Emerging water and carbon market opportunities for environmental water and climate regulation ecosystem service provision

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Abstract

Markets are increasingly part of government, non-government, and private business provision of public environmental interests. Key examples include carbon credit markets and environmental water markets. Market demand for carbon credits from sequestration are expected to expand in size and geographic scope as a result of climate action obligations and increased carbon credit tradability provisions in the Paris Agreement on climate change. Market based reallocations of water are also increasingly common. The increased use of markets for multiple and related environmental good provision will inevitably introduce synergies and risks in joint ecosystem service provision. This study assesses water and carbon ecosystem service supply potential for a joint carbon and water market participation strategy using a case study of the lower Murrumbidgee, in the Murray-Darling Basin. The methodology is a dynamic hydro-economic simulation of river flows, floodplain inundation, forest carbon dynamics, carbon credit value, and water opportunity cost. The study results indicate possible synergies in joint provision of carbon sequestration and environmental flow benefits through a carbon-water trading strategy. This involves funds for environmental water purchases generated through sale of carbon credits from improved floodplain conditions. Results identify limited trading opportunities at the current carbon price (AU$13/t$CO_2$), resulting in an economically viable re-allocation of 2.31GL/year (0.1% of water currently diverted for irrigation) to the environment with frequent years of zero re-allocation. At prices above AU$20/t$CO_2$, there may be additional trading opportunities and as much as 5% of current irrigation diversion was predicted to be reallocated at AU$100/t$CO_2$. While the results are particular to the case study, the conclusions discussing policy design challenges related to realizing effective environmental improvements in interacting carbon and water markets are relevant to many water catchments globally.

Key Words: ecosystem services; environmental water; carbon markets; water markets; Murray-Darling Basin; hydro-economic modelling.

Highlights

- Potential to finance water purchases through generation and sale of carbon credits is demonstrated
- Achieved by allocating water to environmental use to stimulate floodplain carbon uptake
- Results are sensitive to carbon and water prices, foresight, and future climate
- Challenges to implementation include additionality, leakages and sequestration impermanence

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1 Introduction

In many river basins around the world water is fully allocated for consumptive purposes and flow-dependent ecosystems are subsequently degraded (Grafton et al., 2018). Wetlands have disproportionately high carbon sequestration capacity which translates into high economic value for carbon storing and crediting (Patton et al., 2015). However, wetlands are particularly vulnerable to degradation (Davidson, 2014; Settre and Wheeler, 2017) and their declining health can erode sequestration capacity and ecosystem services (Banerjee et al., 2013). Conversely, when wetlands are preserved and restored, they can generate extensive ecosystem services (Bark et al., 2016) including global benefits from carbon storage and climate change mitigation (Patton et al., 2015). Water markets are increasingly used to reallocate water from consumptive to environmental uses to restore wetlands and floodplains or prevent continued degradation. Examples include water reallocation programs in the Columbia River Basin (Garrick et al., 2011), California (Howitt, 1994) and the Murray-Darling Basin (MDB) in Australia (Wheeler et al., 2014).

Carbon and water policy in the MDB is relatively unique is it is a place that has formal, low transaction cost markets for both water and carbon. The MDB water market is currently being used by government actors and environmental non-government organizations (NGOs) to reallocate consumptive water to the environment (Lane-Miller et al., 2013; Grafton and Wheeler 2018; Haensch et al. 2019). Additionally, Australian climate policy includes options to generate potentially tradable carbon credits through active forest and bush management through the Australian Emissions Reduction Fund (ERF). The ERF incentivizes emissions reduction activities across the Australian economy through competitive tenders for payments for actions to reduce or offset emissions. One ERF method pays landholders for the generation of carbon credits by planting new forests or actively managing existing forests to promote carbon uptake and prevent carbon loss (DoEE, 2016).

This study explores possibilities to improve ecosystem service outcomes through governments or NGOs trading in both the carbon credit and water markets. Overbank floods to water-stressed floodplains can improve vegetation condition and stimulate carbon uptake or prevent carbon decay. The existence of water and carbon markets potentially allows for the delivery of strategic floodplain inundation in a pattern which can generate biomass growth and carbon credits of sufficient dollar value to offset the cost of environmental water purchases on water markets required to cause the inundation.

This study adds to the literature examining the potential trade-offs and synergies between provisioning (e.g. agricultural water use) and regulating (e.g. carbons storage) ecosystem services through integration of hydro-ecological and economic principles (Harou et al., 2009; Momblanch et.al, 2016; Settre et al., 2017). Key integrated models representing carbon tradeoffs include Kim et al. (2018) who develop a process-based hydrological model to assess the economic trade-offs for global carbon benefits relative to loss of landscape level provisioning services. Triviño et al. (2015) use multi-objective optimization to find the optimal forest management for carbon and timber services. Patton et al. (2015) integrate spatial estimates of carbon storage in US wetlands and the social cost of carbon to determine an average global carbon value of US$2,800/ha of wetland. Taken as a whole this past research highlights the importance of carbon storage valuation in modelling decisions of land and water allocation and the necessity for more holistic ecosystem service modelling (Monblanch et al., 2016).

While there has been extensive hydro-economic modelling in Australia's MDB (see Settre et al., 2017 for a review), it has primarily focused on agricultural production and economic impact of reallocating water from agriculture to the environment (e.g. Grafton and Jiang 2011). There has also been considerable work studying the economics of MDB water trade (e.g. Qureshi et al., 2013), climate change (Adamson et al., 2009) and the integration of hydrological, biophysical and economic value using stated preference survey results (Akter et al., 2014). In addition, there is a growing body of literature assessing the benefits of public and private environmental water holders (EWHs) using and trading water entitlements (permanent water rights) and/or water allocations (temporary water rights) for environmental outcomes (Wheeler et al., 2013). For example, key studies by Kirby et al. (2006), Ancev (2013) and Connor et al. (2013) evaluated EWH potential to trade temporary water for the environment. All three studies find that flexibility introduced by temporary water trade provides opportunities to raise funds from leasing water to irrigators in times of scarcity and using proceeds to finance water purchases at other environmentally critical periods. Set within this context, the key novelty of this research is a hydro-economic assessment of opportunities to realize more multiple public good ecosystem service provision by strategically trading in carbon credits and water markets.

2 Materials and methods

2.1 Case study: the Lower Murrumbidgee

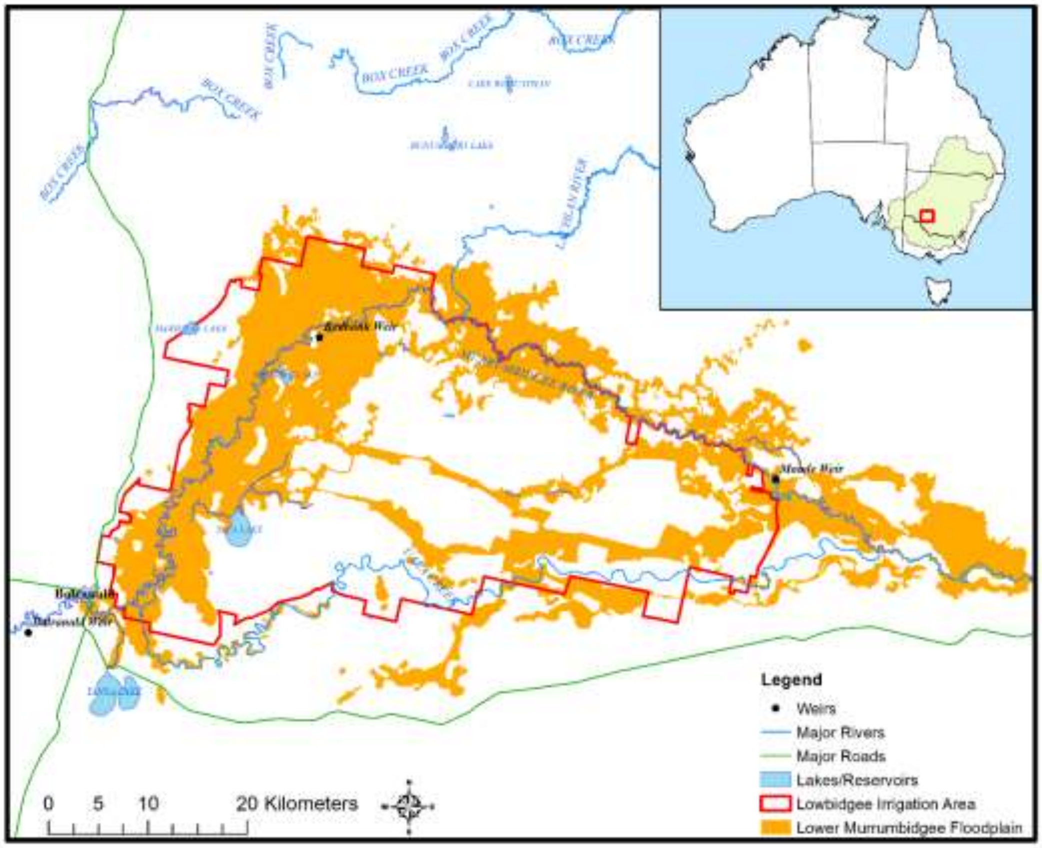
Globally between 64–71% of wetlands have been lost since 1900 with continued high rates of loss across Asia and Africa (Davidson, 2014). In Australia, wetland loss and damage is extensive (Kingsford, 2003; Settre and Wheeler 2017) and restoration efforts are ongoing (Bark et al., 2016). Our study focuses upon a remaining, but degraded, floodplain wetland area in the southern MDB called the Lower Murrumbidgee (the Lowbidgee, Figure 1). The Lowbidgee floodplain is in the Murrumbidgee catchment, covering 8% of the MDB but accounting for 22% of MDB consumptive water diversions (CSIRO, 2008). Irrigated agricultural development occurs along either side of the Murrumbidgee River (Wen, 2009), where the primary crops are annual cereals for grain, including rice, wheat and millet. The Murrumbidgee River is 1,600km in length and flows westwards to a confluence with the River Murray. The Lowbidgee floodplain is in the downstream reaches and covers 51,535 hectares (MDBA, 2012). It is a particularly good example of a floodplain River Red Gum (*Eucalyptus camaldulensis)* forest (Kingsford, 2003). Large-scale catchment modifications have depleted the volume and variability of instream flows to the Lowbidgee and have resulted in considerable floodplain damage and an opportunity for ecological restoration (Fraizer and Page, 2006).

There is an active water market in the Murrumbidgee catchment for water entitlements (permanent water rights sale) and water allocations (annual/temporary water rights). Most of the water transactions in the MDB are water allocation trades and this is the water market considered in this study. Water allocations prices vary considerably and are driven predominately by water scarcity factors (Wheeler et al., 2014).

There is also potential to sell carbon credits that result from land use and management changes through the national emissions reduction payment scheme (ERF). Projects can be proposed to reduce emissions or offset emissions with increased carbon sequestration. Projects that can produce abatement eligible for credits include energy efficiency, low carbon electricity generation, electrification and fuel switching options. Most relevant to this study is the assisted natural vegetation regeneration where a land management change can generate carbon credits. In each auction round bids are solicited where proponents propose abatement activities consistent with ERF rules and the credit price at which they are willing to undertake the activity (Clean Energy Regulator, 2018). Across seven ERF auctions a total of AUD$ 2.45 billion has been committed to achieve 192 Mt CO_2-e abatement at an average price of AUD$11.97 $tCO_2\text{-e}^{-1}$ (Evans, 2018). Of this, land use and management change activities were by far the most funded source of ERF credits with 65% (125.5 Mt CO_2-e) of the total abatement secured through forest-based methods (Clean Energy Regulator, 2018).

This study models the potential for an entity with environmental objectives, for example an environmental water trust, to act in water and carbon markets in the Lowbidgee floodplain area. We model the possibilities to supplement environmental flows with water markets purchases and finance the costs with carbon credits resulting from improvements in floodplain tree carbon sequestrations. The boundaries for the study are defined as the extent of the River Red Gum floodplain population from the upstream water management point at Maude Weir to the confluence of the Murrumbidgee and Murray Rivers.

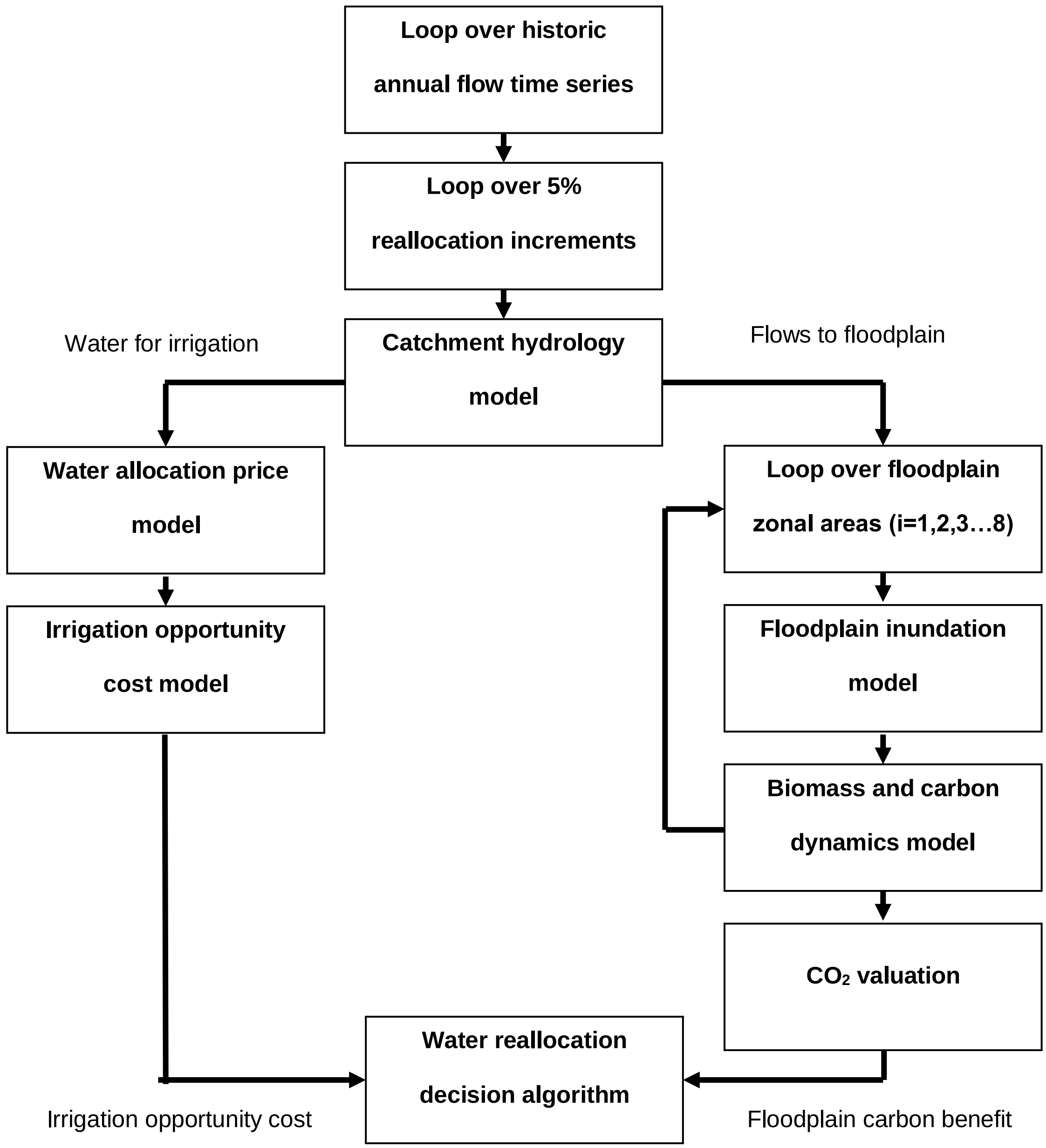
Figure 1 The Lower Murrumbidgee Floodplain, Australia



2.2 Methodology framework

A dynamic hydro-economic simulation model developed for the Lowbidgee wetland consists of four sub-models, namely: a) catchment hydrology model; b) floodplain forest and carbon growth and decay model driven by floodplain inundation; c) economic valuations of environmental and irrigation opportunity costs; and d) a water reallocation decision algorithm. A model schematic is shown in **Error! Reference source not found.**. The model development was guided by the ecosystem services framework which seeks to link biophysical sciences with economic value and human institutions (Braat and de Groot, 2012; TEEB, 2010). The model was programed in the General Algebraic Modelling System (GAMS). Dynamic simulation was chosen as an appropriate solution choice given the many integrated elements, time-steps and spatial units which pose optimization difficulties.

Figure 2 Hydro-economic model



The aim of the model was to simulate incremental reallocation of water from irrigated agriculture to environmental use (i.e. floodplain inundation) and identify a discrete annual reallocation volume for which the marginal benefit of reallocation equals or exceeds the marginal cost of reallocation.

The simulation model began with a catchment hydrology network node model calibrated for the Lower Murrumbidgee. The hydrology model computed irrigation and environmental water availability assuming current water-sharing rules and irrigation development levels. Annual environmental water availability drives the inundation modelling which computed the frequency and extent of inundation for discrete floodplain zonal areas. The inundation modelling results were input to a floodplain carbon dynamics model built using locally calibrated floodplain River Red Gum carbon (C) stock growth and decay functions which respond to water deficits and overbank floods. Because the dominant agricultural activities in the Murrumbidgee are annual cereals and rice, which do not have high long-term potential for vegetation carbon sequestration (with the exception of soil carbon, see Rajkishore et al., 2015), declines in agricultural carbon stocks when water is reallocated from agriculture to the environment is not accounted for. This is further discussed later on.

Modeled additional floodplain carbon was converted to CO_2e and valued as carbon credits using exogenous carbon price levels chosen to represent the Australian market equilibrium price and the global social-cost of carbon (SC-CO_2) for a range of emissions and discount scenarios. A regression-based water cost model was used to relate historic water availability to changes in temporary water prices, which is influenced by annual water reallocation decisions. The regression model was used to estimate the opportunity cost of irrigation water use.

The water reallocation decision algorithm compared the marginal benefit of reallocation (i.e. the value of additional carbon credits generated by inundation) to the marginal cost of reallocation (i.e. the purchase cost of the water required to cause the inundation). The decision algorithm selected a discrete temporary water allocation volume that maximizes the expected additional carbon achievable such that the marginal carbon benefits of reallocation are equal to or greater than the marginal water costs of reallocation. The modelled water reallocations occur for a period of one year and are then returned to the consumptive pool, comparable to a temporary water sale. The final step tested outcome sensitivity to key outcome drivers. As described in section 2.12, this includes varying carbon price, water availability changes consistent with climate change, and foresight horizon (ability to consider possible future states of water availability).

2.3 Catchment hydrology model

The hydrology sub-model is an annual water balance for the Lower Murrumbidgee catchment and is a component of a whole-of-basin hydrological model developed in Kirby et al. (2013). The water balance represents inflows, two dam nodes, consumptive water demands, losses, dam storage and spill, baseline environmental and irrigation water supply, and annual water reallocation volumes. Reservoir inflows, storage and outflows within the hydrological simulation ran over 113 years using the historic climate sequence (1896-2008). Water available to inundate the floodplain is the annual river outflow, which is the sum of dam outflows and catchment runoff less irrigation abstraction, and dam and channel losses.

Water available to inundate the floodplain is altered by the yearly reallocation decisions of the modelled EWH. Reallocations were simulated each year by reducing irrigation water extraction by 5% increments and increasing flow to the floodplain accordingly, bounded by zero and 50% reallocation volumes. The desired flood volumes and return intervals required to meet the environmental requirements of the floodplain River Red Gum forest are shown in Table 1. As shown, higher total catchment inflows relate to higher return intervals and a larger area of floodplain inundation.

Table 1 Desired instream flow volumes in the Lower Murrumbidgee

Flowband *(i)*	**Total inflow to the Lower Murrumbidgee floodplain (GL/year) $(ER_{(i)})$**	**Floodplain inundation return interval (years) $(EF_{(i)})$**	**Area of River Red Gum inundated (ha) [% of total area inundated] $(A_{(i)})$**
1	50	0.95	1,073.6 [2%]
2	100	1.10	2,684.1 [5%]
3	170	1.33	5,905.1 [11%]
4	270	1.43	9,662.9 [18%]
5	400	1.67	13,420.7 [25%]
6	800	2.00	25,231.0 [47%]
7	1700	4.00	45,093.7 [84%]
8	2700	6.67	51,535.6 [96%]

Source: adapted from MDBA (2012; p.13).

The model simulates an EWH who can alter the frequency and magnitude of overbank flooding by annually reallocating water when it is economically justifiable to do so. Annual reallocation volumes are described by the variable $et_{(t,c)}$ (Equation 1):

$$et_{(t,c)} = \left(\frac{c}{20} - 0.05\right) iw_{(t,c)} \quad \textbf{(1)}$$

Where c represents a discrete reallocation portion chosen each year by the modelled EWH and takes values one to eleven in 5% increments representing reallocation between zero and 50% of water initially supplied to irrigation. $Iw_{(t,c)}$ is the irrigation water supply prior to any reallocation exogenously determined from the catchment water balance with current allocation rules.

2.4 Floodplain carbon dynamics model

A stock-and-flow model of annual potential vegetation carbon storage was developed based on the biophysical condition of the floodplain River Red Gum forest in response to overbank flooding and water scarcity events, the driving force of forest productivity (Junk et al., 1989) and carbon assimilation (van der Molen et al., 2011). The model also represents how in the absence of the required water availability, forest productivity (Doody et al., 2015) and carbon sequestration rates degrade (Chaves, 1991).

To simulate this, the floodplain is divided into eight zonal areas, $A_{(i)}$, each of which should be inundated by a flow $ER_{(i)}$ at a desired return period of $EF_{(i)}$ years if it is not to pass into a state of poor condition and carbon decay. For example (referencing Table 1), $ER_7 =$1,700GL is the flow level required to create an inundation of floodplain zone seven which has an area of 45,093ha. It is required at return interval $EF_{(7)} = 4$ years to maintain forest productivity and carbon accumulation (as opposed to a decay in carbon stock due to water deficit).

The carbon dynamics model is described using four ecosystem states representing floodplain productivity and carbon storage potential: i) growth; ii) decay; iii) maximum carbon stock equilibrium; and iv) minimum tree-death equilibrium. Transition between states are modelled as functions of the severity of hydrological disturbances represented with time-dependent iteratively updated flood and drought counter variables (i.e. a high value for the drought counter describes a more severe drought). The counter iteratively sums the number of consecutive years that a flood volume for each floodplain zonal area exceeds the desired return interval for each flowband. The counter reverts to zero when a flood of desired magnitude and frequency is delivered. Carbon growth dynamics are maintained for periods when the drought counter is less than or equal to the desired return period for each flowband, and dynamics switch from growth to decay when the desired return period is exceeded, subject to boundary conditions on maximum/minimum carbon volumes in equilibrium states.

2.5 Modelling maximum and minimum carbon storage capacities

The maximum potential carbon storage volume per hectare *(*$MaxCarbonStorage_{(i,t,c)}$*)* is the total possible volume of carbon stored per hectare of mature, non-water stressed River Red Gum forest derived from in-field sampling conducted in a comparable MDB floodplain forest (Smith and Reid, 2013). This was the volume of carbon stock assumed at the start of the simulation ($t=0$) (Equation 2).

$$MaxCarbonStorage_{(i,t=0,c)} = 104.4tCha^{-1} \quad \textbf{(2)}$$

The minimum volume of carbon stored *(*$MinCarbonStorage_{(i,t,c)}$*)* occurs when an area of forest has decayed to a dead-tree equilibrium at which point carbon uptake is zero. It is greater than zero because carbon remains trapped in woody biomass and coarse litter of dead trees. The minimum carbon volume for River Red Gum is sourced from Smith & Reid (2013) (Equation 3).

$$MinCarbonStorage_{(i,t,c)} = 4.7\ tCha^{-1} \quad \textbf{(3)}$$

2.6 Modelling carbon stock decay in response to water deficit

The annual increment of carbon decay for each reallocation proportion, *c*, relative to the baseline scenario (i.e. no reallocation, $c=1$) is governed by Equation 4. Equation 4 represents *avoided* carbon decay from reallocation as a carbon benefit expressed as difference in smaller carbon decay rate expected with reallocation compared to baseline condition (more rapid decay).

$$AvoidedCarbonDecayIncrement_{(i,t,c)} = e^{vEC_{(i,t,c=1)}} - e^{vEC_{(i,t-1,c)}} \quad \textbf{(4)}$$

Where $AvoidedCarbonDecayIncrement_{(i,t,c)}$ is the annual incremental volume of carbon decay per hectare of River Red Gum forest, v is the decay constant, $EC_{(i,t,c)}$ is the number of sequential years past the desired return interval for floods in flowband, i, for each reallocation proportion, c. As $EC_{(i,t,c)}$ increases, the incremental decay in carbon storage increases at an exponential rate, such that decay in the first year of drought is less than decay in the tenth year of drought, as is consistent with observed ecological tipping points during water stress (Banerjee & Bark, 2013). Absent a monitoring history of sufficient length required to specify the period of time until tree death occurs, we calibrate the decay model to the rule of thumb that ten years past the desired flood return interval and below average rainfall will likely cause River Red Gum mortality of any aged tree. This is implemented by choosing decay constant of $v=0.31$ to reflect that after ten years past the desired return interval (i.e. $EC_{(i,t,c)}>10$), per hectare carbon stock approaches the minimum capacity (Equation 3).

2.7 Modelling carbon stock growth in response to overbank flooding

Carbon stock growth and (re-)growth is modeled using the von Bertalanffy-Chapman-Richards (vBCR) forestry function to capture non-linear growth and sequestration rates (Zhao-gang and Feng-ri, 2003). This approach follows a precedent in modelling of carbon sequestration in native Australian Eucalypt species (e.g. Paterson and Bryan, 2012). The annual increment of carbon growth relative to the baseline scenario (i.e. no reallocation, $c=1$) is governed by Equation 5. Equation 5 calculates the annual growth increment that would occur with reallocation (e.g. higher growth rate) relative to the baseline scenario (e.g. slower growth), and therefore demonstrates the additional carbon benefit relative to the baseline.

$$CarbonGrowthIncrement_{(i,t,c)} = Aexp\left(-Be^{-kEB_{(i,t,c)}}\right) - Aexp\left(-Be^{-kEB_{(i,t-1,c=1)}}\right) \quad \textbf{(5)}$$

Where $CarbonGrowthIncrement_{(i,t,c)}$ is the annual increment of tree growth, used as a proxy for annual carbon uptake, A is the asymptote (i.e. maximum carbon storage capacity per ha), B is a calibration parameter and k is the tree growth rate. In the absence of detailed tree growth data for River Red Gums, the growth parameter for a slow-growing Mallee species (*Eucalyptus kochii)* was used as a starting value ($k=0.06674$) and the equation was iterated to find $k=0.1052$ for a calibration parameter $B= 3.1006$. These parameters were iteratively calibrated to match existing findings indicating approximately 80% of biomass is accumulated within the first 25 years of tree growth, after which point incremental change diminishes as the growth asymptote is approached. Equation 5 scales carbon storage growth to be proportional to tree growth up to asymptotic maximum storage per hectare, such that carbon uptake is assumed proportional to tree growth. $EB_{(i,t,c)}$ is the flood counter that iteratively sums the number of years the desired flooding volume and frequencies have been met. When $EB_{(i,t,c)}>0$, tree condition remains in the growth state and the critical threshold into decay has not been passed.

2.8 Net additional carbon

The net difference in carbon stocks per hectare relative to the baseline scenario is the sum of the additional increments of carbon sequestered or carbon decay avoided, as caused by incremental improvements in floodplain inundation as a result of reallocation, c, relative to that which would occur in the case of no reallocation. The net additional carbon stocks across the floodplain in each year was obtained by summing the carbon stocks per hectare by the total area of the floodplain zones, $A_{(i)}$, as shown in Equation 6. Equation 6 represents the increase in floodplain carbon stocks or prevented decay generated by incremental tree growth promoted by incremental increases in reallocation volumes, governed by parameter, c.

$$Net\_Additional\_Carbon_{(t,c)} = \sum_{i=1}^{8}\left(AvoidedCarbonDecayIncrement_{(i,t,c)} + CarbonGrowthIncrement_{(i,t,c)}\right) \quad \textbf{(6)}$$

In addition, an annual time-series of carbon dynamics is generated by expressing carbon stocks in each year as a function of carbon storage in the previous period, plus or minus incremental carbon growth and decay in the current period. The total volume of carbon across the whole floodplain in each year was obtained by summing the carbon stocks per hectare by the total area of the floodplain.

2.9 Economic modelling

2.9.1 Carbon valuation

Floodplain carbon stock and additional carbon generated through reallocation was converted to carbon dioxide (CO_2) using a conversion factor of 3.667 to be consistent with national carbon pricing systems (ERF, 2014). Assuming all net additional carbon is creditable, the dollar value of additional carbon stocks on the floodplain is given in Equation 7.

$$Additional\_Carbon\_Value_{(t,c)} = CarbonPrice\left(3.667 * Net\_Additional\_Carbon_{(t,c)}\right) \quad \textbf{(7)}$$

A range of carbon prices were modelled representing the social cost of carbon (SC-CO_2) and the recent Australian carbon market equilibrium price. SC-CO_2 is the global economic cost caused by an additional ton of CO_2 emissions being emitted and the amount that would be paid in a mature intergenerational market for carbon (Nordhaus, 2017). The estimated SC-CO_2 for 2020 (in 2007 US dollars) and for discount rates of 5%, 3% and 2.5% are US$12, US$42 and US$62 per t$CO_2$, respectively. Using a long-term conversion rate to AUD (1991-2018) (MacroTrends, 2018), this corresponds to approximately AU$15, AU$55 and AU$80 per t$CO_2$, respectively.

Recent carbon prices set through competitive ERF auctions in Australia run between 2015 and 2017 have established the current Australian market price of carbon to be between AUD$10.2 and $13.9 per tCO_2. The average price per ton of abatement in a recent ERF auction (December 2017) was AUD$ 13.08/tCO_2.

Temporary water price model

The water price model is based on Connor et al. (2013) and is obtained by regressing past average temporary water market prices on allocation levels in the Murrumbidgee catchment from 1996-97 to 2008-09. Modeled water price is shown in Equation 8. Water price varies with the volume of water available for irrigation $iw_{(t,c)}$ driven by system inflows and climate, as well as the volume of water removed from the consumptive pool and allocated to floodplain inundation, $et_{(t,c)}$.

$$WaterPrice_{(t,c)} = 1.754\left(2716 - 798log\left(iw_{(t,c)} - et_{(t,c)}\right)\right) \quad \textbf{(8)}$$

Where $WaterPrice_{(t,c)}$ is the temporary water market price (AUD$/ML), $iw_{(t,c)}$ is the volume of water available for irrigation, and $et_{(t,c)}$ is the volume of water allocated to floodplain inundation each year, as previously described. In simulation and assuming all else constant, $WaterPrice_{(t,c)}$ and the irrigation opportunity cost increases incrementally as more water is reallocated to the environment, thus providing an indication of the cost of irrigation water forgone and the increasing scarcity value of water.

2.10 Water reallocation model and treatment of foresight

The water reallocation model simulates the decisions of an EWH faced with the known water costs each year for each reallocation level and expected carbon benefits of water reallocation over the iteratively updated decision horizon. Accounting for inter-annual dependencies of environmental health is an important consideration in water resource decision-making due to ecosystem path-dependencies and tipping-points. Reallocation decisions should therefore be made with the understanding of how reallocating water in the present year will influence expected future carbon dynamics and carbon-water trade-offs. However, the future is not certain and may therefore be treated probabilistically.

Conceptually, the challenge of optimal allocation of available water in each year over a long planning horizon to irrigation and carbon generation could be thought of as an optimal control problem. However, it is unrealistic to assume the EWH operates with all the information and computational capacity required to compute a stochastic dynamic forward-looking optimization each period. It also becomes computationally intractable with the 113 year time horizon and multiple states of nature involved. Instead, a rolling horizon heuristic is used to model a forward-looking EWH concerned with the impact of reallocation decisions in the current period, t, on future simulation periods ($t+n$). In each period, the modelled EWH is assumed to understand the probabilities of all future state of nature across the decision horizon of t to $t+n$ years. Decisions to reallocate water are made based on the known water availability, water price and ecosystem conditions in the present year, t, and the expected value of additional carbon due to reallocation over a decision horizon from t to $t+n$. For each future year in the planning horizon in period, $t=t+1,\ldots,\ t=t+n,$ all possible values of environmental water availability, consumptive water availability, carbon values and irrigation costs are enumerated for all reallocation proportions, c.

The additional carbon value considered over the decision horizon, $t+n$, is given by the sum of the discounted flow of net additional carbon caused by reallocation for each year of the decision horizon. A discount rate on the future flows of benefits is $\delta=3\%$. The simulation continues to run in iteratively updated decision horizons until the end of the simulation, which occurs at T=113. The present value of the flow of possible carbon benefits generated by annual reallocation is given by:

$$CarbonValuePr_{(t,c)} = \sum_{t=0}^{n} \left(Additional\_Carbon\_Value_{(t,c)} \frac{1}{(1+\delta)^n} \right) \quad \textbf{(9)}$$

Where *t* is each simulation period, *n* is the number of years in decision horizon, δ is the discount rate, and $Additional\_Carbon\_Value_{(t,c)}$ is the dollar value of additional carbon generated in each simulation period. Equation 9 is iteratively updated over the length of the simulation, *T*.

The purchase cost of water required to cause the inundation is given by the volume of water reallocated, $et_{(t,c)}$, multiplied by the annual price of water reallocations, as shown in Equation 10.

$$IrrigOC_{(t,c)} = et_{(t,c)} WaterPrice_{(t,c)} \quad \textbf{(10)}$$

For each period, and accounting for the future flow of carbon benefits, the algorithm compares the cost of water and the floodplain carbon benefit corresponding to all reallocation proportions, *c*. The decision algorithm selects a discrete lease volume that maximizes expected additional carbon achievable such that the marginal carbon benefits of reallocation are equal to or greater than the marginal water costs of reallocation.

In simulation, there is a trade-off between precision and computational cost in expanding the decision horizon. Through experimentation we chose a horizon of three years (i.e. $t+n=3$), such that the modelled EWH considers the present year and two years ahead in the future when considering carbon benefits. A 3-year horizon was deemed to provide a good comprise between increasing computational burden and accurate convergence toward infinite horizon results.

2.11 Modelling scenarios and sensitivity

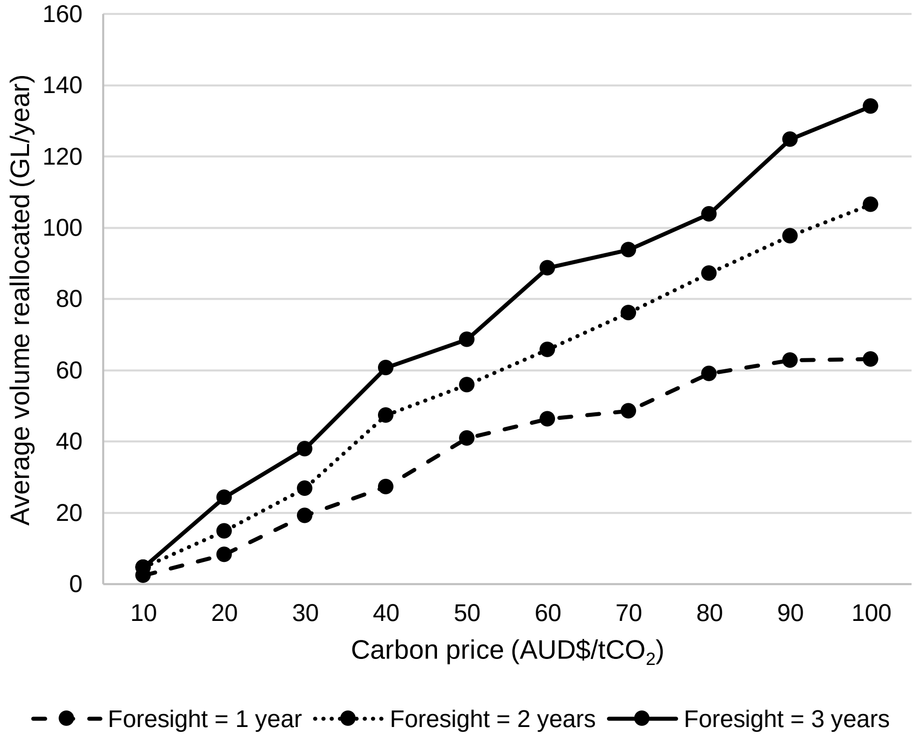
Due to the dynamic and integrated nature of the hydro-economic model, changes to one parameter invariably influence others and impacts the results. To assess the varying parameter conditions that could arise and their effect on the outputs, scenarios were run for three key variables: (i) carbon prices ranging between AU$10-100/t$CO_{2e}$ in increments of AU$10/t$CO_{2e}$; (ii) the number of foresight years (N=1,2,3); and (iii) future water availability (wet and dry future climate projections). A scenario for varying water prices was not run exogenously because water prices are iteratively updated through a regression-based calculation within the model which changes water price in response to varying annual water availability and environmental water re-allocation. The process for the scenario modelling involved running the model for one carbon price and holding all else constant, outputting the results, incrementally increasing the carbon price and re-running the model. This process was repeated for the number of foresight years and future water availability scenarios. This resulted in a total of 90 separate scenarios (i.e. 10 carbon price scenarios, three foresight scenarios, the current climate scenario and two future climate scenarios). The suite of output results for each scenario was then statistically and graphically analyzed. The scenario analysis also served to test the sensitivity of the model. Complex and comprehensive sensitivity analysis was beyond the scope of and unnecessary for the conceptual and demonstrative purposes of this research, although is identified for future research.

3 Results

3.1 Carbon financed water allocation

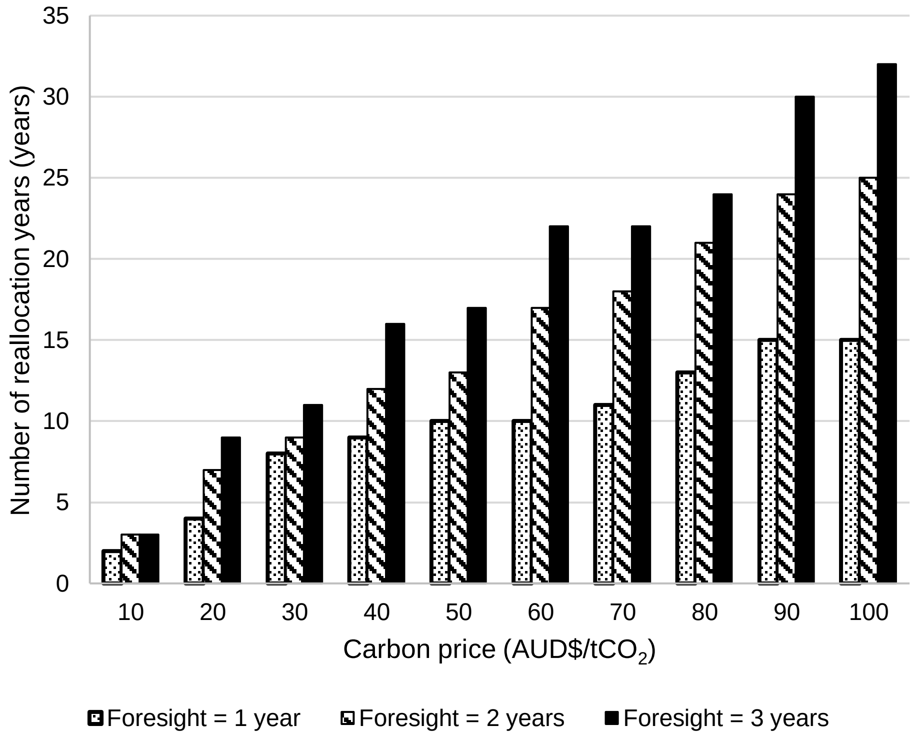
Results show opportunity to improve carbon stocks by reallocating water through the water market in a pattern which serves to prevent drought conditions in isolated low flow years and extend some moderate flood peaks. For the current market price of carbon (AUD$13.08/t$CO_2$) opportunity exists to economically finance water purchases of an average volume of 2.31GL/year with purchases at unit water price up to AUD$45/ML. However, these opportunities for reallocation occur in few isolated years over the historic climate sequence where water price is low (e.g. <AUD$45/ML). As carbon price increases and begins to exceed AUD$20/t$CO_2$, reallocation becomes increasingly more viable and the frequency and volume for reallocation increases (Figure 3 and Figure **4**). For the social cost of carbon (AUD$55/t$CO_2$ in 2020), the average economically justifiable reallocation volume is 72GL/year (8,228 GL over the simulation), with a maximum of 1,029GL/year and a frequent minimum of 0GL/year. This opportunity exists for water prices between AUD$20–274/ML, where the average unit purchase cost of water is AU$62/ML. For context, the 2.31GL and 72GL re-allocations represents around 0.1% and 3% of historic average annual water in the Murrumbidgee diverted for consumptive use (2,257GL/year) (CSIRO, 2008). Re-allocation volume for a carbon price of AUD$100/t$CO_2$ is approximately 5% of the consumptive supply per year on average.

Figure 3 Volume reallocated to inundate the floodplain for varying carbon prices and foresight years



Varying carbon prices also influences the number of years water is reallocated to the floodplain, as shown in Figure 4. Opportunities for economically justified reallocation occur more frequently as the modelled carbon price increases. As stated, there are very marginal opportunities to reallocate water at the current market price of carbon and opportunities existed in only 2% of the modelled simulation years. However, considering the social cost of carbon (AUD$55/t$CO_2$) reallocation is economically viable in between 9-16% of years, depending on the level of foresight considered in the algorithm (see sensitivity analysis). The frequency of reallocation reaches approximately one in four years for a carbon price of AUD$100/t$CO_2$ with a foresight algorithm of three years.

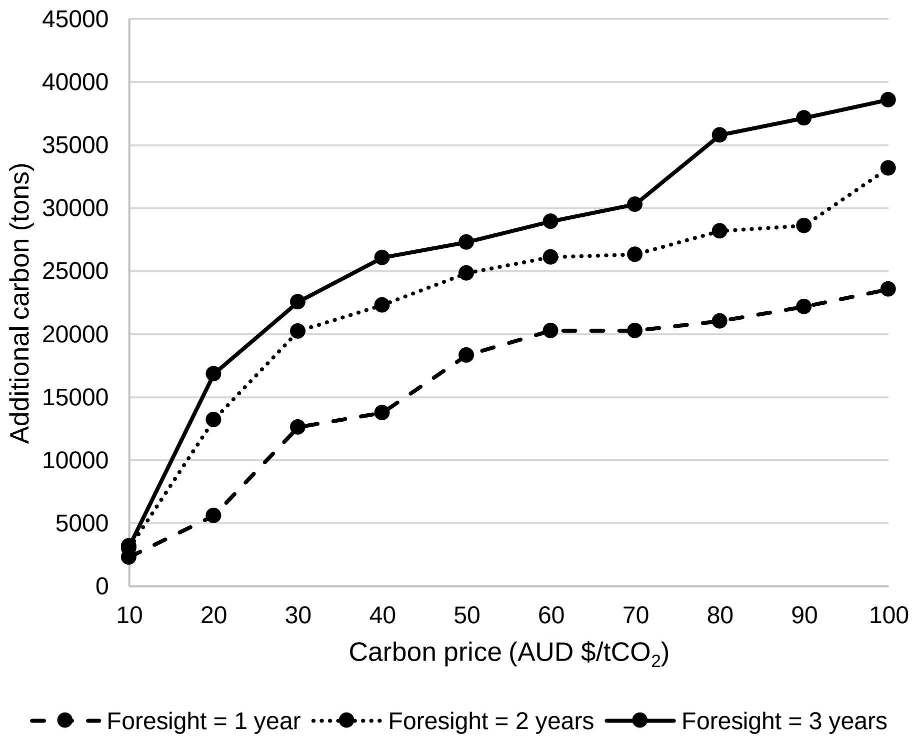
Figure 4 Number of reallocation years for varying carbon prices and foresight years



3.2 Impact of carbon financed water reallocation on floodplain carbon stocks

Error! Reference source not found. shows the modelled average additional volumes of carbon stored on the floodplain caused by reallocation volumes over the 113-year simulation. The observed difference in carbon stocks relative to baseline scenarios indicates carbon that has either been sequestered or prevented from decay that would not occur over the historic climate sequence and historic water extractions for irrigation without re-allocation.

Figure 5 Average additional carbon resulting from reallocation for varying carbon prices and foresight years



For the current carbon price (AUD\$13.08/tCO_2) and a foresight level of three years, the reallocation volume of 260GL over the modelling time horizon to the environment results in an average annual additional volume of 3,177 tons of carbon, an improvement of approximately 6% of the baseline mean carbon floodplain storage. The social cost of carbon scenario in contrast drove 8,228GL of reallocation over the simulation and an additional annual average 28,742 tons of carbon. To place these results in context consider that carbon offset estimated for carbon prices of AUD\$13.8, 55 and 100/tCO_2, equal 0.002%, 0.022% and 0.029% of annual NSW emissions, or 130.2 million tons of CO_2 in 2013-2014 (DoEH, 2014).

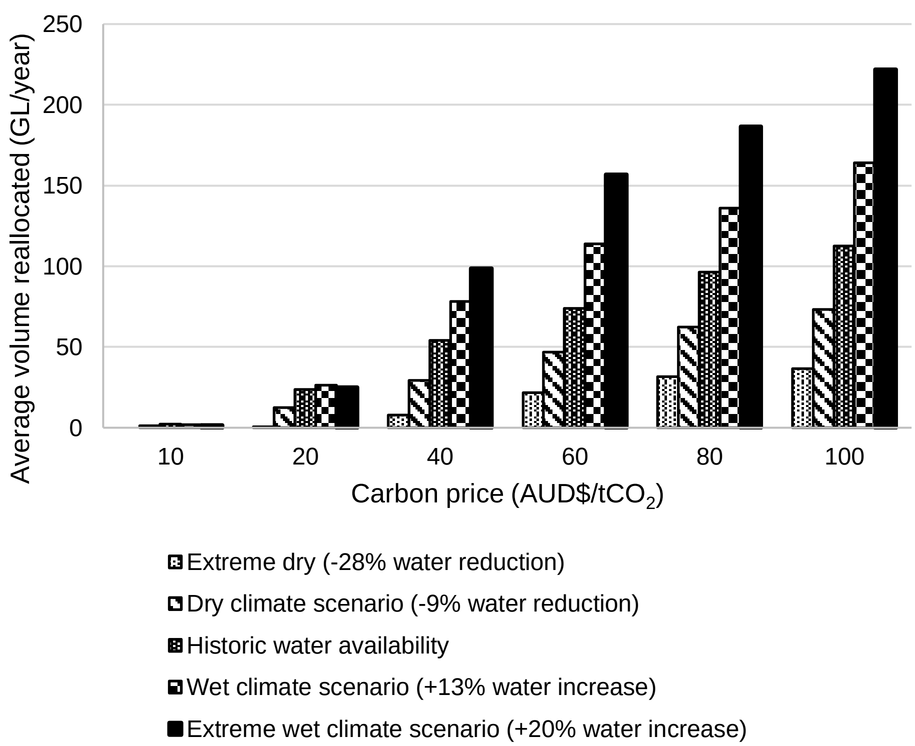
The increased volume of carbon storage relative to baseline is predominately derived by improved floodplain conditions on the lower elevation floodplain areas (flows of 50-800GL/year) which can generate additional carbon valuable enough to offset required water purchase even for low to moderate carbon prices. Marginal improvements in watering conditions are evident in higher elevation floodplain areas (flows of 1,700–2,700GL/year) only for high carbon prices (>AUD$70/t$CO_2$). However, these opportunities are not often viable given the prohibitive cost of acquiring the required volume of water to create these flows. In addition, the relationship between increasing carbon prices and additional carbon is non-proportional due to the non-linearity of the floodplain geomorphology, floodplain inundation response, and carbon growth curves. This is evident in Figure 4 which shows only incremental additional carbon for carbon prices between AUD$40-70/t$CO_2$, followed by an increase in additional carbon supply caused by higher carbon prices driving reallocation to larger floodplain zonal areas.

3.3 Sensitivity analysis

The results are sensitive to a range of uncertain parameters, most notably carbon prices, foresight years and climate driven surface water availability. The modelled ability to probabilistically account for future conditions affected both the volume and frequency of reallocation (Figure 3 and Figure **4**) and hence benefits in additional carbon volume (**Error! Reference source not found.**). On average, the reallocation algorithm using three foresight years allocates 1.7 times more water than the algorithm considering only one foresight year and 1.2 times more water than the algorithm using two foresight years.

Changes in surface water availability (e.g. rainfall) drive the temporary water price and have subsequent impacts for both the volume of reallocation and additional carbon value. Figure 6 shows the reallocation results for varying projected climate change scenarios (extreme dry: -28% water reduction, dry: -9% reduction, wet: +13% increase, extreme wet: +20% increase) and a foresight level of three years. For extreme reductions in water availability it is not economically viable to offset water costs for carbon prices lower than approximately AUD$30/t$CO_2$ due to increases in water prices driven by water scarcity. Under moderate water reductions there remains some marginal opportunities, albeit at lower volumes, typically 40% less than volumes viable under the historic climate scenario. Under the wet climate scenarios, the price of water is driven down, resulting in a greater number of low-cost opportunities for reallocation, with reallocations typically one and a half times greater than volumes reallocated under the historic climate scenario.

Figure 6 Volume reallocated to inundate the floodplain for varying climate scenarios and a foresight level of three years



4 Discussion

Some key findings from this study are similar to findings from the most related previous studies. Like Kirby et al. (2006), Ancev (2013) and Connor et al. (2013) who evaluated water trade to improve environmental outcomes, we also found that flexibility introduced by market re-allocation mechanisms provided opportunities to achieve environmental goals more cost effectively. The novelty of this research is in understanding of opportunities for environmental benefit provision from operating strategically in two markets (water and carbon) simultaneously. Kirby et al. (2006), Ancev (2013) and Connor et al. (2013) all identified similar arbitrage opportunity for an EWH to sell when water was scarce and high value to irrigators but not so valuable environmentally, and to buy when irrigation opportunity cost was low and environmental value high. The mechanism identified here is different as it has no water selling, but rather only buying additional water when it is less valued for irrigation and has high carbon sequestration incremental value.

4.1 Policy opportunity and challenges

Markets are increasingly a feature of how environmental goods are provided, especially for carbon emissions since the Paris Agreement on climate change (United Nations, 2015). Market are also increasingly used to provide for public good environmental water outcomes, such as in the Australian MDB setting. Currently, settings with institutions supporting both carbon markets and water markets in the way described in this study are rare. However, the findings are broadly relevant internationally because the development of such institutions and governance is growing globally both in carbon and water realms.

Future application of this approach may be particularly relevant in comparable semi-arid regions with floodplains which benefit from periodic inundation and could be aided by the proposed carbon-water trading strategy. One such location in the semi-arid south west United States, where flood flows are the driving factor of riparian biomass and biodiversity (Stromberg et al.,

2017) and water market institutions currently support the purchase of in-stream flows (Garrick et al., 2011).

Where supporting institutions may exist, there are both opportunities for additional joint benefit and risks of non-additionality when multiple markets interact. This study is an example of supporting institutions for markets for two highly relevant ecosystem services: carbon sequestration and hydrological floodplain benefits. Results show potential for co-benefits from a wide array of public good benefits (e.g. habitat maintenance, soil productivity, water and nutrient cycling), potentially with value much greater than the value of carbon abatement benefit considered in this study.

While this study identified improved environmental outcome and cost saving opportunities, there are a number of policy design challenges to actually realizing the opportunities identified. One policy challenge relates to potential for non-additionality or “anyway” projects that proponents might take on even without carbon credit payments (Burke, 2016; Lui and Swallow, 2018). The risk of non-additionality is particularly pronounced in the context to reducing livestock grazing to increase growth of woody revegetation that sequesters carbon (Evans, 2018). This is because changes in global commodity prices and terms of trade can significantly motivate marginal land change from grazing to forest cover even in the absence of carbon payment policy (Marcos-Martinez et al., 2019). The type of carbon credit suggested in this study could also incentivize “anyway” projects. This could occur in the sense that an environmental entity like a water trust that already intended to buy water for the environment could finance this from carbon credits. This type of non-additionality may not necessarily create an undesirable outcome if the intent of policy is to tip incentive balance toward primarily unpriced public good values generated by water and climate regulation ecosystem services. In such context, non-additionality could even be seen as positive outcome, if adequate governance is in place to ensure that the entity that takes the actions uses credit income to expand their budget for environmental investments.

Implementing effective carbon credits of the type described in this study would also require policy design to address interrelated challenges of impermanence, risk, and monitoring costs (Meijaard et al., 2014). The challenges arise from uncertainty about future forest and soil carbon stocks changes. For example, from forest fires, reduced establishment success, and reduced tree carbon storage in drought (Evans, 2018). A monitoring challenge arises because the value of carbon abatement with and without actions to generate carbon credits is uncertain in this context.

Non-additionality and impermanence risks are both policy challenges requiring further research and policy innovation. However, pragmatic approaches built into credit payment policy like the Australian ERF, such as discounting the level of credit relative to what is estimated without accounting for risks, can be further developed to deal with these risks (Evans, 2018).

A final observation is that similar strategies involving the trade of other and bundled marketable ecosystem services such as biodiversity, water quality or land degradation credits or payments, may also be a viable means of offsetting environmental water purchase costs.

4.2 Limitations and implications

The model has several limitations which have implications for the interpretation and application of the results. A key difficulty was the limited data available on carbon stored in some local tree species (e.g. Black Box and Lignum), approximating carbon growth rates, and absence of data on carbon stored in grassland, soil and woody debris. This does not, however, limit confidence in results, given the sensitivity testing undertaken. A more fundamental challenge is interpretation of results given the accounting solely for carbon market value in this study. We do not provide estimates for additional values likely to be enhanced through environmental flows such as water quality, fisheries provisioning, supporting ecological function like habitat value, and less tangible existence, sense of place and cultural values.

In principle, the choice of a rolling horizon algorithm with up to three years in foresight imposes the limitation that the three years may not be sufficient to fully represent the long-term effects of annual floodplain carbon stocks. However, scenario analysis of one, two and three foresight years (e.g. N=1,2,3) and testing for longer N=5 year rolling horizon showed that very little value of additional carbon was achieved for longer than three year decision time horizons even though conceptually there can be cumulative incremental tree growth which can persist long beyond inundation. Applying similar methods to another basin would lead to results that differ as a function of factors such as geomorphology and eco-hydrology and endemic tree species of the river basin in question. In particular, the current model is set in a context where competition for irrigation water is largely between annual agricultural crops and perennial natural forests, where the opportunity cost of storing carbon in agricultural biomass is low due to the annual harvest regime. However, in cases where there is competition between perennial tree crops (e.g. almonds, oranges) and native forest carbon sequestration, the potential loss of carbon stocks due to tree crop decay (in addition to other costs), would need to be taken into account when considering the economically efficient reallocation volume. This would likely result in lower volumes being reallocated. However, this limitation does not diminish the wider explanatory and conceptual value to other river catchments facing similar environmental issues, especially in a global context with growing use of markets in public good climate and water.

5 Conclusions

This study presented a dynamic hydro-economic simulation of temporally varying flow, floodplain inundation, floodplain tree carbon storage, and irrigation and environmental water use values, with a case study of the Lowbidgee in the Murray-Darling Basin. Application of the model demonstrated potential cost-neutral opportunities to finance temporary water purchases through the generation and sale of carbon credits in co-existing carbon and water markets. The results suggest that when two formalized markets for public good services exist, there is potential opportunity for the generation of multiple public good ecosystem service values which are joint in production and potential for tangible value to be realized.

Institutional development will be required to facilitate this latent potential. In the case of Australia's MDB, fundamental underpinning water markets are in place, hence the key challenge would be developing accounting methods of generating tradable carbon credits (e.g. through floodplain inundation) and to deal with risks. The results of this case study suggest that there may be value in the further exploration of this idea to assess the generalizability of the suggested approach and applicability to other cases.

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