Integrated hydro-ecological and economic modeling of environmental flows: Macquarie Marshes, Australia

Sonia Akter[1], R. Quentin Grafton[2], Wendy S. Merritt[3,]

[1] Department for Economics, Helmholtz Centre for Environmental Research – UFZ, Leipzig, Germany.

sonia.akter@ufz.de

[2] Crawford School of Public Policy, The Australian National University, Canberra, Australia.

quentin.grafton@anu.edu.au

[3] The Fenner School of Environment and Society, The Australian National University.

wendy.merritt@anu.edu.au

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Statement of authorship

SA conducted the socio-economic output analysis, RQG conceived the combined integrated modelling approach, funded the socio-economic research and helped prepare the manuscript, and WM undertook the hydro-ecological model output analysis. Authorship is alphabetical.

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Highlights:

1. We provide a framework that combines hydro-ecological and economic modelling to valuing environmental flows.
2. This framework can be applied in any river basin where there is sufficient data available.
3. We generate monetary values of water for a wetland of global significance.
4. We demonstrate how the estimated values can be used to evaluate trade-offs across competing water users.

Abstract

This study provides a method to combine hydro-ecological response model outputs and nonmarket economic values of wetland inundation to estimate a unit price of environmental water. We show how using an integrated socio-economic and hydro-ecological modelling approach may assist policy makers with water allocation decisions across competing uses. The IBIS decision support system incorporates a hydro-ecological model and is used to estimate the habitat suitability condition of wetland attributes for a given hydrology scenario. Non-use economic values of wetland attributes obtained by non-market valuation studies are then linked to the hydro-ecological model outputs to estimate marginal value of environmental flows. The contribution is to provide a robust, scientifically and economically valid method to estimate the marginal value of environmental water and to quantitatively evaluate the trade-offs involved in water allocation decisions across competing uses for water.

1. Introduction

River irrigation water demand has increased dramatically over the past few decades and has become the single most important factor in the reduction of stream flows in four continental river basins (Grafton et al., 2012). Critical to promoting better river basin governance is a quantitative method for comparing extractive and non-extractive values for water (Ward and Booker, 2003). Such a framework would allow decision makers to evaluate the trade-offs from re-allocating water used for extractions to environmental flows.

Two key challenges when comparing the payoffs to extractive water use versus non-extractive stream flows include: (1) the need for ecological response models to predict the likely outcomes of increased stream flows and (2) a method to value these ecological responses in monetary terms. These two approaches have rarely been combined in a way that generates a scientifically valid method to estimate marginal values of water that are consistent with economic theory, stream hydrology and modelled ecological response. In one of the earliest integrated modelling application, Ward and Lynch (1996) linked a travel cost model of recreational water use in the New Mexico's Rio Chama basin to an integrated model that optimizes the economic performance of water allocations across upstream hydroelectricity production and instream and downstream recreation demands. Subsequently, a similar approach was applied by several other studies (see for example van den Bergh et al., 2001; Ward and Booker, 2003; Ward and Pulido-Velazquez, 2008; Gürlük and Ward, 2009; Bryan et al., 2010).

A key limitations of the integrated approach used by the previous studies is that it evaluates the benefits and costs of a specific (or multiple) water management or optimization scenario(s). For example, Ward and Lynch (1996) compared the economic benefits and costs of an optimal versus a historical water management plan. Ward and Booker (2003) evaluated ‘with and without minimum stream flows (50 cubic feet per second)’ scenarios to assess the economic benefits of the endangered Rio Grande silvery minnow conservation. This scenario specific approach does not generate a unit value of environmental flows that can be used to assess the net marginal benefits of reduced water extractions under a generic policy context. We respond to these technical and policy challenges by developing a framework that generates a unit price of increased environmental flows. This unit value approach has two advantages compared to the scenario specific approach. First, it can be used to evaluate any generic policy scenario which involves trade-offs across extractive and non-extractive water uses. Hence, it allows a large degree of flexibility in water policy analysis. Second, it can be used to assess economic efficiency (instead of cost-effectiveness)[1] of water allocation decisions by applying the principle of equimarginal value which requires the marginal (incremental) value per unit of water used to be equal across all uses.

The integrated model presented in this paper considers the potential ecological benefits achieved by increasing flows in a wetland ecosystem. Our method combines a hydro-ecological model of a wetland with economic, stated preference models of household willingness to pay for an improvement in environmental attributes. First, we employ hydrological and ecological response models to project the likely outcomes of delivering environmental water to an ecological asset (e.g. wetland) by relating hydrological parameters (e.g. inundation duration) to species water requirements. Second, we use stated preference valuation studies to estimate nonmarket values associated with wetland ecosystem protection. These valuation studies use a hypothetical market or referendum with respondents to elicit

[1] Efficiency involves achieving an outcome with highest possible benefit while cost-effectiveness implies generating an outcome with least possible cost. Efficiency and cost-effectiveness do not necessarily lead to same outcomes.

household preferences for ecological/environmental attributes. Our model thus helps to estimate the economic value of an additional unit of water in the environment (i.e. unit value of environmental water). Such a price allows quantitative monetary comparisons about the marginal benefits of reallocating water to the environment from extractive uses.

Our contribution is three-fold. First, we provide a general framework that combines both hydro-ecological and economic modelling to valuing environmental flows that can be applied in any river basin where there is sufficient data. Second, we demonstrate how the framework can be applied to generate unit value of environmental water using data and models developed for the Macquarie Marshes, a wetland of global significance located in the Murray-Darling Basin of Australia. Third, we demonstrate how the estimated values can be used to evaluate the trade-offs between water extractions for agriculture and stream flows by drawing upon actual data from the Australian water market.

The remaining article is organized as follows: Section 2 presents an overview of the existing integrated modelling approaches. A description of the case study area is presented in Section 3 followed by our methodological framework in Section 4. Section 5 presents the integrated modelling results. Section 6 demonstrates how these results can be used to compare extractive and non-extractive values. Sections 7 and 8 present discussions and concluding remarks respectively.

2. Literature

The need for integrated modelling to promote transparency and efficiency in water resource management has received a significant interdisciplinary attention in recent times. A majority of the integrated water management studies combine either 'hydrologic and economic' or 'hydrologic and ecological' aspects of water resources systems (see Harou et al., 2009 for a review). Integrated hydro-economic models (e.g. AQUARIUS) allow decision makers to evaluate the physical and economic impacts of existing and alternative structural measures, changes in temporal and spatial allocation of flows among competing water uses subject to environmental and institutional restrictions (Diaz et al., 2000; Brown et al., 2002). These models do not explicitly account for ecological responses. Integrated hydro-ecological models, on the other hand, estimate the ecological effects of altering water management strategies by combining ecosystem response models with river hydrology subject to existing infrastructure (e.g., weirs, regulators) and reservoir releases (e.g. Higgins et al., 2011). These models identify the flow and operational regimes that achieve an optimal trade-off between ecological health and human needs. The economic aspect received little attention in these models.

Integrated 'hydro-ecological and economic' models have rarely been developed and applied in environmental flow management decisions although they have been applied in other contexts. For instance, Ward and Pulido-Velazquez (2008) applied an integrated biophysical, hydrologic, agronomic model to assess the likely impacts of (irrigation) water conservation subsidies in the Upper Rio Grande Basin of North America. Their study showed that conservation subsidies are in fact likely to cause water depletion through increased water extraction. The handful of studies that applied integrated hydrology, ecology and economics model for environmental flow analysis have evaluated alternative land-use and infrastructural investment/management scenarios (see for example Ward and Lynch, 1996; van den Bergh et al., 2001; Ward and Booker, 2003; Bryan et al., 2010). Van den Bergh et al. (2001) applied a linked spatial hydrological, ecological and economic model in the floodplain of river Vecht (the Netherlands) to evaluate the economic benefits of three alternative land-use patterns (i.e. agriculture, nature conservation, recreation). Bryan et al. (2010) applied a hydro-ecological and economic model in the River Murray floodplain in South Australia. Their model generated a decision support tool which can identify and rank a range of cost-effective infrastructure investments and a plan for their operation specifying where and when to capture and release water in riparian ecosystems.

3. Study Area: The Macquarie Marshes

The Macquarie Marshes, located on the Macquarie River in New South Wales (NSW), Australia, are the largest of many freshwater wetlands of tributary rivers in the Murray-Darling Basin. The Marshes are a well-known site for breeding of colonial waterbirds (e.g. straw-necked ibis, intermediate egrets and the endangered Australasian bittern) and support a diverse mosaic of vegetation types including the iconic *Eucalyptus camaldulensis* (River red gum). About 10 percent of the Marshes is listed as a wetland of international importance under the Ramsar Convention.

Flows in the Macquarie River catchment have been regulated since 1896, with heavy regulation of flows in the lower Macquarie River and Macquarie Marshes since the installation of Burrendong Dam in 1967 (Kingsford, 2000). This dam has significantly altered the flow regime on the lower Macquarie River by reducing the frequency of large- and medium-sized floods and eliminating many periods of very low or zero flow (Ralph and Hesse, 2010). Remaining flows are further controlled by a series of weirs, regulators, bypass canals, earthen embankments and irrigation channels that aid the diversion and abstraction of water for agricultural, industrial and domestic purposes. The largest benefit of the dam goes to irrigated agriculture, particularly to the water-intensive high-profit cotton industry. The gross value of irrigated agricultural production (GVIAP) in the Central West region of NSW where the Macquarie catchment is located was A$196 million in 2010–11 (ABS, 2012)[2].

Evidence suggests that the regulation of flows in the Macquarie Marshes has affected the ecological integrity of the aquatic systems by changing the food sources and altering waterbird habitat and breeding conditions (Sabella, 2009). Although some waterbird species are adapted to permanent wetlands, many species rely on large flood events across inland Australia and the associated increases in food abundance during these times to support breeding and fledgling recruitment. Extensive regulation of inland systems including the Marshes have been attributed to significant declines in the distribution, diversity and breeding of waterbirds over recent decades (e.g. Roshier et al., 2001; Kingsford et al., 2004; Kingsford and Thomas, 2004). Likewise, the distribution, diversity and composition of vegetation communities have also experienced substantial decline both in quality and quantity (Sabella, 2009).

In NSW, environmental water is in the form of entitlements or rule-based allocations. Water sharing plans developed by the NSW Office of Water (NOW) define rules for sharing water between the environment and extractive users with the purpose of replicating components of the natural flow regime. Entitlements issued by the NOW can be high security, general security or supplementary licences. For each water year (July to June), high security entitlements provide a high degree of reliability in terms of annual water allocations and represent about three per cent, by volume, of the total amount of water on issue to regulated high and general security entitlements in the Macquarie region of NSW (National Water Commission, 2013). By contrast, general security entitlements are low reliability such that

[2] GVIAP data were not available for Macquarie catchment.

the probability of an entitlement holder receiving the full allocation every year is much less than with a high security entitlement. Supplementary licences entitle license holders to access additional water, but typically only during times when natural inflows to dams may result in over-spilling or high flows in regulated rivers.

Environmental water licenses in the Macquarie and Cudgegong Regulated Rivers Water Source, which is used to deliver environmental water to the Marshes, owned by the NSW Office of Environment and Heritage (OEH) exceed 160 gigalitres (GL) of general security entitlements; general security entitlements held in Burrendong Dam total 630 GL. General security entitlements have experienced an average 55 percent allocation over the last 20 years in the Macquarie and Cudgegong Regulated Rivers (MDBA, 2012). An additional 7.7 GL of supplementary licences are held by the NSW OEH. OEH is the agency responsible for the delivery of environmental water in the Macquarie River.

4. Methodological Framework: Integrated Hydro-ecological-economic Model

This section describes the framework of the integrated hydro-ecological-economic model. Figure 1 summarises the key methodological steps. First, the hydro-ecological models are used to predict response of key ecological attributes to different watering scenarios. Outputs from the hydro-ecological modelling are used to estimate changes from a baseline regime to alternate watering regimes. The economic values of these changes are estimated by a direct non-market valuation study (a primary survey). In the absence of a primary survey, an environmental value transfer method can be employed. This method transfers nonmarket values from a study site (where the primary survey was conducted) to a policy site (the site where the value is transferred) while accounting for differences between the two sites (Johnston and Rosenberger, 2010). In this paper we demonstrate, for the Macquarie Marshes, how the integrated hydro-ecological-economic approach can be employed using either a primary study or benefit transfer techniques.

INSERT FIGURE 1 HERE

4.1. Hydro-ecological modelling

The hydro-ecological modelling step shown in Figure 1 uses a hydrology model to simulate a daily time-series of flow and inundation across the wetland system for a given climate sequence and river management scenario. The time-series outputs are used in the IBIS[3] decision support system (DSS) (Merritt et al., 2009) as input to event-based ecological response models for key vegetation and waterbird species.

The NOW developed the Integrated Quantity and Quality Model (IQQM) of the Macquarie Marshes. The spatial configuration in the IQQM model of the Marshes reflects hydrological behaviour (Thomas et al., 2011), current and historical distribution of vegetation communities (Bowen and Simpson, 2009), mapped waterbird breeding sites (Kingsford and Auld, 2003) and location of ecological assets and Ramsar sites. Twenty-four storages (or zones) are defined in both IQQM and the DSS (Figure 2). The inundation area time-series from IQQM are used in the DSS to define flood events across the period of simulation for each storage. A flood event begins when the inundation area exceeds a threshold area and continues until the inundated area falls below a defined threshold area. Hydrological attributes for each flood event drive the event-based ecological response models (ERM) of 12 waterbird species and seven vegetation species. Depending on the modelled species, ecological response is defined

[3] Note that IBIS is not an acronym.

by up to five hydrological attributes: the duration, depth, timing, the rate of fall of a flood event, and the time since the last flood event (the ‘interflood dry period’).

INSERT FIGURE 2 HERE

The ERM in the Macquarie Marshes IBIS DSS are probabilistic habitat suitability models that predict the likelihood of a flood event meeting the water requirements needed to provide habitat conditions that are suitable for breeding of waterbird species, the maintenance and survival of vegetation species and the reproduction and recruitment of vegetation species (Fu et al., 2011). The waterbird models do not predict the extent of breeding (e.g. the number of breeding pairs) or the level of success (e.g. fledgling recruitment). The extent of waterbird breeding depends on a number of other local and continental scale factors which makes it hard to predict the actual occurrence or size of a breeding event given ‘suitable’ conditions. Similarly, the vegetation models do not predict actual ecological response (e.g. crown vigour, mortality or vegetation extent). The outcomes of the models for a flood attribute (e.g. flood duration) and overall habitat condition (e.g. suitability for waterbird breeding) are a probability distribution across three states: poor, moderate and good. A good habitat condition is defined for a given flood attribute when the event matches the ‘ideal’ water requirement for a species. Poor habitat occurs when a flood attribute is outside the conditions that support breeding of a particular waterbird species or when the flood attribute adversely affect survival of a vegetation species or its capacity to reproduce.

For the purpose of this paper, the probability distributions for each flood event are converted into a relative habitat condition (RHC) score where the poor, moderate and good states are assigned an ordinal value of 0, 1 and 2 respectively. A weighted average is calculated and scaled to between 0 and 1 according to

$$\text{RHC} = ((0 \times \Pr[\text{poor}]) + (1 \times \Pr[\text{moderate}]) + (2 \times \Pr[\text{good}]))/2 \quad (1)$$

For an event, the best habitat condition (100% likelihood of being in the ‘good’ state) has a RHC of 1 and the worst habitat condition (100% likelihood of being in the ‘poor’ state) has a RHC of 0. The value of RHC which can be considered to be an acceptable threshold for defining the suitability of a flood event is subjective. Three RHC thresholds are tested in this paper: 0.6, 0.7 and 0.8. With larger RHC values, more of the input hydrological parameters in the ecological response model fall within the ‘ideal’ range for a particular species.

4.2. Non-market valuation of ecological attributes

Nonmarket values of waterbird breeding events at the Marshes are obtained from Morrison et al. (1999) and for the quality of vegetation habitat from Whitten and Bennett (2000). Both studies used the choice experiment technique of stated preference valuation method. A choice experiment constructs a hypothetical market by presenting respondents with a sequence of choices between alternative policy or management scenarios that are described by a number of attributes. These attributes have multiple levels that differ among the alternatives. Respondents are asked a series of questions in which a unique set of alternatives is presented each time. Typically, respondents are asked to pay (as a fee or tax) to secure the environmental improvements. Individual willingness to pay (or implicit price) for each attribute is estimated by computing the compensating surplus – the amount of income paid or received that leaves the households at the initial level of well-being.

4.2.1. Estimating economic value of waterbird breeding using a primary study

The choice experiment study by Morrison et al. (1999) estimated nonmarket benefits of improved ecological health of the Marshes using four attributes: irrigation-related employment, wetland area, frequency of waterbird breeding and abundance of endangered and protected species. The status quo option would reduce the size of the Marshes by reducing its size and ecological integrity. The alternative was to increase water allocation to the Marshes through the purchase of water licences from farmers. This would result in an increase of both wetland size (up to 2,000 km^2) and quality (higher frequency of waterbird breeding from once every 4 years to up to once every year and abundance of endangered and protected species from 12 to up to 25 species). The experiment also accounted for the possible negative impact of increased stream flows through the 'irrigation related (un)employment' attribute. The estimated job losses under alternative water allocation regime ranged from 50 to 150.

Three hundred Sydney households were interviewed during October 1997 using drop-off-pick-up survey mode. Respondents were presented with five choice questions each having three alternatives: the status quo, and two different wetland protection options. Respondents were told that the government did not have sufficient money to purchase additional water. As a result, households were asked to pay a one-off[4] levy ranging between A$20 to A$150 in 1998.

The trade-offs made by the respondents across attributes can be observed from the choices in each choice question. These trade-offs are monetised using the monetary attribute to estimate respondents' willingness to pay for each non-monetary attribute. The estimated monetary values are called implicit prices that the respondents are willing to pay on average for a unit change in each attribute. The estimated implicit (unit) price of waterbird breeding frequency at the Macquarie Marshes was A$25. The 90 percent confidence interval of the implicit price

[4] A one-off payment is the sum of the present values of a stream of future payments an individual is willing to sacrifice for the environment.

was A$15–30. Adjusting for inflation for the period of 1998 and 2012 (Reserve Bank of Australia, 2013) generates a mean implicit price of A$36 (90% CI: A$21–A$43) at 2012 prices. Note that the Morrison et al. (1999) study was conducted before the Millennium Drought of 2001–2009 (the longest period with below median rainfall in southeast Australia since 1900). A valuation survey after the drought is likely to have generated higher implicit price for waterbird breeding as the habitat condition for bird breeding experienced significant stress because of the drought (MDB, 2013). Therefore, the inflation adjusted implicit price (A$36) is most likely to be an underestimation of society's current willingness to pay to improve the waterbird breeding conditions.

4.2.2. Estimating economic value of healthy wetland via environmental value transfer

To illustrate that a direct survey, although ideal, is not required for a hydro-ecological and economic model, we use the example of a study that estimates the economic values of healthy wetlands for Murrumbidgee River floodplain, NSW, Australia (Whitten and Bennett, 2000). Whitten and Bennett (2000) interviewed 2,800 households in the Murrumbidgee region of NSW, the Australian Capital Territory and South Australia. As with Morrison et al. (1999), their study presented respondents with three alternatives in each choice question: a status quo alternative and two wetland protection strategies that required a one-off levy in return for environmental improvement. The positive environmental impacts involved increases to the area of healthy wetlands (from 2,500 ha to up to 12,500 ha), woodland bird populations (from 40% to up to 80%) and native fish population (from 20% to up to 60%). Like Morrison et al. (1999), Whitten and Bennett (2000) also included the unemployment attribute (i.e. farmers leaving) to reflect the negative impacts of environmental water enhancement scenarios on farm viability.

Area of healthy wetland is the key attribute of our interest as the economic value of this attribute can be integrated to the IBIS DSS outputs that measure habitat suitability condition for native vegetation species under a set of assumptions. Whitten and Bennett (2000) estimated an implicit (unit) price of healthy wetland at A$11 per thousand hectares at the Murrumbidgee River floodplain. To make this implicit price useful at the Macquarie Marshes, we employ environmental benefit transfer that uses data at the Murrumbidgee River floodplain (the study site) to generate values at the Marshes (the policy site).

Environmental value transfer applies one, or a mix, of the following approaches: unit-value, marginal-value, value-function and meta-value function (Johnston and Rosenberger, 2010; Morrison and MacDonald, 2010). In unit-value transfer a single value estimate from one study or an average of multiple value estimates from several studies is transferred without adjustment. Marginal-value transfer allows value transfer with adjustment to site differences. In value-function transfer a value function is transferred by accounting for differences in observable site attributes such as income and education. In meta-value-function transfer, the value estimates obtained from a number of relevant studies are analyzed and synthesized to control for differences in study design and sites.

Marginal-value transfer approach is commonly used when values are derived for attributes of a good, such as results of a choice experiment (Morrison and MacDonald, 2010). The advantage of this approach is that it is possible to more closely match the changes in environmental quality at the study and policy sites. Regardless of the approaches used, benefit transfer may result in over- or under-estimation of values (i.e. transfer error) (Rosenberger and Loomis, 2003). Uncertainty in the transferred value caused by transfer error can be accounted in a number of ways (see Akter and Grafton, 2010 for a review). The risk and simulation (R&S) approach is one of the most dynamic ways of accounting for transfer error in benefit transfer applications (Akter and Grafton, 2010). The R&S approach assigns a range of possible error distributions to the value to be transferred and applies Monte Carlo simulations to account for potential errors in the evaluation of benefits of a policy.

Transfer accuracy is largely determined by site similarity – including similarity over populations, resources, markets and other site attributes. Navrud and Brouwer (2007) propose an error of ±40 percent when the study and policy sites are similar and an error of ±100 percent in case of a relatively larger degree of site dissimilarity. The Murrumbidgee River floodplain is about 650 km south of the Marshes. They are both located in the Murray-Darling River Basin and are geographically and environmentally quite similar. Native vegetation is a key environmental asset in both sites (Morrison and MacDonald, 2010). However, a substantial proportion of the Macquarie Marshes network is listed as Ramsar site. This causes a significant difference between the two sites with regards to their conservation significance: The Marshes provide more valuable ecological functions than the Murrumbidgee River floodplain. A direct study at the Marshes, most likely, would have generated larger values for healthy wetland. Offsetting these factors is, public access to the Marshes Nature Reserve is restricted for most part of the year which reduces its use value through recreational opportunities (OEH, 2013). The Murrumbidgee River floodplain, on the other hand, is much more accessible to the general public which may generate a higher use value for healthy vegetation at the study site. The impacts of these possibilities can be accounted for through an uncertainty analysis in the value transfer method using the R&S technique (Table 1).

INSERT TABLE 1 HERE

A direct transfer of marginal value from the Murrumbidgee River floodplain generates an implicit price of A$15 for a thousand hectares of healthy wetland at the Marshes (adjusting for inflation between 2000 and 2012). Assuming a ±40 percent transfer error and three alternative error distribution functions, the mean implicit price of healthy wetland at the Marshes vary within A$15 to A$16 per household per thousand hectares (Table 1).

5. Results

5.1. Hydrology scenarios

Three scenarios provided by the NSW OEH are used to model ecological response. These scenarios use hydrological time series data from 1st July 1991 to 30th June 2007. The 'Current' scenario (Figure 3a) predicts wetland inundation area under 2010 rules for allocating water between the alternative uses (e.g. agriculture, environment, human consumption) of water in the Macquarie Catchment. The 'Current+104GL' and 'Current+135GL' scenarios model inundated area with an additional 104GL and 135GL of environmental water released from Burrendong Dam per water year (Figure 3b). Under the 'Current' scenario, 16 flood events are defined over the 16 year period of record with a median duration of 2.7 months. There are 18 and 15 flood events calculated under the 'Current+104GL' and 'Current+135GL' scenarios, respectively. Fewer, albeit larger, events are calculated over the hydrological record for the 'Current+135GL' scenario.

INSERT FIGURE 3 HERE

5.1. Valuing environmental flows using waterbird breeding frequency

5.1.1. Ecological response of waterbird breeding

For the analyses in this paper, an event is considered to provide conditions suitable for a species to breed if the RHC exceeds the specified thresholds. The frequency of suitable flood events under the 'Current', 'Current+104GL' and 'Current+135GL'environmental watering scenarios for three threshold values are shown in Figure 4a. The Morrison et al. (1999) study outlined in Section 4.2.1 does not specify particular waterbird species or sites within the Marshes system. Thus, the calculated frequencies are based on conditions suitable for one or more of the modelled bird species in Gum Cowal/Terrigal Ck storage (S2) – the most frequently inundated part of the modelled system where waterbird species have been recorded. Over the length of the modelled record, if an event was suitable in any storage other than S2, for any of the waterbird species, conditions were suitable in S2 for one or more storages. For all RHC threshold values, the inundation results in events that are hydrologically 'suitable' for waterbird breeding.

In order to consistently map the IBIS outputs of suitable waterbird breeding events to the outputs of Morrison et al. (1999) study[5] we calculated 'suitable event frequency' that measures the number of years within which a suitable event occurs (e.g. 1 suitable event in x year). With a RHC threshold of 0.6, the 'Current' scenario has a suitable event approximately 1 in 6 years compared with 1 in 2 years for the 'Current+135GL' scenario. As the RHC threshold increases (i.e. stricter definition of suitable events), so does the difference in the frequency of suitable events between the 'Current+135GL' and other scenarios.

INSERT FIGURE 4 HERE

5.1.2. Economic valuation of environmental flows

The hydro-ecological model outputs were combined with the economic values using the following equation:

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[5] Morrison et al. (1999) measured willingness to pay for changes in waterbird breeding events from once in 4 to once in 3, 2 and 1 year.

$$MB_W = (F_{Current+X,r} - F_{Current,r}) * IP \qquad (2)$$

In Equation 2, MB_w refers to the marginal benefit of water per household, F is the frequency of suitable events, r denotes RHC threshold, X is additional stream flow and IP stands for implicit price.

Since the measurement unit of F is consistent across the economic and ecology models, their marginal changes can be integrated without any further adjustments. The difference in the frequency of suitable events for RHC threshold 0.6 between the current and the 'Current+104GL' scenarios is 0.44. Multiplying this number with the implicit price for waterbird breeding (A$36) as per Equation 2 generates A$17 value for 104 GL stream water. This number indicates the average amount a Sydney household is willing to pay to improve the condition of waterbird breeding along the Marshes. Dividing the one-off implicit price by the amount of additional water to the stream (i.e. 104 GL) we obtain the value of 1 GL water equivalent to A$0.17 per Sydney household.

Figure 4b presents a range of such values obtained under the 'Current+104GL' and 'Current+135GL' scenarios and three RHC thresholds (0.6, 0.7 and 0.8). Both scenarios generate marginal values that vary between A$0 and A$2. The marginal values obtained under the 'Current+135GL' scenario are substantially higher than the values obtained under the 'Current+104GL' scenario. This is because a larger volume of environmental flow is likely to generate higher frequency of suitable events for waterbird breeding for any given RHC. Hence the economic value of water increases with an increase in the volume of stream flow. The difference in the marginal values of water between these scenarios increases with an increase in RHC threshold implying that a larger volume of environmental flow not only generates higher number of suitable events but also produces better quality events.

5.2. Valuing environmental flows using vegetation habitat suitability as a surrogate for wetland health

5.2.1. Ecological response of vegetation

The IBIS DSS does not include a directly comparable measure of healthy wetlands which comprises wetland function as well as the health of flora and fauna communities. To demonstrate how environmental value transfer methods can be applied within the modelling framework, the healthy wetlands attribute in Whitten and Bennet (2000) is linked to the habitat suitability of flood events for maintenance and survival of flood-dependant vegetation species.

Water requirements for species maintenance and survival vary between the vegetation species modelled in the Macquarie IBIS DSS. For example, *Eucalyptus camaldulensis* (River red gum) can tolerate much larger inter-flood dry periods than wetland grass species like *Paspalum distichum* (Water couch) and *Phragmites australis* (Common reed) (Rogers and Ralph, 2011; Appendix 1). The analyses in this paper use modelled ecological outcomes for five wetland species: *Acacia stenophylla* (River cooba), *E. camaldulensis, Muehlenbeckia florulenta* (Lignum), *P. australis* and *P. distichum*. As with the waterbird breeding analyses, the RHC statistic (Equation 1) is used to assess the impact of the scenarios on vegetation habitat and the same threshold values (0.6, 0.7, and 0.8) are used to demonstrate the sensitivity of ecological and economic results to the selection of this threshold.

Although the water requirements of the vegetation species considered in this study vary, the differences between the three hydrologic scenarios are similar. This is demonstrated in Figure

5a for three vegetation species present in the eastern (S2 and S5 to S7 in Figure 2) and northern storages (S15 to S20, S22 and S23 in Figure 2). The eastern storages are inundated more frequently than the northern storages. As a result, more suitable events in the eastern storages are calculated over the 16-year time-series for each scenario. In the eastern storages, the additional water delivered under the ‘Current+135GL’ scenario, compared with the ‘Current+104GL’, increases the duration of inundation can merge consecutive flood events that were defined in the DSS for the ‘Current+104GL’ scenario. For river red gum there is little difference between these two scenarios with RHC threshold values greater than or equal to 0.7. However, lignum is less tolerant to long periods of inundation and a lower count of ‘suitable’ events is tallied for the ‘Current+135GL’ scenario regardless of the RHC thresholds. In the less frequently flooded northern storages, more ‘suitable’ events are calculated under the additional watering scenarios with better outcomes with the ‘Current+135GL’ scenario than the ‘Current+104GL’ scenario.

INSERT FIGURE 5 HERE

5.2.2. Economic valuation of environmental flows

The vegetation habitat modelled in IBIS DSS is used as a proxy for healthy wetland attribute used in Whitten and Bennett’s (2000) experiment. Equation 2 was slightly modified to adjust for the unit (i.e. hectare) used to measure the vegetation attribute. The average storage area occupied by each species for each event over the modelled hydrologic period is computed to combine with the economic value of environmental water. More specifically,

$$MB_W = \frac{\sum_s [(N_{Current+X,r,v} * Area_s) - (N_{Current,r,v} * Area_s)]}{\sum v} * IP \qquad (3)$$

In Equation 3, N refers to the number[6] of suitable events, v stands for vegetation species and s denotes storage within the Marshes. The key underlying assumptions that led to the formulation of Equation 3 are: (1) Improvements in habitat condition for vegetation species is a strong indicator of wetland health; (2) Area of healthy vegetation is determined by solely by water (other factors such as grazing pressure is ignored); (3) Equal weighting across all vegetation species; and (4) 'Suitable' event means conditions for vegetation species are good across the whole storage. These assumptions are highly simplistic and do not account for potential non-linearity in the biophysical system, but were necessary for the demonstration of the value transfer approach.

These results are aggregated for the eastern and northern storages to explore the implication of variability in inundation patterns across the Marshes. Figure 6 presents the estimated marginal values of water. For the eastern storages, under the 'Current+104GL' scenario, the additional water produces suitable vegetation conditions. Thus, it generates a positive marginal value varying between A$0.5 and A$1.4 (Figure 6(a)). Unlike the case of waterbird, the 'Current+135GL' scenario substantially reduces the marginal value of water for vegetation species in the eastern storages. This means that more water does not necessarily produce better outcomes. As these storages remain inundated for an extended period of time, additional water is likely to reduce the suitability of a flood event for the maintenance and survival of a vegetation species. In the less frequently inundated northern storages, additional water yields positive values for both 'Current+104GL' and 'Current+135GL' scenarios

[6] Note that there is a subtle difference between frequency (*F*) and number (*N*) in Equations 2 and 3. F in Equation 2 refers to the number of years it takes for one suitable event to occur, e.g. 1 suitable event in 5 years. *N* in equation 3 refers to the total number of suitable events occurring within a given period of time, e.g. 5 suitable events in 16 years.

(Figure 6(b)). Like the case of waterbird, the marginal value of water is substantially higher under the 'Current+135GL' scenario compared to the 'Current+104GL' scenario implying that a larger flow generates higher frequency of suitable events. However, the difference in marginal value of water across the scenarios decreases as the RHC threshold increases. An RHC of 0.8 reflects the stricter definition of what constitutes a good event under where most or all hydrological parameters are within the ideal range for the modelled vegetation species. These events are relatively infrequent and although more common under the alternate watering scenarios the difference in the count of suitable events from the current scenario is less than if a lower threshold RHC is used.

INSERT FIGURE 6 HERE

6. Comparing Extractive vs Non-extractive Water Uses

We illustrate how the estimated economic value of environmental water may be used to make efficient water allocation decisions across extractive versus non-extractive uses. The economic approach of efficient water allocation is 'the principle of equimarginal value'. It requires the marginal value of water used to be equal across all uses. If marginal values across competing users differ, greater benefit will be achieved by transferring water from a lower- to a higher-valued use. When equality of marginal values is achieved across all users, further redistribution can make no user better off without making another user worse off.

A key step to applying the equimarginal principle is to aggregate the sample mean willingness to pay across the population of interest. Aggregation errors are reduced the more representative is the sample of the overall population and proper consideration is given to the non-response bias in the sample. Mitchell and Carson (1989) propose the following technique to control for non-response bias:

$$\overline{WTP} = \sum_{m} WTP + \sum_{n} \lambda(WTP) \qquad (4)$$

In Equation (4), $\overline{WTP}$ refers to the weighted aggregated willingness to pay, m is the proportion of respondents, n is the proportion of non- respondents and λ is the multiplier or weight attached to the non-respondent population. If λ =1, non-respondents have the same mean willingness to pay as respondents, and if λ =0, non-respondents have zero willingness to pay.

To illustrate, we use the willingness to pay estimates obtained from the Morrison et al. (1999) study which had a relatively higher response rate (76%) compared to the Whitten and Bennett (2000) study (30%). First, we average the sample average marginal benefit (MB_W) obtained for three RHC values under the 'Current+104GL' and 'Current+135GL' scenarios respectively (i.e. A\$0.13 and A\$1.43). These values represent an upper and lower bound of sample average one-off willingness to pay for a GL of environmental flows. Assuming λ =0 the aggregation of the upper and lower bounds of marginal benefit over 1.7 million Sydney households generates A\$0.15 million (lower bound) and A\$1.70 million (upper bound) aggregate marginal benefit under the 'Current+104GL' and 'Current+135GL' scenarios respectively. Note that these are the most conservative estimates for three reasons. First, these estimates do not account for non-market values of other ecological benefits of increased stream flow such as healthy vegetation and fish population. Second, non-respondents are likely to have some positive values for improved waterbird breeding condition. Finally, although Sydney households are the most relevant population for the study, households in rest of Australia are also likely to have some positive willingness to pay. Given that this exercise is for demonstration purpose only, we present one of the many possible cases of aggregation.

These benefits are compared against general security entitlement prices observed for the Macquarie region during 2007–2013 (Department of Sustainability, Environment, Water, Population and Communities, 2013). The entitlement prices varied between A$0.91 million and A$1.26 million with an average of A$1.1 million per GL between 2007 and 2013. Incidentally, these values are quite similar to average GVIAP (A$0.99 million) per GL of water used in agriculture in the Central West NSW in 2010–11 (ABS, 2012). However, the GVIAP estimate refers to the gross value of agricultural commodities produced with the assistance of irrigation (ABS, 2012). It does not refer to the value that irrigation adds to production (ABS, 2012). Hence, the ABS strongly cautions against using the GVIAP figure as a proxy for determining the highest value water uses. Further, the per GL GVIAP is an average rather than marginal value and an estimated or implicit value of water rather than explicit price determined at a point in time by the buyers and sellers.

Accurate estimation of irrigation's value addition to agricultural production is difficult. In a well-functioning and perfectly competitive market, price transmits information about the relative scarcity and value of water resource in use. Hence, the historical entitlement price is a fair representation of willingness to pay (or marginal value) of water in agriculture. Therefore an understanding regarding economic efficiency of water allocation along the Marshes can be obtained by comparing the entitlement price with the economic value of water in the stream. As shown in Figure 7, the marginal value of per GL of water in the agriculture exceeds the lower bound but lies below the upper bound of the marginal value of water in the environment. This means, under the 'Current+135GL' scenario, a GL of water used in the environment appears to generate a higher economic value for the environment than its alternative use in the agricultural sector.

INSERT FIGURE 7 HERE

7. Discussion

The formulation of the ERM in the IBIS DSS raised three main issues when relating outputs to the economic analyses. Firstly, habitat suitability approaches are relatively simple approaches that can be easily applied to different wetland systems. However, they are of more value for individual species-based assessments than to ecosystem analyses (e.g. Lester et al., 2011). The outputs from the IBIS DSS are species-specific while the choice question did not specify particular species. For the waterbirds analysis, we assumed that if conditions were suitable for any of the modelled species then this could be related to the frequency of waterbird breeding. Another issue with habitat suitability models and other index based approaches is that the interpretation of model outputs can be subjective: for example, deciding what constitutes an acceptable outcome and relating indices to measurable ecological indicators (e.g. canopy cover, mortality). The latter has particular implications for the economic analyses. Lastly, assumptions were required to translate habitat suitability to the indicators of ecological condition used in the economic analyses.

The analysis in Section 5.1 assumes that good habitat suitability directly relates to waterbird breeding. While considered a reasonable assumption, this is not strictly true. While suitable hydrologic conditions are necessary for waterbirds to nest, breed and achieve fledgling recruitment, the existence of these hydrologic conditions does not necessarily mean birds will arrive in the Marshes and breed. Many other local and continental factors influence actual waterbird breeding at a site including land management, food availability and habitats in other areas. The use of habitat suitability for the maintenance and survival of vegetation as a surrogate of wetlands health is more tenuous. While it is reasonable to expect correlations between habitat suitability and actual condition of vegetation and between vegetation condition and wetland health, measuring or predicting wetland health is more complex. Given this, the analyses presented in this paper demonstrate the transfer of environmental values within a hydro-ecological and economic modelling framework rather than provide a rigorous analysis of the value of environmental water for wetland health. To undertake the latter using the IBIS DSS would require the complex task of developing alternate ERM that capture how wetland function processes and the diversity and structure of ecological communities change under different watering scenarios.

There is a high degree of uncertainty associated with hydrology and ecological predictions and economic value estimations. River flow and ecological response models are potentially useful tools for assessing the ecological impacts of land and water management interventions and climatic conditions. Hydrological models can often represent adequately in-stream hydrological processes although can struggle to predict the wetting and drying patterns of floodplain wetlands. The latter is particularly relevant in inland wetland systems in Australia like the Macquarie Marshes where complex wetting and drying patterns across the landscape are governed by rivers of naturally high variability and highly complex interactions between rivers and the larger floodplain and associated wetlands. These interactions ultimately determine the distribution and response to flooding of vegetation communities and biota. Uncertainties associated with ecological models largely reflect limitations in ecological knowledge over time and space and limit the capacity to predict how species and ecosystems respond to environmental water allocations across a range of hydrological conditions (Colloff and Baldwin, 2010).

8. Concluding remarks

In this paper, we develop a framework to combine non-market valuations of attributes of wetland ecology to hydro-ecological and economic models so as to provide an economic valuation of a marginal increase in environmental flows. Our contribution lies in the development and application of the integrated hydro-ecological-economic framework to evaluate the net benefits of water reallocation between extractive uses and stream flows. What it provides over other modelling approaches is the ability to generate a unit price of water for extractive and non-extractive uses and, thus, allow for the possibility that water can be reallocated using the equi-marginal principle. Ultimately, if the hydro-ecological and economic methods we have developed were to be widely applied, they offer the promise of better water governance and improved environmental outcomes for ecologically stressed streams and rivers.

While our methods can be improved, our framework provides valuable information about the trade-offs between extractive and non-extractive uses. Key challenges for the wider application of our methods are the existence of suitable non-market valuation studies for riparian assets and a hydro-ecological model capable of providing likely ecological responses to changes in hydrology. Ideally, the non-market valuation study should be developed taking into account the outputs of hydro-ecological model. In practice, either non-market valuation study or the hydro-ecological model, or neither, will be available and which will require a substantial investment to be developed. The benefit of developing a coupled hydro-ecological and economic modelling capacity will depend on the potential for better decisions about whether to, and to what extent, water should be reallocated between competing extractive and non-extractive uses. The greater the water scarcity and the larger the perceived misallocation among competing water uses, the more valuable will be a hydro-ecological and economic modelling approach.

The caveats and challenges of integrated hydro-ecological-economic modelling identified in this study offer important avenues for future research. The main issues raised by this study were that the choice questions did not directly relate to outputs from an ecological response model. This has three implications for future valuation studies to ensure the compatibility of the two components and to provide a more robust and useful methodology for valuing environmental water. The first is that the nonmarket valuation studies need to consider species-specific valuation approach. Given the significant variation in water requirement across different species, the generic valuation approach poses significant challenges in integrating ecological and economic outputs. Second, future valuation studies need to be more explicit in defining the location being valued (e.g. wetland health in the North Marsh Nature Reserve) or, at least, to consider spatial outcomes in a large wetland system like the Macquarie Marshes, where the ecological outcomes of environmental water vary across the system.

Finally, uncertainties and errors in the hydrologic and ecological modelling need to be accounted for by assessing the sensitivity of the economic value calculated based on model outputs for an environmental water scenario to the parameterisation of the ERM in the IBIS DSS. There are two main parts of the ERM to which the sensitivity of the model outputs need to be assessed in further studies. These are the hydrological thresholds which define the start and end of flood events (i.e. the number of events) and the uncertainties associated with imperfect knowledge about species water requirements (i.e. the outcome of each defined event). A discussion of the sensitivity of these types of ecological response models to the knowledge source used to parameterise the model is given in Fu and Merritt (2012). This uncertainty can have significant impacts on estimated ecological outcomes and the certainty around relative benefits between alternate watering scenarios, which in turn will affect any economic valuations based on model outputs.

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Figure 1. Key steps to implementing hydro-ecological and economic model of environmental flow demand.

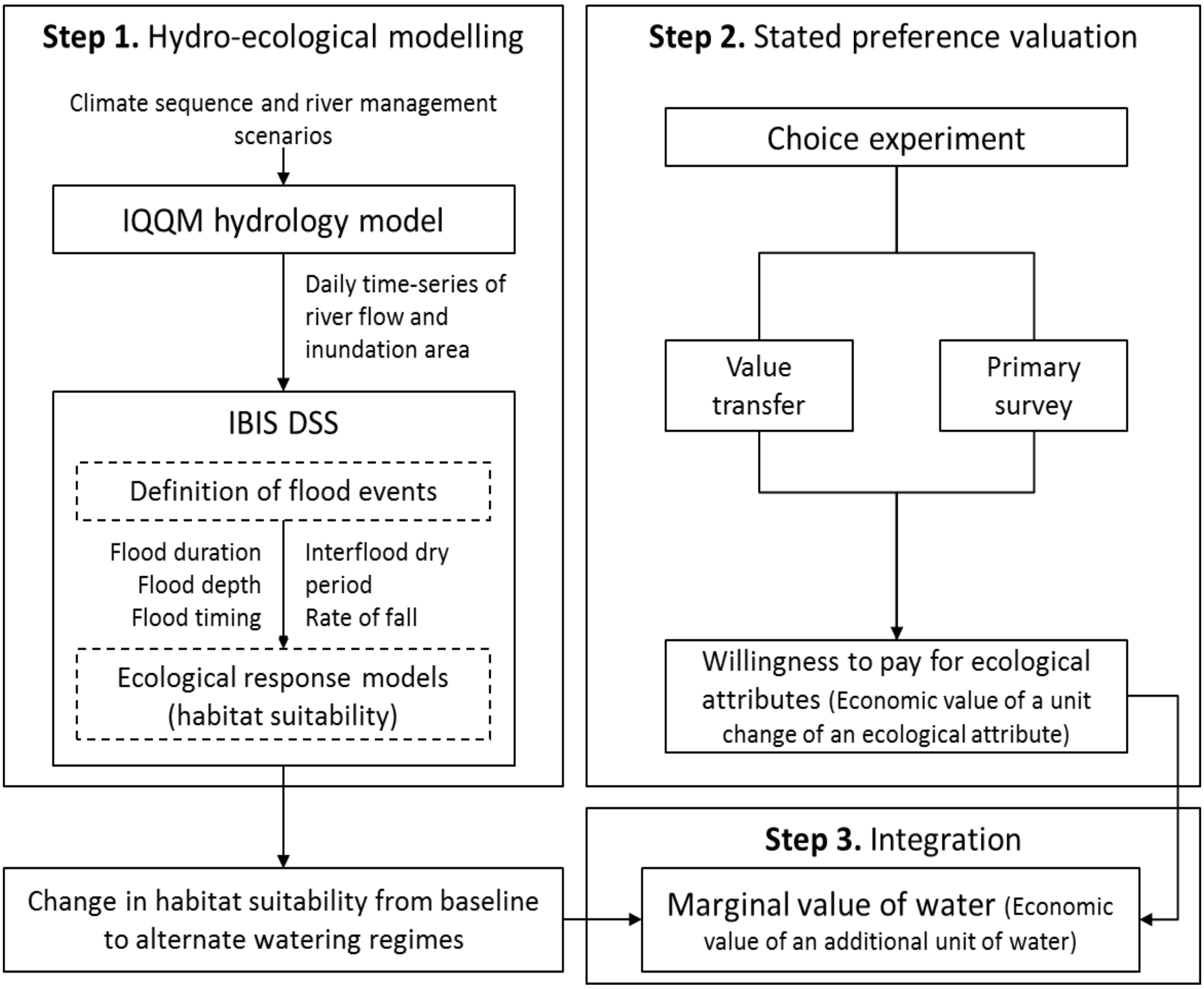


Figure 2. Storages modelled in the Macquarie Marshes IBIS DSS.

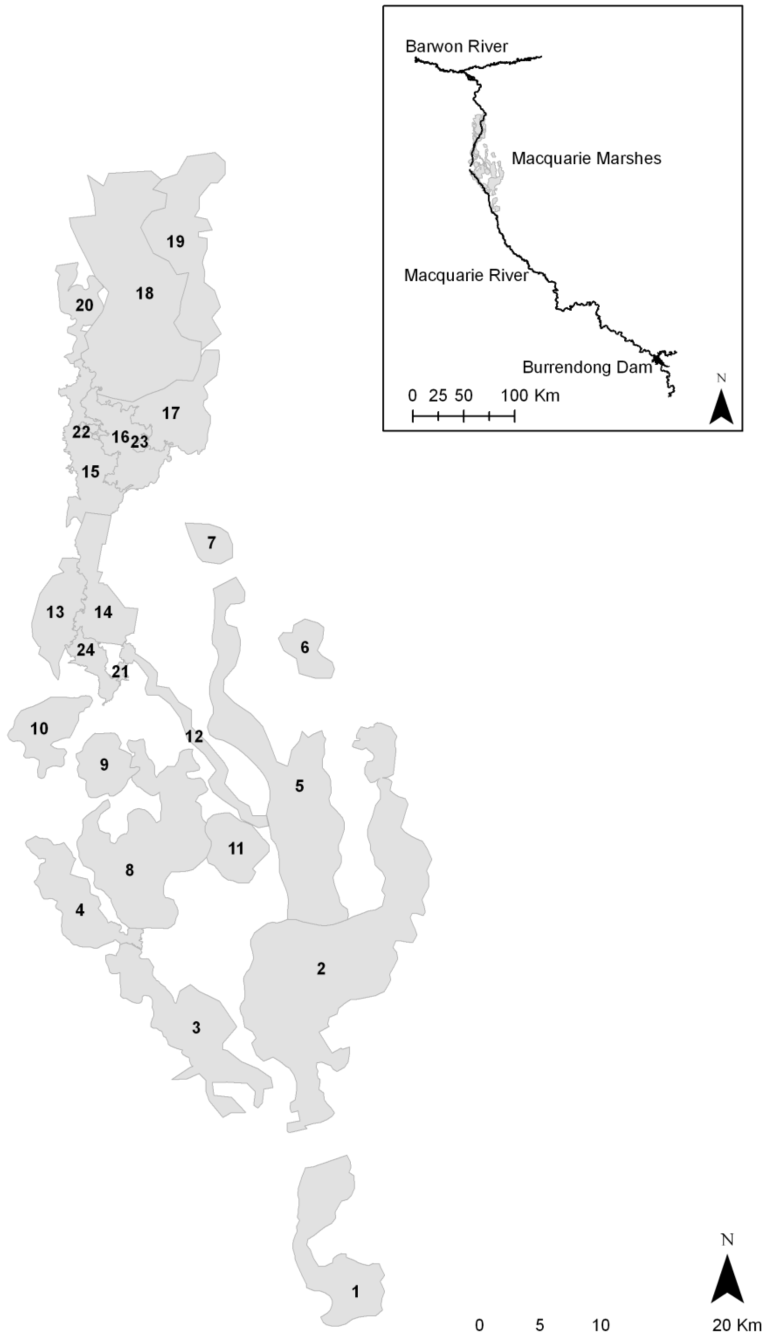
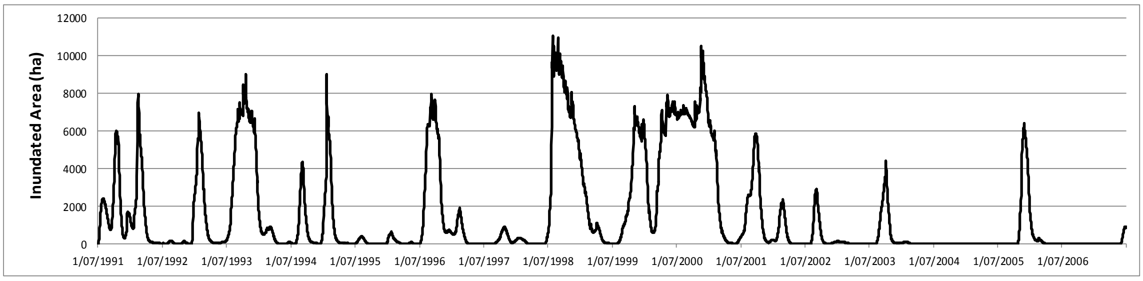
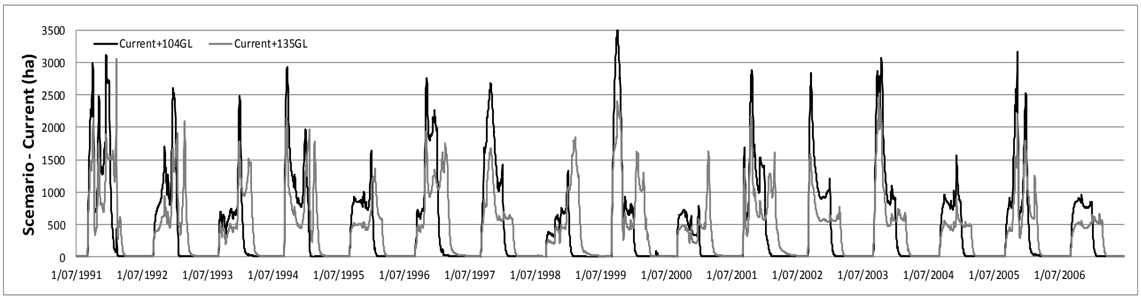


Figure 3. Hydrological scenarios used to calculate ecological response: (a) Inundated area of the Gum Cowal /Terrigal Creek storage (S2) under the Current scenario, and (b) difference in inundated area between the environmental watering scenarios and the current scenarios.



(a)



(b)

Figure 4. Valuing environmental flows using the frequency of flood events suitable for waterbird breeding: (a) frequency of suitable habitat conditions for waterbird breeding in the Gum Cowal/Terrigal Ck storage (S2 in Figure 2) and (b) economic values of environmental water estimated using water-bird breeding event.

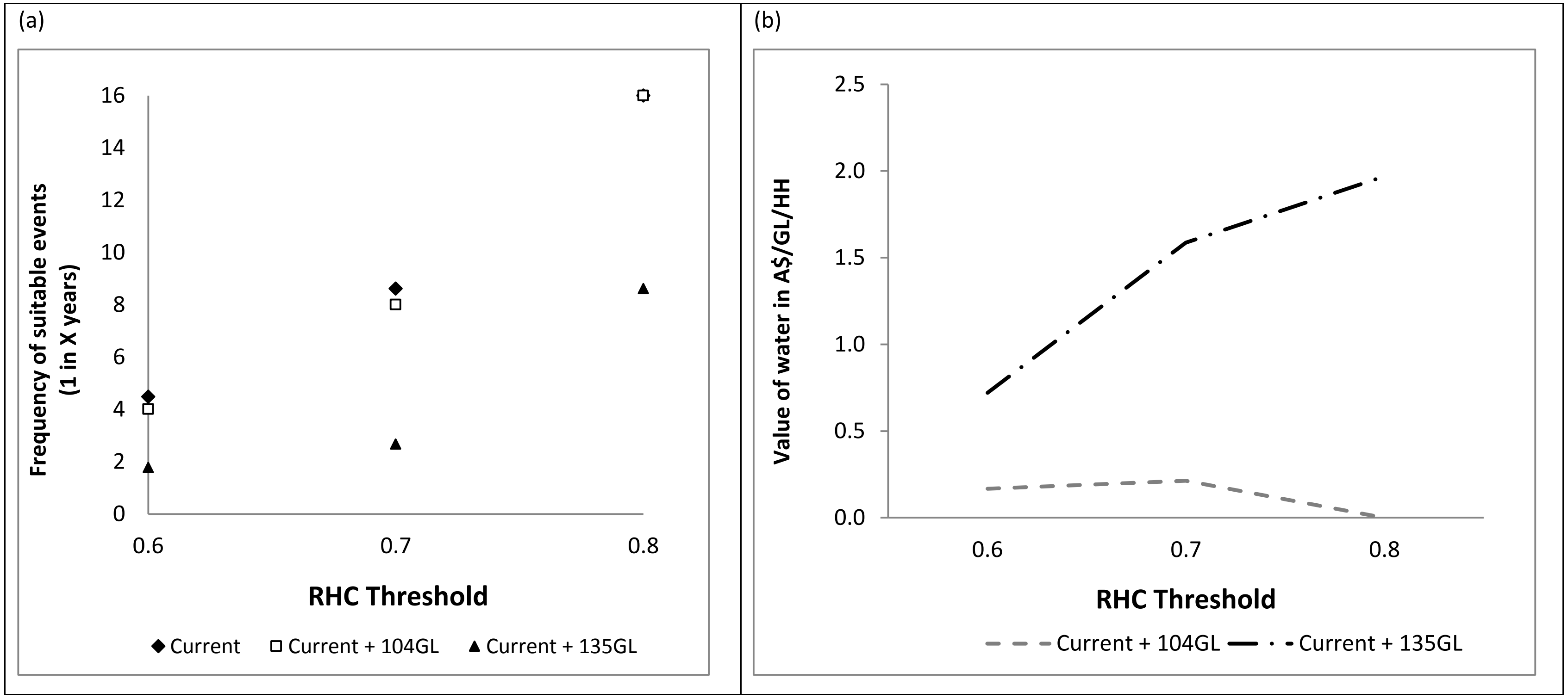


Figure 5. Number of events that are suitable for supporting the maintenance and survival of four vegetation species common to (a) the eastern and (b) the northern storages.

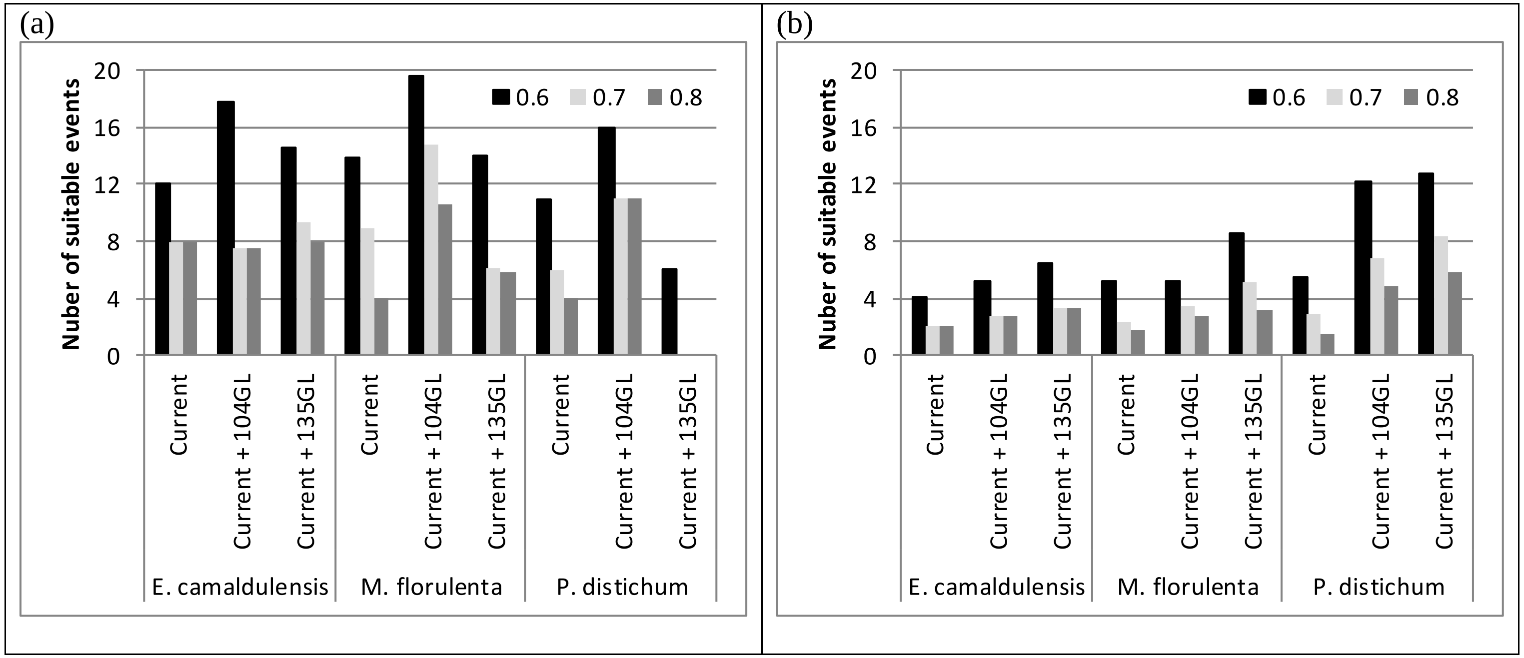


Figure 6. Economic values of environmental water estimated using area of healthy vegetation species (a) the eastern storages and (b) the northern storages

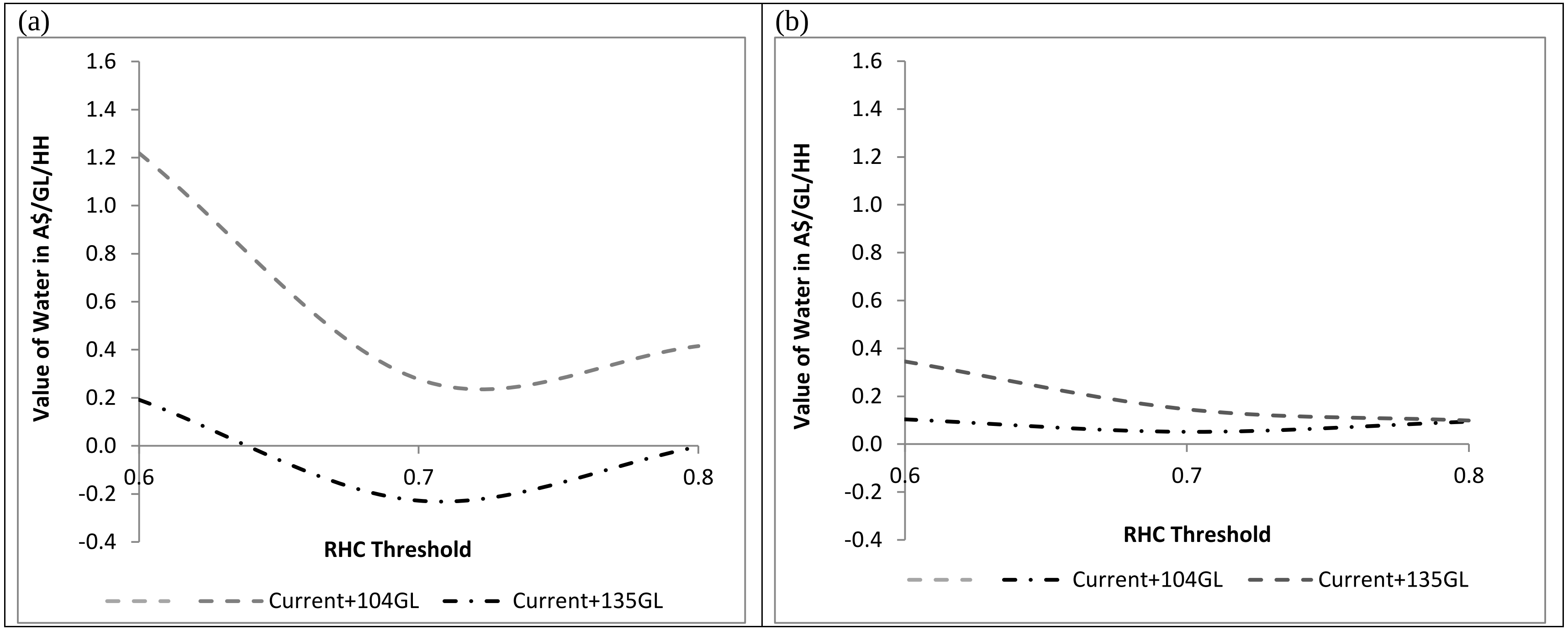
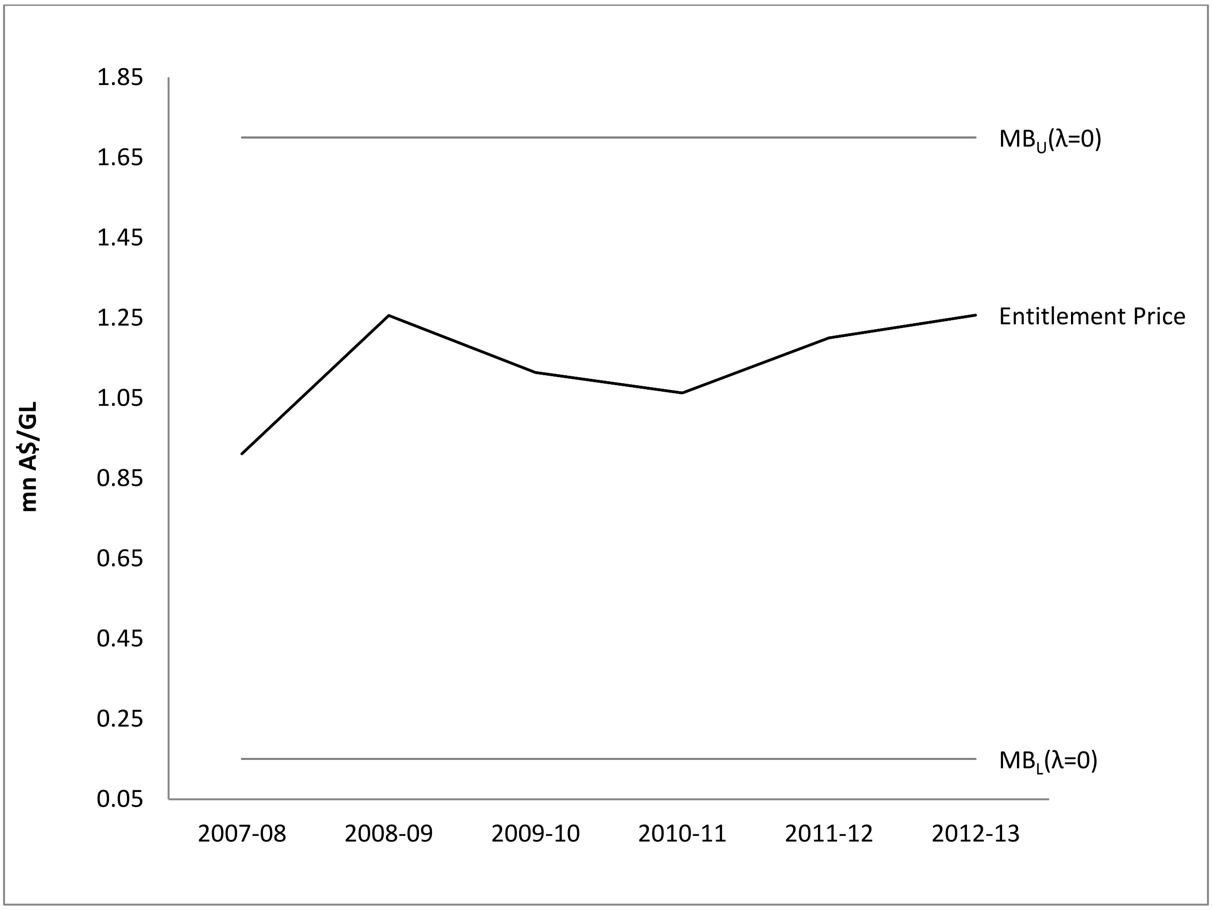


Figure 7. Marginal benefit of water across extractive and non-extractive uses in the Macquarie Marshes.



<u>Notes:</u>

- λ =0 means non-respondents have zero willingness to pay.

-MB_U is the upper bound of marginal benefit of water in the environment (i.e. the aggregated implicit price of a --GL of water obtained through waterbird breeding event under the ‘Current+135GL’ scenario).

-MB_L is the lower bound of marginal benefit of water in the environment (i.e. the aggregate implicit price of a ---GL of water obtained through waterbird breeding event under the ‘Current+104GL’ scenario).

-Entitlement price is the yearly average price of general security water entitlements traded along the Macquarie region.

-Implicit prices are aggregated across Sydney households to provide a conservative estimate.

Table 1 Uncertainty analysis for transferred value of healthy wetland through error distribution [a] function

	Triangular error distribution[b]	Histogram error distribution[c]	Discrete error distribution[d]
Mean	16	15	16
Median	16	16	15
Mode	18	17	18
95% CI	11–20	10–20	9–21
Minimum	10	9	9
Maximum	21	21	21

Notes:

[a] The choice of an error distribution function depends on the analyst's judgement of the nature of the error. In this particular case, although negative error values are quite probable (due to limited public access to the Marshes), positive values are more likely (due to the higher ecological significance of the Marshes). The error distribution functions were chosen to reflect these possibilities.

[b] Transfer error follows a triangular distribution, i.e. a continuous probability distribution function with a minimum (-0.4), maximum (0.4) and most likely value (0.2).

[c] Transfer error takes different values within the range of ±40% with different probabilities. The continuous probability distribution function (value {probability}) is (-0.4,0.4,{0.1,0.2,0.2,0.4,0.1}).

[d] Transfer error follows a discrete probability distribution within the range of ±40% with different probabilities. The probability distribution function {value (probability)} is {−0.40 (0.1), −0.20 (0.2), 0 (0.2), 0.20 (0.4), 0.40 (0.1)}.

Appendix 1

Species water requirements for selected vegetation species modelled in the IBIS DSS (Source: Rogers and Ralph, 2011)

Species	Flood Duration	Flood Timing	Flood Depth	Interflood Dry Period
Acacia stenophylla	Ideal flood duration: 1-6 months; maximum flood duration: 12 months.	Ideal flood timing: spring to early summer.	Ideal flood depth: 0-60cm.	Ideal inter-flood dry period: 1-10 years.
Eucalyptus camaldulensis	Ideal flood duration: 2-8 months; maximum flood duration: 24 months.	Ideal flood timing: winter to spring; maximum flood timing: winter to early summer.	N/A	Ideal inter-flood dry period: 5-15 months; maximum inter-flood dry period: 36 months.
Muehlenbeckia florulenta	Ideal flood duration: 1-6 months; maximum flood duration: 12 months.	Ideal flood timing: spring to early summer.	Ideal flood depth: 0-60cm.	Ideal inter-flood dry period: 1-10 years.
Paspalum distichum	Ideal flood duration: 1-2	Ideal flood timing:	Ideal flood depth: < 60 cm;	Ideal inter-flood dry period: 236

	months or 163 days in 2 years; maximum flood duration: 513 days in 2 years.	summer; maximum flood timing: spring to summer.	maximum flood depth: 200 cm.	days; maximum inter-flood dry period: 290 days.
Phragmites australis	Ideal flood duration: ~6 months; maximum flood duration: 12 months or permanent.	Ideal flood timing: spring; maximum flood timing: any.	Ideal flood depth: static: +20 to -50 cm; fluctuating: +-30 cm; maximum flood depth: 200 cm.	Ideal inter-flood dry period: few months; maximum inter-flood dry period: 12 months.