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A bioeconomic analysis of conserving freshwater values in an agricultural landscape

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Abstract. Water is a finite resource that must be shared among multiple users and economic and conservation objectives can often be seen as being in conflict. We explored this perception by conducting an integrated bioeconomic analysis of irrigated agriculture and the conservation of freshwater attributes in an agricultural landscape, the Tasmanian Midlands. We constructed a simple bioeconomic model based on current hydrology, water allocation, land use and freshwater ecosystem values, and quantified the economic returns from irrigation under a range of future climate, agricultural development and conservation scenarios. We found that projected climate conditions and conserving freshwater values in good condition had small effects on economic returns to irrigators, and that enterprise diversity and the area irrigated were major drivers of economic returns in this landscape. The availability of land suitable for irrigation rather than irrigation water itself appeared most likely to limit the economic returns from irrigation in the future. We provide a multi-criteria analysis for comparing development and conservation scenarios at a regional scale to inform planning and decision making in conservation and natural resource management. Our approach brings irrigation and conservation concerns into the same context and demonstrates that conservation need not necessarily limit agricultural development.

Additional keywords: bioeconomic modelling, climate projections, conservation planning, land use, optimisation, scenario comparison.

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Introduction

Reserving land and water for conservation measures, which can be perceived as an imposition on private landowners and irrigators (Main *et al.* 1999; Shandas 2007; Stoeckl *et al.* 2015), has launched the development of incentive and covenanting programs to facilitate conservation on private lands in agricultural landscapes (Iftekhar *et al.* 2014). Such programs appear to be successful in achieving terrestrial conservation goals (Langpap and Kerkvliet 2012), and have potential for improving river health (Langpap *et al.* 2008). Over half the world's accessible freshwater is used by humans, largely for agriculture (Postel *et al.* 1996), and freshwater ecosystems are the most degraded ecosystems on earth (MEA 2005). Given that water is often a finite resource that must be shared among multiple users, economic and conservation objectives can often be seen as being in conflict, requiring the comparison and negotiation of trade-offs among stakeholders (Grafton *et al.* 2011; Hermoso *et al.* 2012; Bryan *et al.* 2013).

The necessity to analyse trade-offs has led to the development of integrated water management approaches that bring multiple disciplines into the same framework and provide information on the economic consequences of water management (Letcher *et al.* 2007; Brouwer and Hofkes 2008; Harou *et al.* 2009). One approach has been to couple separate economic and hydrological models and tailor a solution from these components (e.g. Bharati *et al.* 2008). These coupled approaches allow the combination of simulation and optimisation modelling, and therefore, analysis at finer detail, but generally require major software engineering to enable their articulation (Bharati *et al.* 2008; Cai 2008). A more efficient approach, particularly for exploring alternative scenarios at broad scales, is considered to be the use of integrated bioeconomic models that are solved endogenously (Letcher *et al.* 2007; Brouwer and Hofkes 2008; Cai 2008), although they may require poorer resolution inputs in order to achieve a solution (Bharati *et al.* 2008; Harou *et al.* 2009). In the case of catchment management, such models are built on the underlying hydrology of the system, routing flows through nodes and attributing water usage to each node according to properties such as urban supply or agricultural demand. These models can then generate optimal solutions of water use based on the economic value of water for different uses and subject to various physical, environmental or policy constraints (Brouwer and Hofkes 2008; Harou *et al.* 2009; Grafton *et al.* 2011). Historically, optimisation models have been used primarily by irrigation engineers and agricultural economists (e.g. Burt 1964), and are now being

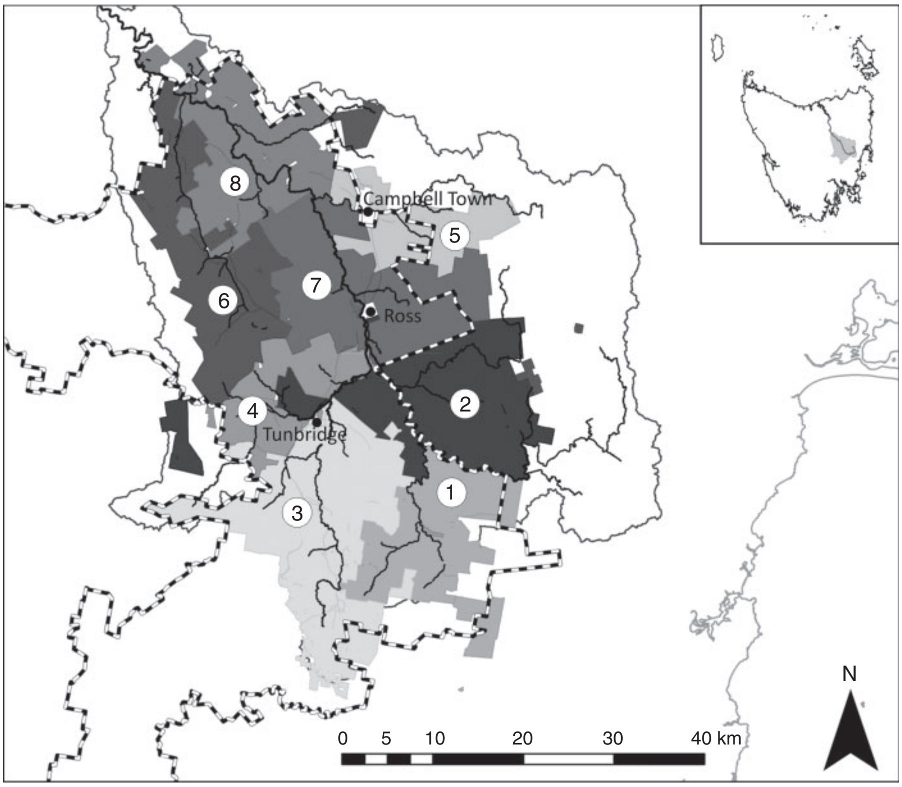

Fig. 1. Map of the study region (main image) and its location in Tasmania (inset). The Macquarie River catchment, with its main watercourses (black lines) and three main towns of Campbell Town, Ross and Tunbridge (black dots) is shown, as is the irrigation area for the Midlands Water Scheme (dashed line). The eight regions used in the bioeconomic model are shaded grey and numbered. Regions were defined by major watercourses, the location of water extractions and irrigation activity, and property boundaries.

used more frequently to explore water management and policy scenarios at catchment scales (e.g. Ward and Booker 2003; Harou *et al.* 2010).

Such models provide considerable scope for the integration of detailed ecological information as well as modelled climate projections, although these types of applications are only recent and still rare (Brouwer and Hofkes 2008; Harou *et al.* 2009; Hermoso *et al.* 2012). Recent demonstrations show that bioeconomic modelling can be useful for exploring potential conflicts between socioeconomic and environmental goals in the design of managed flow regimes (e.g. Grafton *et al.* 2011; Akter *et al.* 2014). In a detailed, high-resolution study of the lower Murray River in South Australia, trade-offs between environmental and socioeconomic values were not as extensive as assumed, but achieving both objectives would require different infrastructure and flow management regimes; findings that have since informed government policy and investment (Bryan *et al.* 2013). One of the major advantages of bioeconomic modelling is the provision of optimal solutions under varying scenarios to inform and support decision making by water managers, stakeholders and policy makers (Letcher *et al.* 2007; Harou *et al.* 2009; Hermoso *et al.* 2012; Bryan *et al.* 2013).

Integrated bioeconomic modelling to inform catchment water management in Australia has been confined to the Murray–Darling Basin (e.g. Grafton *et al.* 2011; Bryan *et al.* 2013; Qureshi *et al.* 2013; Akter *et al.* 2014) and has not yet been applied in Tasmania. Currently, water management planning in Tasmania relies on hydrological and ecological catchment assessments that are based on a sound scientific framework (Bobbi *et al.* 2014), but as yet there is no objective means of integrating these assessments with economic values to explore trade-offs under different scenarios, a common situation globally (Ward and Booker 2003; Hermoso *et al.* 2012; Akter *et al.* 2014; Stoeckl *et al.* 2015). We use the Tasmanian Midlands, one of the oldest agricultural landscapes in Australia, as a case study to demonstrate how bioeconomic modelling can reveal the relationships among hydrological, ecological and economic drivers at a regional scale. The economic returns from major irrigation development in the Midlands, and how these returns might be influenced by the development of irrigated agriculture, projected climate change, and the protection of freshwater ecological values, are not clearly quantified. We use simple bioeconomic modelling to address these issues and provide a comparison of economic returns (defined here as the optimal sum of gross margin returns) from irrigation under a range of scenarios. We demonstrate that achieving environmental goals may not be the constraint on economic returns they are often perceived to be, and present a straightforward and rapid approach for exploring water use scenarios to support informed decision making by natural resource managers and stakeholders.

Table 1. Four major variables (hydrology, water allocations, land enterprise mix, and freshwater ecosystem values), with their respective levels defined (and data sources identified), selected to evaluate their relative effects on economic returns from irrigated agriculture in the study region

Variable levels		Data source and metrics
Hydrology		Hydrological time series derived from six CCAM dynamically downscaled global climate models and an AWBM two-tap rainfall-runoff model (Bennett *et al.* 2010):
H1	Modelled natural flows, current (1991–2009)	Multi-model mean of median annual discharge of runoff generated by six GCMs
H2	Modelled natural flows, future dry (2040–2069)	Median annual discharge of runoff generated by GCM with driest projection (CSIRO-Mk3.5)
H3	Modelled natural flows, future wet (2040–2069)	Median annual discharge of runoff generated by GCM with wettest projection (UKMO-HadCM3)
Water allocations		Extracted from water allocation registers:
W1	State_HR	Tasmanian Government; high reliability allocations only
W2	State_private_HR	Tasmanian Government and private; high reliability allocations only
W3	State_HRF	Tasmanian Government; high reliability, low reliability and flood allocations
W4	State_private_HRF	Tasmanian Government and private; high reliability, low reliability and flood allocations
Land enterprise mix		Current enterprise mix derived from landholder survey (Lockwood *et al.* 2015) and media reports, and future enterprise mix derived from DPIPWE's enterprise suitability mapping:
L1	Dryland_production	Current enterprise mix but with no irrigation (modelled without allocations)
L2	Current_limited	2 dryland + 10 currently irrigated crops, each limited by currently irrigated area of individual crops
L3	Current_unlimited	2 dryland + 10 currently irrigated crops, limited by total area irrigated (water allocated to high-value crops)
L4	Future_limited	2 dryland + 20 future irrigated crops, each limited by the area suitable for their growth
L5	Future_unlimited	2 dryland + 20 future irrigated crops, limited by total irrigable area (water allocated to high-value crops)
Freshwater ecosystem values (each × 4 condition classes)		Extracted attribute condition scores for ~6500 river sections from Conservation of Freshwater Ecosystem Values database (CFEV 2005, DPIW 2008), converted to area of river section catchments, and summed over condition classes: best (C4), good (C3), moderate (C2) to poorest (C1):
F1	Overall condition	Naturalness score (≥0.85, <0.85, <0.6, <0.3), derived from multiple variables
F2	Fish assemblage condition	Fish condition score (1, 0.5, 0, −9), based on presence of native fish and flow alteration
F3	Macroinvertebrate assemblage condition	Macroinvertebrate condition score (≥0.9, <0.9, <0.6, <0.3), based on natural density and composition
F4	Riparian vegetation condition	Riparian vegetation condition score (≥0.8, <0.8, <0.6, <0.2), based on proportion of native vegetation
F5	Aquatic plant communities	Macrophyte assemblage type (M6, M4A/M5A, M4B/M5B, other), based on assemblage density and type
F6	Threatened species	Number of significant species and communities (≥4, ≤3, ≤2, ≤1)

Materials and methods

Model structure

The study region, located in the Tasmanian Midlands, is the part of the Macquarie River catchment that lies within the irrigation area of the new Midlands Water Scheme (MWS), which began operation in 2014 (Fig. 1). This region has a long history of producing fine merino wool and is over 95% privately owned, and is also one of Australia's 15 biodiversity hotspots listed by the Australian Government (Lockwood *et al.* 2015). The 184 000-ha study area has a population of less than 2000 (Gadsby *et al.* 2013) and annual rainfall averages ~515 mm.

The Macquarie River was originally an intermittent river but was dammed in 1865 and now has regulated low flows to supplement irrigation and hydropower production; spates and floods are essentially natural (DPIPWE 2012). The median natural annual flow through the study region is ~70 000 ML year^{-1}, with an average annual flood peak of 48 000 ML. Based on current land use data (see below), ~80% of the study region is used for grazing sheep and cattle (one quarter of which is on native vegetation), and 7% is used for irrigation, predominantly poppies for medical opiates. The remaining proportions are reserved (10%), forestry plantations (1%), watercourses and waterbodies (1.5%), or under intensive use such as urban infrastructure (0.5%).

The model comprised eight regions (11 000–35 000 ha), each defined by major watercourses and delineated by cadastral property boundaries (Fig. 1; base image by TASMAP (www.tasmap.tas.gov.au), © State of Tasmania). Ninety-five properties were aligned with the major watercourses, based on the location of their water extraction points and irrigation activity, to form the regions (average of 12 ± 8 properties per watercourse region). All spatial data were derived using ArcGIS V.10.

Four major variables (hydrology, water allocations, land use, and freshwater ecosystem values, each with 3–6 levels) were defined and modelled in a range of combinations or 'scenarios' to assess the relative effects of hydrological, ecological and economic drivers on economic returns from irrigation (Table 1 and detailed in subsections below). Hydrology was evaluated at three levels reflecting projected climate change: current flows, future dry flows and future wet flows. Water extraction was modelled at four levels based on irrigation provider (state and private) and allocation reliability (high reliability and low

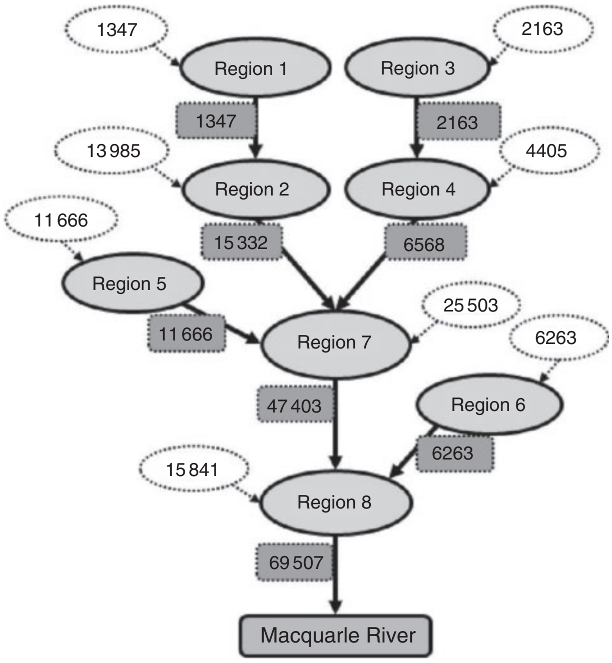

Fig. 2. Model schematic of flows routed through regions. Hydrological inflows (dotted ellipses) and outflows (dotted rectangles) for each region (solid ellipses), connected by major watercourses (solid arrows). Flows are for current natural conditions (i.e. without allocations), and are the multi-model means of the median annual discharge (ML year^{-1}) of runoff generated by six dynamically downscaled global climate models. Future climate conditions were evaluated by replacing these values with the median annual discharge from the driest, or wettest, projections for the medium-term future (2040–2069).

reliability + flood allocation; Table 1). Land use was modelled at five levels based on the area currently or potentially irrigated, and the diversity of crops currently or potentially irrigated (Table 1) to explore current and potential future development. Freshwater ecosystem values were modelled at 6 × 4 levels: six levels representing a different 'value' and each level evaluated at four classes of condition status or 'naturalness' (Table 1).

The variables were modelled in two separate sets of model runs. Hydrology (H1–3), water allocation (W1–4) and land enterprise (L1–5) variables were modelled as fully crossed combinations in the first set, resulting in $3 \times 4 \times 5 = 60$ model runs or irrigation 'scenarios' for comparing economic returns. This first set of model runs excluded freshwater ecosystem values that would have required $60 \times 6 \times 4 = 1440$ model runs. Instead, a subset of hydrology, water allocation and land enterprise levels (H1, W2, W4, L3 and L5) were evaluated with the freshwater ecosystem values (F1–6) to produce a more manageable second set of model runs ($1 \times 2 \times 2 \times 6 \times 4 = 96$). The selection of specific levels for this second set was made *a priori* and is explained below in each subsection.

Hydrology

Each region was linked according to the hydrological drainage such that the outflow of each region corresponded to a hydrological subcatchment (Fig. 2). All hydrological data was obtained from the Department of Primary Industries, Parks, Water and Environment (DPIPWE) and generated under the Climate Futures for Tasmania project (CFT; Bennett *et al.* 2010). The outflows under natural conditions (i.e. without extractions) were derived from the modelled natural runoff projections (on a daily time-step) generated from six CCAM dynamically downscaled global climate models (run on a high emissions A2 scenario) and an AWBM Two Tap rainfall-runoff model (Bennett *et al.* 2010). Current natural flows were represented by the median annual discharge for the current time period (1991–2009), averaged across all models (i.e. the central estimate; level H1). Future natural flows were represented by the median annual discharge for the medium-term future (2040–2069) under two conditions representing the extremes of the projection range for Tasmania: the driest projection (CSIRO-Mk3.5; level H2) and the wettest projection (UKMO-HadCM3; level H3; Table 1). All three hydrology levels were evaluated in the first set of model runs, but only current flows (H1) were evaluated in the second set because the freshwater ecosystem values used are current (DPIPWE 2012) and potentially not valid for medium-term future projections.

Water allocations

Water allocations were manually modelled for each region and distinguished between different irrigation providers (Tasmanian Government and private irrigation) and water reliability levels (high reliability and low reliability + flood harvesting allocations (levels W1–4; Table 1). Modelled current runoff projections (also CFT data) were used to validate the manual extraction of water allocations in the model. Water allocations for each region were obtained from DPIPWE's water register on the Water Information System of Tasmania (see http://wrt.tas.gov.au/wist/ui, accessed 19 March 2015) and from the water register for the new MWS. Approximately 110 000 ML year^{-1} is allocated by DPIPWE, over half of which are low reliability and (predominantly) flood allocations, although the flood allocations are largely unused as yet. Approximately 12 000 ML year^{-1} high reliability allocations have been privately sold to the study region through the MWS, with the water being transferred from outside the catchment and distributed via pipelines and major watercourses. Most properties in the study region have either DPIPWE or private allocations; less than 20% have both, and ~5% have no allocation. Two of the model regions (Regions 2 and 5; Fig. 2) are largely outside the irrigation area and receive no private water. All water allocation levels were modelled in the first set of model runs, but only those incorporating both state and private allocations (W2 and W4) were selected for the second set evaluating freshwater ecosystem values. State and private allocations are now in operation and our interest was in exploring potential effects of flood allocations that represent nearly half of all allocations.

Land enterprise mix

The proportion of major land use categories for each model region was calculated using the Tasmanian Land Use Summer 2009–2010 spatial dataset (www.thelist.tas.gov.au/listmap, accessed 3 December 2015), incorporating the Tasmanian Reserve Estate spatial dataset, production forestry, updated waterbodies from Geoscience Australia, and structured to align with the Australian Collaborative Land Use and Management Program. This dataset was used in conjunction with the Tasmanian Vegetation spatial dataset (TASVEG 3.0; State of Tasmania, 2013, www.thelist.tas.gov.au/listmap) and an irrigated land update to 2013 (Felicity Faulkner, pers. comm., DPIPWE). Major land use categories were reserved land (6–21% per region), grazing on native vegetation (12–49% per region), dryland grazing (31–63% per region), other dryland production including forestry (<3% per region), irrigated cropping (1–14% per region), urban infrastructure (<0.5% per region) and water (<2% per region).

Land enterprise mix was modelled at five levels (L1–5) based on current and future crops irrigated and the area of land irrigated. The current enterprise mix in each region was estimated from the land use data, media reports and data from a landowner survey conducted in 2012 (Lockwood *et al.* 2015), and validated by spatial scientists and agronomists from DPIPWE (Rhys Stickler and Hugh Griffiths, pers. comm., DPIPWE). The current mix comprised 12 enterprises: wool (dryland), lamb, beef (dryland), dairy, barley, wheat, oats, lucerne, canola, poppies, potatoes and cherries. This current enterprise mix was modelled at the first three levels: level L1, *dryland_production* with no irrigation (i.e. wool and beef only); level L2, currently irrigated crops limited by the area currently irrigated for each individual crop (*current_limited*); and level L3, currently irrigated crops only limited by the total area currently irrigated, that is, the model allocates water to the most high-value crops first (*current_unlimited*; Table 1).

The future enterprise mix for each region was estimated from enterprise suitability mapping conducted by DPIPWE. Digital soil mapping, localised climate data and rules defining characteristics for best crop performance (e.g. frost tolerance, soil salinity) were used to derive maps of suitability classes for 20 irrigated crops in a subsection of the study area (www.thelist.tas.gov.au/listmap). We used the proportion of 'well suited' and 'suitable' land (ha) to estimate the potential area of future crops across the study region. The future mix comprised the current 12 enterprises as well as a further 10 crops: ryegrass, pyrethrum, industrial hemp, carrots, carrot seed, blueberries, raspberries, hazelnuts, olives and wine grapes. The future enterprise mix was modelled at two levels: level L4 future irrigated crops limited by the area suitable for their growth within the total available irrigable area in each region (*future_limited*); and level L5, the model allocates water to the most high-value future irrigated crops and area is only limited by the total available irrigable area in each region (*future_unlimited*; Table 1). All land enterprise levels (L1–5) were evaluated in the first set of model runs, whereas only *current_* and *future_unlimited* (L3 and L5) were modelled in the second set with freshwater ecosystem values. These two *_unlimited* levels were selected to represent the likely increase in crop diversity associated with the operation of private allocations, and to evaluate an extreme scenario of irrigation development and identify potential limiting factors.

The total irrigable area for each region was defined as land that was both available and suitable for irrigation and represented a constant constraint across all variables and levels. It was determined from the land use spatial dataset and the land capability spatial dataset (www.thelist.tas.gov.au/listmap), a classification of land capable of supporting agricultural activities without degradation of the land resource (Grose 1999). The classification is based on soils, topography and climate, and land capability is ranked from Class 1 (best) to Class 7 (poorest, unsuitable for agricultural use). Only Classes 4, 5 and 6 are present across the study region, and Class 6 is deemed severely limited for agricultural use with low productivity and high erosion so was not included. Irrigable area was calculated as the area of land capability Classes 4 and 5 (suitable for irrigation) that corresponded with land area currently cleared, and therefore, available for irrigation. The total available irrigable area represents ~51% (±13) of each region.

Gross margins ($ ha^{-1} year^{-1}) and crop water requirements (ML ha^{-1} year^{-1}), calculated for the low-rainfall district of the Midlands and updated in June 2014, were obtained from DPIPWE (http://dpipwe.tas.gov.au/agriculture/investing-in-irrigation, accessed 3 December 2015). Gross margins provide estimates of the financial difference between returns from commodity sales and the costs associated with producing that commodity and are a commonly used measure of economic returns in agricultural economic modelling.

Freshwater ecosystem values

Freshwater ecosystem values (F1–6) were derived from Tasmania's Conservation of Freshwater Ecosystem Values (CFEV) spatial database (CFEV 2005; DPIW 2008). The CFEV database provides a relative conservation value of all Tasmanian freshwater-dependent ecosystem units (e.g. river sections, waterbodies) based on their naturalness (i.e. condition status), distinctiveness and representativeness (i.e. distribution) in a state-wide context (DPIW 2008). We extracted a range of ecosystem attributes from all 6532 river sections across the study region to represent six freshwater ecosystem values identified as environmental objectives and targeted for preservation in the catchment water management plan for the Macquarie River (DPIPWE 2012; Table 1). In the absence of hydro-ecological response models required to sustain community- and ecosystem-level attributes (e.g. Akter *et al.* 2014), we used the catchment area of each river section (average 23 ha ± 40) to enable these attributes to be expressed in hectares and modelled as land excluded from agricultural production and conserved for freshwater values (Harou *et al.* 2009).

The attributes extracted were scores of overall condition (level F1, derived from multiple biological, geomorphic and threat variables; DPIW 2008), fish assemblage condition (level F2), macroinvertebrate assemblage condition (level F3), riparian vegetation condition (level F4), aquatic macrophyte community (level F5), and threatened species and communities (level F6). The scores for each attribute were ranked into four classes from poorest condition to best (Classes 1–4), based on their classification in the CFEV database (CFEV 2005; Table 1). Each attribute was modelled as the aggregate area required to conserve it in its best condition (Class 4), with each subsequent condition class consecutively added to indicate the increasing area required to conserve attributes in good, moderate or poor condition (i.e. attributes were prioritised according to their condition status). As described above, freshwater ecosystem values (and their condition classes) were only evaluated in the second set of model runs, with a subset of hydrology (H1), water

Table 2. Annual economic returns from irrigation from the first set of model runs evaluating effects of hydrology, water allocation and land enterprise variables

All results presented are under current flows (H1), and effects of projected flows (H2 and H3) under current land enterprise levels (L2 and L3) are also presented; projected hydrology had no effect on economic returns under future land enterprise levels (L4 and L5) so are not presented. Note that dryland production (L1) modelled economic returns in the absence of irrigation. Values are rounded to the nearest AU\$1 million, and percentage change from current to projected flows is presented in parentheses

	Water allocation levels							
Land enterprise and hydrology levels	W1 (State_HR)		W2 (State_private_HR)		W3 (State_HRF)		W4 (State_private_HRF)	
L1 (dryland production)	53		53		53		53	
L2 (current_limited)	83		88		87		90	
H2 (future dry flows)	82	(−1.2%)	87	(−1.1%)	86	(−1.1%)	90	(−0.1%)
H3 (future wet flows)	84	(+1.2%)	88	(+0.5%)	88	(+1.1%)	90	(+0.1%)
L3 (current_unlimited)	284		344		421		477	
H2 (future dry flows)	279	(−1.7%)	343	(−0.3%)	369	(−12.4%)	426	(−10.7%)
H3 (future wet flows)	291	(+2.4%)	347	(+0.9%)	494	(+14.8%)	563	(+15.3%)
L4 (future_limited)	1802		1802		1802		1802	
L5 (future_unlimited)	2689		2689		2689		2689	

allocation (W2 and W4) and land enterprise variables (F3 and F5).

Optimisation modelling

The bioeconomic model was coded in the General Algebraic Modelling System (GAMS) language (Brooke *et al.* 1996) and solved using the CONOPT algorithm. The objective function of the model (Eqn 1) was to maximise economic returns under each model run (Table 1):

$$\max \sum gm_i x_{i,j} \tag{1}$$

Subject to a hydrological constraint (Eqn 2), an allocation constraint (Eqn 3), allocation source (Eqn 4), a land area constraint (Eqn 5), an irrigable land area constraint (Eqn 6), land enterprise mix (Eqn 7) and an environmental constraint (Eqn 8):

$$\sum ml_i x_{i,j} \leq \sum influ_j + \sum infl_j - \sum ml_i x_{i,j} \ \forall j \tag{2}$$

$$\sum ml_i x_{i,j} \leq allocation_j \ \forall j \tag{3}$$

$$\sum sall_{l,j} + pall_{k,j} \leq allocation_j \ \forall j \tag{4}$$

$$\sum x_{i,j} \leq la_j \ \forall j \tag{5}$$

$$\sum x_{i,j} \leq lg_j \ \forall j \tag{6}$$

$$\sum x_{i,j} \leq ls_{i,j,s} \ \forall i, j, s \tag{7}$$

$$\sum x_{i,j} \leq la_j - lc_b \ \forall j, b \tag{8}$$

where ml_i is the volume of water (ML) required to grow 1 ha of crop i; $x_{i,j}$ is the area of crop i grown in region j; $influ_j$ is the inflow of water to regions upstream of region j; $infl_j$ is the inflow of water to region j from outside the system; $allocation_j$ is the allocation of water at region j; $sall_{i,j}$ is the state water allocation i to region j; $pall_{k,j}$ is the private water allocation k to region j; la_j is the land area of region j; lg_j is the area of irrigable land in region j; $ls_{i,j,s}$ is the land available to grow crop i in region j under land enterprise mix s; and lc_b is the area of land set aside under environmental condition b.

Results

The model was able to find an optimal solution for all scenarios on both sets of model runs. The first set of model runs evaluating hydrology, water allocations and land enterprise mix on economic returns (and excluding freshwater ecosystem values) demonstrated that current irrigation increased economic returns from dryland production (L1) by over 35% (Table 2), even when the total area irrigated only represents ~7% of the total available area (L2). Economic returns increased by 5–7% with the addition of private (W2) or low reliability + flood allocations (W3), or both together (W4). Economic returns were more than tripled when the model could allocate irrigated area to the most high-value crops first (L3), quadrupled with the addition of private or low reliability + flood allocations (W2 and W3), and quintupled with the addition of *both* private and low reliability + flood allocations (W4; Table 2). Projected flows (H2 and H3) had only small effects on economic returns under land enterprise levels, except when high-value crops were irrigated first (L3) and low reliability + flood allocations were included: flood allocations considerably increased the effects of projected flows such that economic returns were reduced (under dry projections) or increased (under wet projections) by 10–15% (Table 2).

By far the greatest influence on economic returns was the effect of future land enterprise scenarios. Increasing the diversity of irrigated crops (but constraining them to the area suitable for their growth; L4) increased economic returns by over 20 times to AU\$1.8 billion annually, and by over 30 times to \$2.7 billion when individual crops were unlimited by the area

Table 3. Annual economic returns from irrigation from the second set of model runs evaluating effects of freshwater ecosystem values
Each value was modelled as increasing the land area annexed (from irrigation) for each class of decreasing condition status (Class 4–1, best to poorest condition). Freshwater ecosystem values were modelled with a subset of variable levels: current flows (H1), with water allocations excluding and including flood allocations (W2 and W4 respectively), and under the *current_unlimited* (L3) land enterprise mix. While modelled, the *future_unlimited* (L5) level is not presented: it showed no variation depending on freshwater ecosystem values. Values are rounded to the nearest AU$1 million, and percentage change from economic returns *without* taking into account freshwater ecosystem values is presented in parentheses

Freshwater ecosystem values	W2 (State_private_HR) Class 4 (best)		Class 3 (good)		Class 2 (moderate)		Class 1 (poor)	
Overall condition	330	(−4.0%)	321	(−6.7%)	299	(−13.1%)	230	(−33.1%)
Fish assemblage condition	318	(−7.5%)	318	(−7.5%)	316	(−8.1%)	230	(−33.1%)
Macroinvertebrate assemblage condition	327	(−4.9%)	301	(−12.5%)	293	(−14.8%)	234	(−32.0%)
Riparian vegetation condition	329	(−4.4%)	325	(−5.5%)	311	(−9.6%)	230	(−33.1%)
Aquatic macrophyte community	341	(−0.9%)	337	(−2.0%)	332	(−3.5%)	257	(−25.3%)
Threatened species and communities	344	(0%)	344	(0%)	327	(−4.9%)	230	(−33.1%)
Freshwater ecosystem values	**W4 (State_private_HRF) Class 4 (best)**		**Class 3 (good)**		**Class 2 (moderate)**		**Class 1 (poor)**	
Overall condition	464	(−2.9%)	455	(−4.8%)	432	(−9.6%)	306	(−36.0%)
Fish assemblage condition	452	(−5.4%)	452	(−5.4%)	450	(−5.9%)	306	(−36.0%)
Macroinvertebrate assemblage condition	461	(−3.6%)	435	(−9.0%)	426	(−10.9%)	310	(−35.1%)
Riparian vegetation condition	462	(−3.3%)	458	(−4.2%)	444	(−7.1%)	306	(−36.0%)
Aquatic macrophyte community	474	(−0.8%)	470	(−1.7%)	465	(−2.7%)	333	(−30.3%)
Threatened species and communities	478	(0%)	478	(0%)	461	(−3.6%)	306	(−36.0%)

suitable for their growth (L5). In both the *future_limited* and *future_unlimited* scenarios, there was no effect of water allocation or hydrology (Table 2). The availability of land suitable for irrigation (irrigable area) was the factor limiting further growth in irrigation, and thus, economic returns.

The second set of model runs demonstrated that annexing land from irrigation to conserve freshwater ecosystem values in best condition had small effects on annual economic returns (<7.5%), but varied according to the value used (Table 3). Conserving instream fish assemblages in their best condition had the greatest impact on economic returns (compared to threatened species and aquatic plant assemblages, for example) and this was consistent with and without flood allocations (Table 3). Annexing land to conserve values in best *and* good condition (Class 3) resulted in greater impacts on economic returns (up to 12.5%), but in this scenario was driven by macroinvertebrate rather than fish assemblages. For all values, increasing the amount of land annexed from irrigation to conserve values in moderate and poor condition (Classes 2 and 1) had greater impacts on economic returns, although these impacts were somewhat mitigated when flood allocations were included. Annexing land to conserve freshwater ecosystem values, regardless of their condition, reduced economic returns by up to 36%, but only under the *current_unlimited* (L3) land enterprise level (Table 3). Annexing land for conservation had no effect on the *future_unlimited* (L5), as for variables in the first set of models runs, because economic returns under this scenario were only limited by the availability of irrigable area.

Discussion

We have demonstrated that simple bioeconomic modelling can reveal the relative influence of hydrological, ecological and economic constraints on economic returns from irrigation at a regional scale. We found that agricultural development, particularly the area and diversity of irrigated agriculture, was the strongest driver of aggregate gross margin returns in the study region. The model further indicated that the availability of land suitable for irrigation was the major limitation on economic returns in the future. Projected flows under climate change scenarios and conserving freshwater ecosystem values represented small limitations on economic returns. Our findings have great potential for natural resource planning, demonstrating the usefulness of bioeconomic modelling as a simple and rapid ‘snapshot’ approach for bringing conservation and irrigation concerns into the same arena and determining their relative influence on economic returns from agricultural production.

While hydroeconomic modelling has supported agricultural planning for many decades (e.g. Burt 1964), the incorporation of specific conservation goals is recent, largely due to the difficulty in resolving the scales of different data inputs (Brouwer and Hofkes 2008; Harou *et al.* 2009; Stoeckl *et al.* 2015). Two studies incorporating ecological data to explore the economic trade-offs of providing environmental flows suggest that achieving environmental goals need not come at the expense of achieving socioeconomic goals (Bryan *et al.* 2013; Akter *et al.* 2014), a conclusion supported by our study. Indeed, delivering water for environmental rather than irrigation purposes has been shown to increase overall economic benefits (Grafton *et al.* 2011; Akter *et al.* 2014).

Both Bryan *et al.* (2013) and Akter *et al.* (2014) used specific flow-ecology response relationships to define environmental flow regimes, and found that some trade-off between environmental and socioeconomic goals would be required in flow regime design. We did not use flow-ecology response relationships as they tend to be quite context-dependent, specific to particular sites and species (or communities), often rely on incomplete ecological knowledge, and cannot be reliably extrapolated for broadscale overviews (Ward and Booker 2003; Akter *et al.* 2014). Bryan *et al.* (2013) were able to define inundation responses for ecohydrological units and model these at fine spatial and temporal resolutions, allowing specific ecosystem units to be targeted and overcoming the problem with capturing detailed ecohydrological processes (Akter *et al.* 2014).

Flow-ecology response relationships have been developed for sites in the Macquarie River (including sites in our study region), indicating that broadwater macrophyte communities are a key value for the catchment and sensitive to changes in flood duration and timing; these relationships support environmental flow recommendations and underpin the availability of flood allocations in the study region (DPIPWE 2012). Our approach is not intended to supercede these site specific environmental flow studies that recommend specific volumes and timing of water, but has the capacity to incorporate such recommendations in future bioeconomic analyses and may be useful for broadscale planning. Our findings indicate that aquatic plant communities have negligible impacts on economic returns and that conserving the major ecological values of the Macquarie River can be achieved without trading off economic returns from irrigation.

These findings challenge some perceptions of water resource stakeholders in the study region (Ketelaar *et al.* 2012) as well as more broadly (Main *et al.* 1999). Assessments of land use tend to assume that land is used for either conservation or commodity production, suggesting a trade-off between the two must occur (Stoeckl *et al.* 2015). Nelson *et al.* (2009) found a negative relationship between commodity production and biodiversity conservation in terms of the area of land used for each objective. We found that if land area could be expressed as gross margins per hectare for particular crops, then such land area trade-offs do not necessarily translate to an impact on economic returns, at least at a regional scale. Stoeckl *et al.* (2015) found that if biodiversity and economic objectives are treated as equivalent farm ‘outputs’ at the individual property scale, then there is little evidence of market outcomes being compromised. This was particularly apparent for properties managed with positive conservation attitudes and with diversified agriculture (Stoeckl *et al.* 2015), and is supported by research on landowner attitudes in our study region indicating that nimble and innovative land management styles have more capacity to adapt to future change (Lockwood *et al.* 2015).

We found that water appears to be somewhat underused in the study area, with private and flood allocations each contributing ~5% to annual economic returns. We note that at the time of modelling the MWS had only sold approximately two thirds of its allocations and only began delivering irrigation water in October 2014. Moreover, flood allocations (representing approximately half the government allocations in the study region) are incorporated in the draft water management plan for the catchment (DPIPWE 2012) but are not yet being taken up. Consequently, over half the water we have modelled as being fully allocated is not reflected in current production or the economic returns from that production. More substantive contributions from these water sources can be seen under the future scenarios where more water is put towards higher value irrigation, resulting in considerable increases in economic returns, and that water availability is not likely to be a major factor limiting agricultural production.

Our modelling suggested that the effects of projected climate scenarios on economic returns from irrigation were small and primarily seen in the use of flood allocations. Rainfall and runoff projections for the study region show little change in annual volumes from current circumstances, however, the timing of peak rainfall events is projected to change considerably, from winter and spring to autumn (Bennett *et al.* 2010). This may result in less opportunity to harvest flood allocations before the summer irrigation season and suggests that effects of projected climate scenarios are more likely to be observed at finer temporal scales such as seasonally or monthly. While our study is one of the first to use bioeconomic modelling to explore effects of projected climate scenarios on economic returns from irrigation, the annual scale of hydrological data we have used, while standard for economic modelling, is coarse for hydrological modelling (Brouwer and Hofkes 2008; Bryan *et al.* 2013) and has potentially resulted in underestimating effects of the climate scenarios. Monthly time-step data would enable more accurate scenarios of flood and drought regimes to be explored and could be used to inform environmental flow regime design (e.g. Bryan *et al.* 2013).

The differences in economic returns among levels within each major variable (e.g. among projected flows or freshwater ecosystem values) provided an indicator of uncertainty among the data inputs (Brouwer and Hofkes 2008; Grafton *et al.* 2011) and suggested the model was most sensitive to changes in land use. For example, the condition classes for each ecological attribute were defined by the literature and expert opinion (DPIW 2008), yet the small effects between best, good and moderate condition status on economic returns showed that changing the cut-off points for each condition class would have had minimal effects on the results.

The primary purpose of our bioeconomic model was to provide a rapid, regional scale overview of the economic returns from irrigation in an agricultural landscape, and how those benefits are likely to be modified under a range of possible future scenarios. Our approach demonstrated that conserving freshwater ecosystem attributes is not necessarily a limiting factor on agricultural development, supporting recent findings from both the economic (Akter *et al.* 2014) and conservation planning literature (Stoeckl *et al.* 2015). In our system, the availability of irrigable land was by far the strongest limiting factor on future agricultural development. The model outputs enable water managers, stakeholders and policy makers to engage in a more informed dialogue about the regional planning of land and water resources in the Tasmanian Midlands. However, the approach has potential in any agricultural landscape by providing an objective means of integrating hydrological, ecological and economic values to explore trade-offs in the use of limited water resources.

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