increase or decrease their area under irrigation, change their total demand for water resources by g and h by k and s, and their attitudes to risk. These changing parameters will continually force the EM into adaptive responses. If the EM has surplus water the last thing they should do is permanently trade away the surplus. Caution though is needed. Should farmers assume that trade will always be in one direction, they will become exposed if the EM enters the market and buys water—especially in times of drought.

The annual approach of the SCA model assumes that all catchments are concurrently experiencing identical states of nature. Consequently, the good and bad states provide a theoretical worst case and best case respectively. While this helps explore the tails of the distribution, it negates some management solutions where good seasons in one part of the basin may offset bad states in different catchments.

Conclusion

Welfare gains from re-allocating water both in terms of equity and total consumptive use are possible. However, the restoration of flows to the environment poses complex problems, and when rights are reallocated in the presence of compensation, rent seeking emerges. The use of models like that described herein are needed to assess and/or justify the net welfare gains from reallocation of water. However, the value of the model in that discussion is only useful if the trade-offs between all water users are understood, as well as the impact that those outcomes have on the river system.

The division of water inputs into separate roles for keeping capital alive and achieving production outputs expands our thinking about how and why we use water, and the risks posed to *both* private and natural capital when water is not available. This insight forces decision-makers to consider how policy incentives may transform behaviour and input use at the local scale (per ha) and/or the industry scale, creating feedback problems (hydrological and risk attitudes). Such structural changes create irreversible outcomes if the policy fails to correctly predict behavioural changes to the inherent non-convexity that arises when uncertainty determines the supply of the input in question.

If policy solutions create irreversible outcomes, then total welfare is reduced. However, by treating the environmental manager as just another user, and reducing environmental objectives in an economic model to: i) applied water, river flows and water quality; and ii) exploring reactions to key management options such as carryover, trade and water-use efficiency, we may be able to examine with greater clarity the role of water supply risk and uncertainty for all users within a river basin. Further, by modelling all water-users together, synergies can be explored to examine how management options can be used to reduce that risk. Importantly, the key welfare risk (i.e. prevention of irreversible welfare losses) can be determined in the model by examining if there is sufficient water to meet fixed water requirements for all water users—inclusive of the river flows.

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