Hydrol. Earth Syst. Sci., 18, 2141–2166, 2014
www.hydrol-earth-syst-sci.net/18/2141/2014/
doi:10.5194/hess-18-2141-2014

Hydrology and Earth System Sciences
Open Access

A prototype framework for models of socio-hydrology: identification of key feedback loops and parameterisation approach

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Received: 15 December 2013 – Published in Hydrol. Earth Syst. Sci. Discuss.: 14 January 2014
Revised: 30 April 2014 – Accepted: 8 May 2014 – Published: 13 June 2014

Abstract. It is increasingly acknowledged that, in order to sustainably manage global freshwater resources, it is critical that we better understand the nature of human–hydrology interactions at the broader catchment system scale. Yet to date, a generic conceptual framework for building models of catchment systems that include adequate representation of socioeconomic systems – and the dynamic feedbacks between human and natural systems – has remained elusive. In an attempt to work towards such a model, this paper outlines a generic framework for models of socio-hydrology applicable to agricultural catchments, made up of six key components that combine to form the coupled system dynamics: namely, catchment hydrology, population, economics, environment, socioeconomic sensitivity and collective response. The conceptual framework posits two novel constructs: (i) a composite socioeconomic driving variable, termed the Community Sensitivity state variable, which seeks to capture the perceived level of threat to a community's quality of life, and acts as a key link tying together one of the fundamental feedback loops of the coupled system, and (ii) a Behavioural Response variable as the observable feedback mechanism, which reflects land and water management decisions relevant to the hydrological context. The framework makes a further contribution through the introduction of three macro-scale parameters that enable it to normalise for differences in climate, socioeconomic and political gradients across study sites. In this way, the framework provides for both macro-scale contextual parameters, which allow for comparative studies to be undertaken, and catchment-specific conditions, by way of tailored "closure relationships", in order to ensure that site-specific and application-specific contexts of socio-hydrologic problems can be accommodated. To demonstrate how such a framework would be applied, two socio-hydrological case studies, taken from the Australian experience, are presented and the parameterisation approach that would be taken in each case is discussed. Preliminary findings in the case studies lend support to the conceptual theories outlined in the framework. It is envisioned that the application of this framework across study sites and gradients will aid in developing our understanding of the fundamental interactions and feedbacks in such complex human–hydrology systems, and allow hydrologists to improve social–ecological systems modelling through better representation of human feedbacks on hydrological processes.

1 Introduction

> The history of mankind can be written in terms of human interactions and interrelations with water. (Biswas, 1970)

The vital importance of water as a resource for human well-being has been recognised since ancient times in civilisations such as Egypt, India and China. In modern times, many are now familiar with the adage that "water will be the oil of the 21st century" (Annin, 2006). However, as Gleick (1993) highlighted, this phrase omits the critical point

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that water, unlike oil, has no viable substitutes for humanity. As a result of growing populations, rapid and extensive industrialisation, and over-allocation and mismanagement of freshwater resources, a looming global water crisis that is said to be "unprecedented in human history" has been predicted (Falkenmark, 1997; Biswas, 1999; Postel, 2003; Pearce, 2007; Barlow, 2007; Biswas and Tortajada, 2011; Fishman, 2011).

It is widely recognised in the field of hydrology that human actions have myriad impacts on hydrological dynamics at the catchment system scale, including via land use changes, the alteration of flow regimes through the construction of dams and weirs, the deterioration of water quality through the pollution of waterways, as well as numerous impacts on biogeochemical cycles and riverine and lake ecology (Carpenter et al., 2011; Montanari et al., 2013). Similarly, it is acknowledged in the social sciences that the well-being of human societies is extraordinarily dependent upon what has been termed the "planet's life-support system", not only in terms of global water needs, but also with respect to its role in food production, poverty alleviation, energy production, human health, transport, climate regulation and ecosystem services (Falkenmark, 2001, 2003). Falkenmark (2003, p. 2038) makes the point that "to support the growing world population, balancing will be needed between emerging societal needs and long-term protection of the life-support system upon which social and economic development ultimately depends". This sentiment is echoed in numerous other studies (Biswas, 1997; Folke, 1998; Rockström et al., 2007, 2009; Varis, 2008). To date, major advances in the disciplines of hydrological sciences and water resources management have helped us understand these challenges, yet it remains critical that we better characterise and quantify the dynamic nature of human–hydrology interactions, in order that we can effectively manage them in a sustainable manner (Montanari et al., 2013; Thompson et al., 2013).

Notwithstanding that the dynamic interconnection of human and natural systems has long been documented (e.g. Marsh, 1864; Thomas Jr., 1956; Falkenmark, 1979; Turner et al., 1990; McDonnell and Pickett, 1993; Kates and Clark, 1999), a practical understanding of the complex co-evolution processes and interactions therein is still limited (Low et al., 1999; Kinzig, 2001; Liu et al., 2007a). Integrated Water Resources Management (IWRM) has historically been the framework within which interactions between human development and water resources have been explored. The limitation of such an approach is that the examination of single system components in isolation, such as treating scenario-based water management solutions as boundary conditions to hydrological models, is insufficient to capture the more informative co-evolving coupled dynamics and interactions over long periods (Liu et al., 2008; Sivapalan et al., 2012). As a result of this knowledge gap, interdisciplinary research efforts have emerged, such as the Coupled Human and Nature Systems (CHANS) (Liu et al., 2007a, b) and Social-Ecological Systems (SES) communities (Berkes and Folke, 1998). The focus of these efforts is on furthering our understanding of the complex interactions within the continually evolving coupled system, in terms of the feedbacks, nonlinearities, thresholds, transformations and time lags. As observed by Schlüter et al. (2012, p. 221), "while the importance of the human dimension and social dynamics for sustainable resource management is well recognised, the uncertainty generated by human responses to institutional or environmental change has only received limited attention so far". The need for a prescriptive conceptual framework which seeks to examine complex dynamics resulting from interactions and feedbacks between agents, resources and institutions on multiple levels has been highlighted (Berkes and Folke, 1998; Anderies et al., 2006b), with the caveat that "any theory devised to understand SESs… would span cognitive science, psychology, economics, ecology, biogeochemistry, mathematics, physics, etc." (Anderies et al., 2006b, p. 1). In spite of the seemingly Herculean task at hand, several recent important strides have been made to this end (Schlüter and Pahl-Wostl, 2007; Ostrom, 2009; Epstein et al., 2013; Lade et al., 2013; Schlüter et al., 2013). An excellent review of SES model applications using a host of different approaches within various fields of research, including fisheries, rangelands, wildlife, ecological economics and resilience and complex systems theory, can be found in Schlüter et al. (2012).

Out of these initiatives, examples relevant to water resource management have been presented (Schlüter and Pahl-Wostl, 2007; Schlüter et al., 2009). However, it is being increasingly acknowledged that an integrated "socio-hydrology" or "hydro-sociology" approach is required to engage hydrologists to more proactively bridge the gap that presently exists in the interdisciplinary divide (Falkenmark, 1997, 1999; Sivapalan et al., 2012, 2014; Montanari et al., 2013; Carey et al., 2014). Socio-hydrology effectively tackles the holistic integration of the socioeconomic and environmental facets of hydrology, focusing on the exploration of fundamental scientific principles of interactions, feedbacks and co-evolution of human behaviour with the hydrological system. It is important to note that, despite the many similarities, unique challenges are faced in investigations of coupled socio-hydrological systems relative to other coupled SES studies. Specific issues pertinent to catchment systems include the potential for large-scale hydrologic infrastructure development (dams and river regulation) and links between water availability and water quality. Furthermore, the resolution of the slow processes that characterise the hydrological system, non-stationarity in climate, and long timescales required to monitor threshold shifts are all distinct features of such investigations. Finally, the specific vulnerability and responsiveness that the hydrological coupled system displays in regard to climate change (Ribeiro Neto et al., 2014) presents an additional challenge. Despite such challenges, recent innovative socio-hydrology studies have proposed conceptualised models focusing on human–flood interactions (Di Baldassarre et al., 2013a, b), urban water security (Srinivasan, 2013), and downstream use of glacier runoff (Carey et al., 2014), while others have focused on tailored case-specific coupled model formulations (Liu et al., 2014; Pande et al., 2013; van Emmerik et al., 2014).

The development of a robust internationally applicable theoretical framework that has the capacity to guide the formulation of localised socio-hydrology models is needed for application across diverse study sites and application contexts. In doing so, such a framework can draw on emerging themes in the social sciences and SES literature to augment current directions in hydrology research. The resultant framework would enable extensive empirical examination of co-evolving dynamics across climate, socioeconomic and political gradients, with the ultimate aim of identifying underlying fundamental principles inherent in the integrated system.

Given the challenging nature of the exercise, in order to begin to detect certain key feedbacks and drivers in a highly complex coupled system, as a starting point this paper outlines a model framework within the context of catchments that are simplified "uni-dimensional" systems in terms of economic activity and development. In light of the fact that agriculture now covers almost 40 % of the world's terrestrial surface and accounts for approximately 85 % of global consumptive freshwater use (Foley et al., 2005; Carpenter et al., 2011), it is especially pertinent to examine agriculturally focused catchments given their global footprint. As a result of changes in land use, land cover and irrigation, agriculture has significantly transformed the global hydrological and ecological cycles (Gordon et al., 2010), with some studies documenting co-evolutionary dynamics (e.g. Anderies et al., 2006a; Kandasamy et al., 2014), thus making it an ideal focus for the study of socio-hydrology.

This paper therefore outlines a conceptual framework to examine the coupled dynamics of integrated agricultural socio-hydrology catchment systems. The paper proposes a composite socioeconomic driving variable that acts as the missing link tying together one of the key feedback loops of the socio-hydrology system. It goes on to specify six key functional components of the generic framework, showing the flexibility inherent therein to account for both the macro-scale context, as well as unique catchment-specific aspects, which can be captured through locally tailored "closure relationships". The paper concludes by demonstrating how such a framework would be applied to two site-specific Australian case studies, with a discussion on the parameterisation approach and characterisation of closure relationships for each.

2 Conceptual basis for a model of socio-hydrology

The conceptual framework put forward in this paper is a necessary simplification of an extremely complex coupled system. The intention however, is to build an approach able to support a grassroots understanding of how the coupled system might function, and to stress-test certain basic assumptions prior to progressing to more advanced and fully parameterised models. We can thus begin to comprehend the crucial components, flows, nonlinear interactions, feedbacks and responses of key system attributes that are essential steps in the development of models for interdisciplinary and complex problems (Heemskerk et al., 2003; Schlüter et al., 2012).

It is well established in the resilience literature that change (whether drastic or incremental) acts as a catalyst to response (e.g. Forbes et al., 2004; Dale et al., 2010). The question is, what magnitude of change in what composite of factors is sufficient to drive a measurable reaction in the first instance? Furthermore, once a response is invoked, what are the determinants of the immediacy and degree of that response, and what, if any, are the lagged responses? Marginal changes in the social, economic and environmental components of the socio-hydrological system may be driven by exogenous factors external to the catchment (e.g. climate, market prices and demand, political changes) or endogenous factors generated by internal feedbacks within the catchment (as stipulated in the assumptions and component equations of the model framework). Such changes invariably feed back to the hydrological sub-system via a behavioural response from the human sub-system, since humans will change the rate at which they interact with the catchment water balance. In this way, the two sub-systems are perpetually co-evolving through time, and this forms the basic premise of the proposed framework. The fundamental question we are motivated to answer through application of a socio-hydrological model is what *drives* the human response within the human sub-system. As outlined above, the impacts of land and water management decisions on the hydrological system, in terms of water balance, flows and quality, are presently well understood and modelled. However, the *drivers* of the human feedback component at a system scale have remained elusive. The goal of a socio-hydrology model is therefore to identify, conceptualise and eventually quantify these drivers, so as to formulate generalised principles that will form the basis of a broadly applicable coupled model.

We are effectively employing a systemic approach in our analysis of this particular brand of social-ecological system. Resilience theory at its essence is indeed rooted in systems thinking as it focuses on capturing and categorising the dynamics of change (Gunderson and Holling, 2002). We know from general systems theory that complex systems, such as that described here, display highly nonlinear tendencies with attractors to certain stable states or repellors from unstable states, and thresholds and rapid responses between state transitions may therefore emerge (Scheffer, 2009; Lade et al., 2013). In formulating policy, understanding these system-scale behaviours and the emergence of such dynamics can offer guidance as to what the sustainable limits of a catchment system are (Schlüter and Herrfahrdt-Pähle, 2011), and to what extent complex trajectories (e.g. hysteresis) may exist between catchment states. For instance, a 3-D matrix of

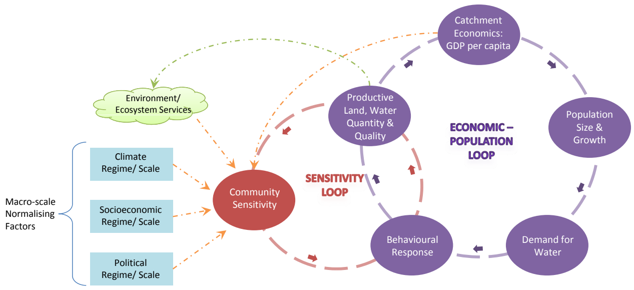

Figure 1. The socio-hydrology model as two interconnecting feedback loops.

catchment states defined by water availability (S_{T}), degraded land (A_{D}) and catchment GDP (E_{C}) could potentially show a catchment moving from state 1 point of "high S_{T}–low A_{D}–low E_{C}" to state 2 of "low S_{T}–high A_{D}–high E_{C}". Negative feedbacks will stabilise the system such that it will resist being pushed away from its original position. However, positive feedbacks and unstable dynamics will induce a shift to the second position. The three key emergent properties we seek to understand in the interaction of the socio-hydrological system from a systems perspective are (1) the resilience of the system in terms of its stability and threshold behaviour from one state to another, and in so doing to gauge the nature and magnitude of negative and positive feedbacks, respectively, (2) the differences in timescales and inherent lags between system interactions and feedbacks, and (3) the degree of adaptation and learning intrinsic to the human system (Allison and Hobbs, 2004; Gunderson and Holling, 2002). These behaviours are what we are aiming to investigate with the socio-hydrology model in order to better understand the workings of the coupled system. Understanding these key system features, as well as the system variables that characterise its dynamics, would enable targeted policies and management strategies that promote sustainable water resource management, especially with respect to case studies at earlier stages of the evolving cycle.

2.1 The two key feedback loops

In this section we highlight two principal feedback loops that emerge in the dynamics of the coupled system (Fig. 1). The first is referred to as the "Economic-Population Loop" and the second as the "Sensitivity Loop". With respect to the former, the increasing trend in global water use has been closely linked to both population growth and economic development over the past few centuries (Vörösmarty et al., 2005). If we take a pristine catchment (pre-human influence) we would observe certain hydrological variables as a result of its climate and geophysical make-up. These effectively determine the initial condition for available water quantity and quality. A certain proportion of this available water would be employed towards economic gain (for example, for normal household use and agriculture). This economic gain would be distributed (often unequally) on a per capita basis throughout the catchment community. It follows that, as the per capita economic gain increases, the catchment presents a more attractive lifestyle proposition causing a net migration of people into the catchment, such that population size would increase, as well as its rate of growth, similar to Myrdal's (1957) concept of "circular and cumulative causation". A growing population would be accompanied by higher levels of demand for water and land, by virtue of increased household consumption and a growing requirement for economic development to sustain the larger community (Molle, 2003). In addition, as a rural catchment with a predominantly agricultural micro-economy increases in prosperity, water demand will originate from additional sources independent of population growth, to a point, e.g. from the manufacturing sector, thermoelectric sector and increasingly sophisticated domestic household needs (as observed by Flörke et al., 2013).

This heightened demand is likely to be one of the key drivers feeding into water management decisions, such as extraction rates, land clearance rates and the construction of storage facilities. Management decisions would be reflected in the community's economic prosperity in the short term, and filter through to water quantity and quality variables over a longer timescale. From this point, the water variables can be viewed more as limiting variables or lower boundary conditions, whereby economic growth will continue to be possible until such time that the quantity or quality of natural resources impede further growth. Water use efficiency measures would feed into the cycle to extend the life or economic productivity of these limiting variables. However, to the extent that water flows reduce, water quality deteriorates or land degrades, economic growth will naturally be constrained. In the case of common pool resources, the resource that underpins development, in this case the freshwater resource, is often prone to over-exploitation, which can ultimately lead to a deterioration in local social and economic conditions (Hardin, 1968). This will in turn encourage migration out of the catchment as people go in search of other work and income opportunities, which will in turn reduce the demand for water and land. Management decisions might then reasonably respond by reducing extraction rates and environmental restoration. This is the first feedback loop that merits investigation.

The second loop is modelled on a three-pronged exposure–sensitivity–response paradigm, and introduces a Community Sensitivity state variable. There is support for such a fusion of vulnerability and resilience approaches when examining complex coupled human–environment systems in sustainability science (Turner, 2010). Turner et al. (2003) propose a generalised framework for assessing the sustainability of coupled systems that employs aspects of exposure, sensitivity and resilience. Their framework is broadly consistent with the conceptual framework proposed in this paper, in that the *exposure* to change (whether drastic or gradual) in the socio-hydrological system is captured by the primary sub-system functions originating from a change in the land and water variables; the *sensitivity* to that exposure is what we seek to capture in our sensitivity variable; while the demonstrated *resilience* of the system is effectively reflected within a behavioural response function that drives actual change within the catchment (further discussed in Sect. 3.6).

The underlying premise of the Sensitivity Loop is that behaviour and water management decisions are directly driven by a community's social and environmental values, local action, lobbies and the like, all of which reflect community sensitivity to direct and indirect impacts of a marginal change in one or more of the water variables. The behavioural response, as before, will impact future available water quantity and quality. The proposition in this paper is that as the Sensitivity state variable displays an upward or downward shift, there will be a corresponding observable shift in a Behavioural Response function. It is hypothesised that as Sensitivity increases, behaviour and management decisions will tend towards reducing the community's impact on the basin's hydrological signature (i.e. a move towards a more natural environment). Conversely, lower sensitivity rates will be associated with more aggressive behavioural responses that tend towards manipulating available water resources to the community's needs (i.e. a more observable anthropogenic footprint).

The assumption of rational behaviour in this context pertains to the likelihood that overarching community behaviour will tend towards the longer-term collective good, rather than the short-term individual good. One of the challenges associated with the management of water resources is that it is a common pool, open access resource, and as a consequence it is potentially prone to overharvesting as individuals seek to optimise use, otherwise known as the "tragedy of the commons" (Hardin, 1968). In recent decades however, the prediction of collective over-exploitation of the resource under the rational-agents paradigm has been called into question (Ostrom et al., 2002). It has become increasingly apparent that such individual optimisation is not always the case, and that in fact the degree of collective co-operation in commons dilemmas is influenced by both micro-situational variables (e.g. heterogeneity among agents, group size, communication, reputation, time horizons) and the broader context (Anderies and Janssen, 2011; Tavoni et al., 2012; Anderies et al., 2013). This is in line with Giddens' (1984) early work on structuration theory, which posits that social phenomena are the result of both agency and social structure. Indeed, Kinzig et al. (2013) note that as adopters of a particular behaviour reach a critical quorum, which may be as few as 10 % of the population, a tipping point may be reached that causes the new norms to be more widely adopted by the community, such that a collective move towards more environmentally sustainable practices occurs. Thus, a composite variable based on collective community sensitivity as a driver to co-

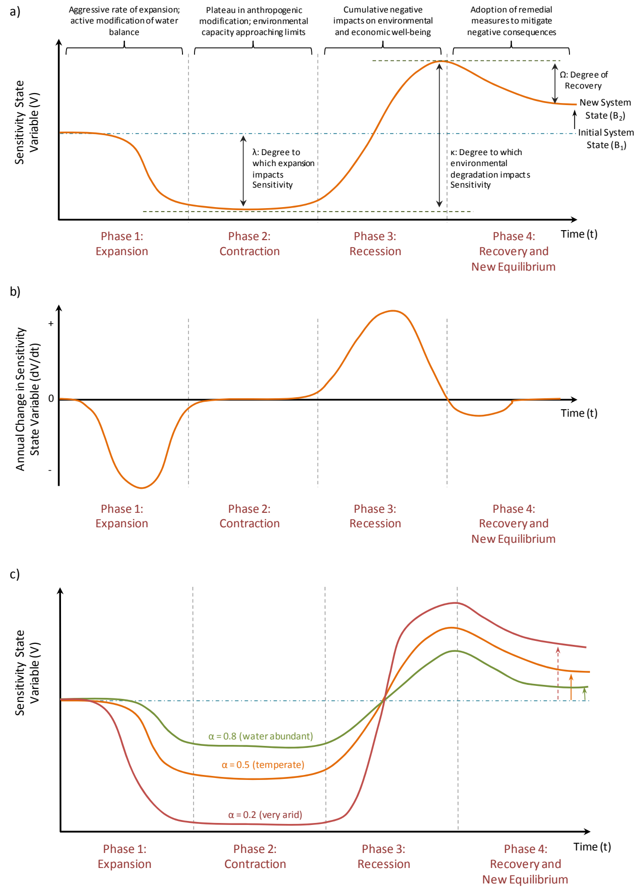

Figure 2. An idealised sketch showing a hypothetical trajectory over time for the **(a)** Sensitivity state variable, and **(b)** the change in Sensitivity $\left(\frac{\mathrm{d}V}{\mathrm{d}t}\right)$ in the case of an example catchment. A comparison of the idealised trajectory across a regional climate gradient is shown in **(c)**.

operative action is achievable with the use of agent-based models (Tavoni et al., 2012) that would account for the diversity of actions by individual actors within the catchment community.

Figure 2a, b illustrate hypothetically how the Sensitivity state variable is expected to vary over time in an idealised case. In the first phase (Expansion) there is a strong drive for anthropocentric behaviour, with active modification of the water balance via land clearing, water extractions and the construction of storage infrastructure. A sharp decline in community sensitivity levels would be observable in this phase driven by increasing economic prosperity and an aggressive rate of environmental modification to suit development needs. The second phase (Contraction) is characterised by a cessation in environmental modification. Economic prosperity could still be increasing during this phase (e.g. driven by increased efficiency and mechanisation) however it is generally offset by the appearance of negative environmental consequences. The first two phases can be thought of as the positive feedback loop indicative of the Economic-Population Loop described in Fig. 1. The third phase (Recession) is comprised of a sharp cumulative decline in the condition of environmental resources, accompanied by a downturn in economic prosperity linked to the resource degradation. The negative feedbacks inherent in the Sensitivity Loop take effect during this phase. The fourth phase (Recovery and New Equilibrium) is characterised by a shift in behaviour patterns towards enviro-centric management and policy focused on alleviating the negative consequences of a legacy of expansion. The system shifts to a new equilibrium state (B_2), that may then be subject to a relatively small magnitude of oscillation beyond this point. There are numerous examples in the literature where this pattern of a positive feedback loop followed by a negative feedback loop (i.e. aggressive development followed by remedial management efforts) has been observed in local and regional-scale socio-hydrological systems: the Saskatchewan River basin in Canada (Gober and Wheater, 2014), the Tarim River basin in China (Liu et al., 2014), the Murrumbidgee River basin in eastern Australia (Kandasamy et al., 2014), the West Australian "Wheatbelt" (Allison and Hobbs, 2004), the California Delta in the US (Norgaard et al., 2009) and several other basins around the world (Molden et al., 2001; Molle, 2003; Vörösmarty et al., 2005; Kinzig et al., 2006; Savenije et al., 2014). Indeed, these dynamics have also been observed at a larger global system scale (Cosgrove and Rijsberman, 2000). We acknowledge that Fig. 2 is hypothetical and real-world cases will exhibit departures from the idealised trajectory depicted.

2.2 Identifying the missing link: community sensitivity as a state variable

A clear starting point in the development of a systems model spanning water resources and human activity requires the definition of a set of state variables and the core "currencies" of the model. In general terms, these relate to: (a) water availability and environmental quality, (b) economic value of the catchment system, and (c) social and population dynamics and structure. However, the challenge in modelling both socioeconomic and hydrological systems is that it is difficult to define what connects this collection of catchment system variables.

In the framework, we propose a composite driving state variable that can be thought of as the community's sensitivity to a change in hydrological variables, as it begins to manifest in associated economic and environmental variables. In the simplest sense, the greater the collective sensitivity, the greater will be the stimulus to take enviro-centric action (Falkenmark, 1997; Folke et al., 2010) and this creates a negative feedback that will promote stability. Likewise, the lower the sensitivity, the less likelihood that a change in hydrological variables will lead to meaningful enviro-centric action, and the population will continue to drive the system towards a different state-space location that may be more or less sustainable. The drivers of collective human values, emotions, perceptions and behaviour, already forms a body of research within the psychology and natural resource management fields, with myriad theories and ongoing debate (Ajzen, 1985; Broderick, 2007; Stein et al., 1999; Vanclay, 1999, 2004; Vaske and Donnelly, 1999; Armitage and Christian, 2003; Seymour et al., 2010; Mankad, 2012). This paper does not aim to contribute to these debates. Rather, from a purely socio-hydrological context, we are seeking to simplify these drivers into observable proxies that enable an understanding of how the coupled system interacts. We define these proxies as socio-hydrological "closure relationships", which refer to the formalisation of certain contextually-specific relationships with mathematical functions in order to fully resolve interdependencies required to make equations determinate.

This paper puts forward the suggestion that a community's sensitivity stems from its perceived level of threat to its quality of life, which could also be thought of in terms of a disruption to its established norms and behaviours (Kinzig et al., 2013). The more a community perceives its quality of life to be under threat, the more likely it is to display heightened sensitivity to a marginal change in factors that could subsequently negatively impact its quality of life. Conversely, the less a community perceives its quality of life to be under threat, the less likely it is to be sensitive (and hence react) to marginal changes in such variables. In this way, the sensitivity is related to how any marginal change in hydrological variables manifests itself in the economic, social and environmental dimensions that more directly pertain to a community's overall quality of life. Indeed, there is evidence to support the notion that the behaviour of a watershed community, with respect to water management, is dependent upon its held *perceptions* of the severity and magnitude of problems it faces (Molle, 1991, 2003; Turral, 1998; Zilberman et al., 2011). Although most of this literature addresses response management to severe water shortages or disasters, these are still extreme manifestations of the inherent causal link between perceptions of threat and action.

There is support in the psychology literature for the use of a "perceived threat" variable as a precursor to action. According to protection motivation theory (Rogers, 1975) the notion of a threat can be broken down into three components: (i) threat vulnerability, or the likelihood that the threat will affect the individual directly, (ii) threat severity, or the degree of personal impact that would result to the individual, and (iii) response efficacy, or the belief as to one's ability to cope with the threat (which could also be couched in terms of perceived resilience). In so far as this theory has been applied to the environmental sciences, Mankad and Tapsuwan (2011) found that perceptions of threat vulnerability and severity in relation to future water shortages were significantly related to adaptation and mitigation behaviour. Similarly, Baldassare and Katz (1992) found that personal threat perception was a more robust predictor of pro-environmental behaviours relative to demographic variables. Furthermore, there is ample evidence in the literature to support the view that people's perceptions and propensity to act are directly related to their degree of physical proximity and personal experience with the issues faced. Put another way, people tend to be most sensitive to those things that impact *directly* upon their quality of life (Kollmuss and Agyeman, 2002; Rolfe et al., 2005; Broderick, 2007; Gooch and Rigano, 2010).

Resilience, in its traditional sense, hinges upon the notion of positive, adaptive responses that may be preventative or responsive in nature, in order to avoid or moderate negative consequences (Masten et al., 1990; Luthar et al., 2000). Although the concept of resilience originated in the ecological sciences (Holling, 1973) it has been found to be particularly useful in the examination of coupled human–nature system studies (Berkes and Folke, 1998; Berkes and Jolly, 2002; Berkes et al., 2003; Falkenmark, 2003; Folke, 2003; Anderies et al., 2004; Folke, 2006; Forbes et al., 2009; Amundsen, 2012). Whether used in the field of psychology, ecology or social science, the concept is based upon the premise of a system's response to *change*. Negative consequences in our model are analysed with respect to the catchment community's quality of life. Given that sensitivity, as applied in this paper, is essentially a subjective variable, it could prove ultimately impossible to quantify in absolute terms in any widely applicable way. This paper therefore posits the use of a *relative scale*. In this way, the scale would reflect a marginal change, as opposed to reporting an absolute value, thus shifting the focus to the direction and relative magnitude of any movement.

From a model standpoint, the overall objective is to develop a lifestyle sensitivity variable that is capable of adequately capturing a community's shifting perception of its own vulnerability, such that it is a reasonable precursor to observable action. Community perceptions have generally been canvassed using qualitative means (i.e. interviews, surveys) as it is an inherently subjective trait (Broderick, 2007; Guimarães et al., 2012; Tolun et al., 2012). Given that it is only possible to canvass perceptions in this manner at a given point in time, we are precluded from doing this in the present context as we are attempting to capture phenomena and feedbacks over historical periods of a century or more. At present, there is no prescriptive method for quantifying or modelling human perceptions to changes in their environment (environmental, social, economic or otherwise) (Jones et al., 2011; Lynam and Brown, 2012) and we therefore resort to proxies (discussed in more detail in Sect. 3.5). However, it is conceivable that at some point in the future, advancements in mental models research will enable the substitution of a more sophisticated parameterisation of our sensitivity variable.

3 The six key components of a generic framework

The conceptual foundations outlined above are used to underpin the construction of a prescriptive socio-hydrology

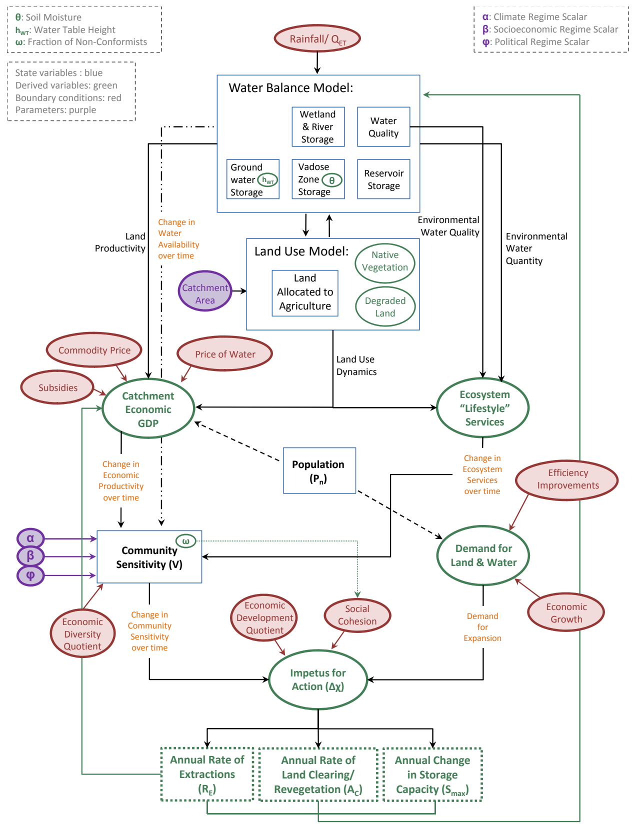

Figure 3. A generic socio-hydrology conceptual framework for application to agricultural catchments.

framework for application to agricultural catchments. The framework in a generic form consists of six components that together combine to form a coupled system capturing the feedbacks previously highlighted (Fig. 3). The following section describes each of the main framework components with discussion of associated functional relationships that are required to be parameterised (the reader is referred to Appendix A for a complete list of variables and associated measurement units). The first four components can be modelled in numerous ways, with the level of complexity inherent in the chosen method up to individual practitioners to determine, depending on the relative importance of each aspect to the investigation at hand. However, to demonstrate how the framework would be applied, we have sketched some generic basic concepts that could be applied to realise each component.

3.1 Catchment hydrology

A suitable water balance model is required to conduct the coupled simulations and this may take the form of a simple conceptual water balance model (e.g. Farmer et al., 2003), or a more complex hydrological model. At a minimum, the model must accommodate an array of input variables based

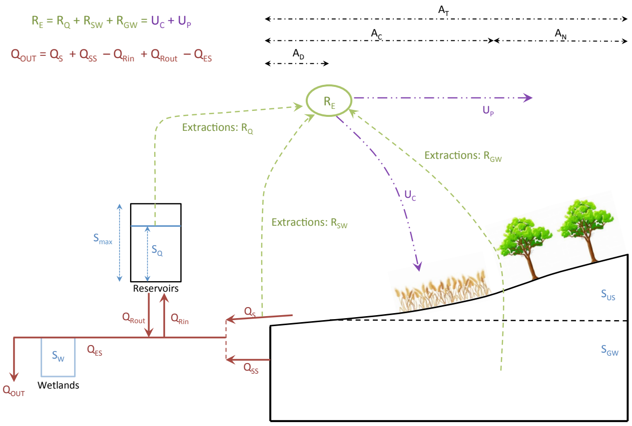

Figure 4. A simple catchment water balance model that includes the minimum necessary components for application of the conceptual framework. Groundwater (S_{GW}), vadose zone (S_{US}), wetland (S_W) and reservoir (S_Q) water stores are included. Cumulative man-made storage capacity (S_{max}) must also be included. Net flows (Q_{OUT}) are modelled using surface flow (Q_S), subsurface flows (Q_{SS}), managed diverted (Q_{Rin}) and released (Q_{Rout}) flows, and environmental flows (Q_{ES}). Managed extractions made from the system from each of groundwater (R_{GW}), surface (R_{SW}) and reservoir (R_Q) sources must be accounted for. Land use decisions must also be reflected, such that area of land cleared (A_C), area of remaining land comprised of natural vegetation (A_N) and area of degraded land (A_D) must form a dynamic part of the model.

on the basic geophysical properties of the catchment, climate forcing, and also allow for anthropogenic influences on the hydrological signature of the basin. For most cases a model setup where the catchment is divided into sub-catchments (i.e. semi-distributed) with each accounting for dynamics of soil moisture, groundwater stores, evapotranspiration and surface water runoff and routing or storage (as relevant) would be suitable. Where the underlying socio-hydrologic case study requires resolution of changes to water quality, then this model must be extended to simulate water quality dynamics.

For the purposes of this paper, the specification of the water balance model is only covered in general terms, as depicted in Fig. 4, and individual case study applications of the framework would require contextually relevant implementations. The key attributes the model must have however, to support simulation of the coupled dynamics, is to allow for a link to water-related management decisions relevant at the catchment system scale. These include ability to accommodate within the catchment water balance: (i) changes in land cover (A_C; e.g. due to clearing of native vegetation), (ii) changes in the rate of extractions of either surface water or groundwater for economic activity (R_E), and (iii) changes in the capacity for water storage (S_{max}; e.g. development or removal of reservoirs or other forms of river regulation). Based on behavioural responses outlined below, these three mechanisms form the core links that allow the water balance to be modified by the catchment population.

3.2 Population dynamics

The Demographic Transition Model has been used extensively to model the relationship between development and population in human geography (Jones, 2012), and may be employed to calculate the catchment population. This approach bases population dynamics on changes in the birth rate and death rate as a country moves through five different stages of development. Extensions and variations of the core model have been developed for various countries, which allow the potential for more tailored versions of the model to be applied. In addition to the birth rate and mortality rate, the net permanent migration rate can be calculated by accommodating various "push" and "pull" factors that focus on local economic, environmental and political conditions (Fouberg et al., 2010). In general terms, the population state variable, P_n, would evolve according to

$$\frac{\mathrm{d}P_n}{\mathrm{d}t} = (b - m + \mu)P_n, \tag{1}$$

where b is the annual birth rate, m is the annual mortality rate, and μ is the annual net migration rate. Migration is driven by a wide range of local and external factors and is beyond the scope of this paper to cover in detail, however depending on the application context, it could be driven by internally derived variables related to the catchment system, for example, economic benefit of crop production or ecosystem services and conditions that support a high quality of life. Additional factors such as natural (e.g. earthquakes, drought) and man-made (e.g. war) hazards could act as "push" factors.

Various applications of this socio-hydrology framework may elect to parameterise this variable differently – for example, by employing a locally developed population model, or indeed by holding population as an externally provided boundary condition if the rate of change of population is not a core part of the relevant investigation, as the case may be.

3.3 Economic function

Within the model framework the economics of the catchment, captured in its simplest form, can be made up of a benefit component (i.e. land productivity) and a cost component (i.e. broken down into agricultural cost and water supply cost). The first component relates to the economic benefits resulting from agricultural activities and can be calculated using an income per m^2 metric based on global commodity prices, considering the dominant local agricultural enterprise undertaken (e.g. rice, wheat, beef, dairy, etc.). Where relevant, this can be tempered by a land degradation scaling factor, which reflects the extent of salinisation or other form of landscape dysfunction, thus reducing the effective area available for economic expansion. This component will be partially driven by land-use management decisions (A_C). With respect to the second component, this can be represented by a cost per m^2 metric intended to capture direct farming costs (i.e. labour, machinery, fertilisers, etc.) and a cost per m^3 metric intended to capture the cost of water to sustain the catchment population (i.e. for irrigation and other household and industrial use). The latter water supply component is thus driven by management decisions regarding the amount of water that is available for supply and allocation ($S_{\max}$ and R_{E}), and would take into account any subsidies offered on the price of water, as well as supply-driven changes in the price of water to the extent that water resources become scarce. It should be noted that water usage in this instance should already reflect any potentially negative impacts of deteriorating water quality below drinking/irrigation grade, and may have a graded-scale of cost to account for local complexities.

These components together provide the direct net basic economic benefit. However, it is widely accepted within the environmental economics literature that agriculture multiplier effects exist, as basic earnings are disseminated further into non-agricultural sectors of the local and national economy (Johnston and Mellor, 1961; Byerlee et al., 2005; Bezemer and Headey, 2008). This may be captured by a multiplier, τ_{A}, that can be incorporated for a more realistic indication of the community's prosperity derived from agricultural productivity growth. Although a complex calculation of τ_{A} is beyond the scope and intent of this paper, a simplistic calculation tied to the annual national household savings rate could be used, or alternatively, τ_{A} could be set to 1 in the simplest case. Thus an economic function of the form

$$E_{\mathrm{c}} = \left[p_{\mathrm{c}} A_{\mathrm{C}} \left(1 - A_{\mathrm{D}}\right) B_{\mathrm{c}}\right] \tau_{\mathrm{A}} - \left[\left(c_{\mathrm{A}} A_{\mathrm{C}} \left(1 - A_{\mathrm{D}}\right)\right) + \left(p_{\mathrm{wc}} U_{\mathrm{c}} + p_{\mathrm{wp}} U_{\mathrm{p}}\right)\right] \pm E_{\mathrm{ext}} \tag{2a}$$

$$E_{\mathrm{pc}} = E_{\mathrm{c}} / P_n \tag{2b}$$

can be adopted, where E_{c} is the total economic gain within the catchment economy, p_{c} is the global commodity price of the predominant agricultural crop or activity, A_{C} is the cleared land allocated to agriculture, A_{D} is the fraction of degraded cleared land within the catchment unsuitable for agricultural production, B_{c} represents the crop or pasture biomass, τ_{A} is the economic multiplier of agriculture, c_{A} is the non-water related cost of undertaking the relevant agricultural crop or enterprise, p_{wc} is the price of irrigation water, p_{wp} is the price of water supplied for household use, and U_{c} and U_{p} are the total quantity of water supplied for irrigation and household and other use, respectively, within the catchment. In a dryland farming context, the available biomass from within A_{C} will depend upon the recent climatic conditions and will respond to periodic shifts in average soil moisture, θ, for example. The land productivity component is thus directly driven by the outputs of the hydrology model (e.g. crop or pasture productivity will increase during suitable soil moisture conditions and irrigation water supply and will be limited by expansion of degraded land area), and management decisions from the Behavioural Response model, described below, will alter the rate at which A_{C} increases or decreases. To the extent agricultural subsidies are in place, given the diverse forms such subsidies may take, we leave it to individual practitioners to determine the most appropriate catchment-specific approach (e.g. via a reduction in the agricultural cost component) depending on the nature of the subsidy in question.

It is important to note that such metrics are felt to sufficiently capture the economics of a predominantly agricultural catchment. E_{ext} is included as an optional variable to account for income generated within the catchment from industry sources independent of agriculture, and could be set to zero in the simplest case. To the extent that the catchment in question has additional industries, such as a strong fishing industry, manufacturing industry or hydropower plants, the income generated from such industries could be captured in one of two ways. The first is through a dynamic model or equation similar to Eq. (2a) tailored to the industry in question. Alternatively, such income could more simply be treated as a boundary condition and incorporated via E_{ext} (i.e. dollar per annum metric derived from the relevant industry). We leave it to individual practitioners to determine which approach is more appropriate depending on the nature of the investigation being undertaken, and we highlight the opportunity this presents for the model framework to couple with more complex economic models. To the extent that a more detailed catchment-specific economic model is available, there is scope to integrate such a model with the more generalised function outlined above.

3.4 Ecosystem services function

In addition to the economic growth driving activity within the catchment, the benefit derived from lifestyle-related ecosystem services (L_{ES}) must be considered. Given that the accurate valuation of ecosystem services continues to be an extremely complex undertaking (Bengston, 2008), the framework proposes to account for L_{ES} via an incremental scale that demonstrates the relative magnitude and direction of an improvement or degeneration. This circumvents the need to directly measure ecosystem services, by providing a lumped indicator that could be customised for specific applications. For the sake of argument, a number of general proxies could be used, such as changes in measured water quality parameters, and surveys measuring the abundance and number of species of fish, vegetation and birds. It could also incorporate the percentage of natural vegetation (denoted as A_{N}). Changes in each of these factors may be measured on an absolute basis, equating to a net positive or negative percentage change in overall L_{ES} that is then used to effectively reduce or increase the Sensitivity variable described next.

If we consider an example of an agricultural catchment impacted by water diversions and experiencing problems associated with water shortage, wetland degradation and eutrophication, then a simple L_{ES} function may be envisioned that is able to link predictable functional relationships amongst certain core primary hydrology and land use variables to the extent of consequential damage to ecosystem services, for example

$$L_{\mathrm{ES}} = f\left(W_{\mathrm{Q}}\right) + f\left(Q_{\mathrm{ES}}\right) + f\left(S_{\mathrm{W}}\right) + f\left(A_{\mathrm{N}}\right), \tag{3}$$

where W_{Q} is a measure of the relevant water quality variables (total suspended solids, total nutrients, cyanobacteria, pathogens etc.), Q_{ES} relates to environmental or residual flow in riverine environments, S_{W} represents river and wetland water storage to the extent that important wetlands exist within the catchment, and A_{N} is the fraction of the landscape covered by natural deep-rooted vegetation. As it stands, the above equation assumes equal weightings for each of these variables, however inclusion of different weightings may be more suitable if the weighting factors may be appropriately derived (e.g. similar to Imberger et al. (2007) where stakeholder survey techniques were employed, bearing in mind however that such a technique would only provide present-day user-defined weightings). We acknowledge that this is highly simplified but use this example to demonstrate how empirically observable trends in the condition of the catchment's land and water resources can be used to develop a proxy indicator that reflects the community's view of environmental benefits that the catchment is providing.

3.5 Sensitivity state variable

The Sensitivity function proposed here is comprised of six elements, three of which are national or regional in scale (macro-scale contextual parameters) and three of which pertain more specifically to local catchment dynamics. This approach of using local dynamic variables supplemented by the regional and national context in the examination of coupled human–nature systems, is supported in the literature (Liu et al., 2007a). The three macro-scale parameters to be applied comprise the regional climate regime in which the catchment is located (α), the national socioeconomic development context (β), and the national political regime (φ), whilst the three catchment-scale elements pertain to water abundance, and the economic and environmental well-being experienced by the catchment community.

The first macro-scale contextual parameter we introduce, α, reflects the underlying regional climate regime within which the catchment is located, with drier catchments expected to display a greater reaction in sensitivity levels, compared with catchments that have abundant water resources, as the same magnitude of change in water quantity will elicit different consequences (Cumming et al., 2005; Simane et al., 2012). Thus a "dryness" scale is adopted, with 0 corresponding to a very arid catchment, and 1 corresponding with an extremely wet catchment. Whilst several metrics may be used for this purpose, widely used indices include the Dryness Index (E_{p}/P) or the UNEP (1997) Aridity Index (P/Q_{ET}).

The second macro-scale contextual parameter, β, reflects the influence of the national socioeconomic regime on perceived catchment community sensitivity levels. As nations move along the scale from rural to transitional to industrialised, it is expected that perceived resilience levels increase. Some studies have explained evidence of this connection by virtue of the increase in income diversification as countries move along the development scale from a rural economy dependent upon a narrow resource base, to an industrialised economy dependent upon a more diversified resource base (Adger, 2000; Biswas and Tortajada, 2001; Molle, 2003; Briguglio et al., 2009; Smith et al., 2012). Others have focused on the increased social and economic capacity to respond to change that goes hand in hand with more developed and technologically advanced economies (Allan, 1996; Folke, 2003; Sherrieb et al., 2010). In this way, we seek to capture the way in which β interacts with the Sensitivity variable, V, where wealthier more developed economies are more able to proactively respond to water stress by modifying the catchment water balance, thus making such societies less sensitive to these pressures. This does not in itself imply that the society will in fact implement such changes (with rigidity and "lock-in" traps being noted examples of such failures; Scheffer and Westley, 2007), but rather that it has the ability to do so, and thus its perceived level of threat is lower. The Human Development Index (HDI) has been employed by the UNDP since 1990 to compare economic development across nations (UNDP, 1990), and it is proposed that the HDI scale be incorporated into our analysis, such that 0 represents a subsistence level rural economy, and 1 is a fully industrialised economy. For example, the inequality-adjusted HDI (Human Development Report, 2013) for a developed nation such as Australia is 0.864 (labelled "very high human development"), whilst transitional economies such as China and Vietnam score 0.543 and 0.531 respectively (medium human development), and a developing economy such as Ethiopia scores 0.269 (low human development). It is noteworthy that there is a marked observable difference between the climate regimes of developed versus developing countries, which may amplify certain effects. In 1961, a United Nations report observed that developed nations are generally located in temperate climate zones while developing nations are predominantly located in tropical and semi-tropical regions where seasonal rainfall patterns are more pronounced (Biswas, 2004).

The third macro-scale contextual parameter to be captured is the national political regime, φ, in which the catchment operates. This is used as a moderating variable to reflect how responsive the government is to community sentiment. For instance, in a democratic society where government elections are regularly held such that community sentiment must be taken into account, it is expected that the behavioural response, at the government level in particular, is relatively more responsive to community sentiment. In contrast, in an authoritarian regime, it is expected that the signal would be diminished due to corruption or self-interest within government. Molle (2003) concedes that the degree of decentralisation and democratisation of government can influence how negative impacts are perceived and addressed. However, the evidence is not definitive as to which model is best able or likely to affect change. For our purposes, the proposed political scalar is more concerned with whether the political regime in place is an *impediment* to the wishes of the community. To this end, the more democratic a regime, the less likely that there will be an active impediment between community sensitivity and response. It is also worth noting that Forbes et al. (2004) found a link between the stability of a political regime and community vulnerability – the greater the stability and stronger the regulatory framework, the lower the vulnerability of the community. Therefore, it is proposed that a scale such as the Corruption Perception Index (CPI) by Transparency International (2012), would be appropriate, though others may also emerge depending on specific contexts. By way of example, the CPI for Australia is 85 (i.e. considered "very clean"), whilst China scores 39 (considered "somewhat corrupt"), and Russia scores 28 (i.e. deemed "very corrupt"). Therefore, for lack of additional data the proxies to be used for the three macro-scale parameters, namely climate, socioeconomic and political regimes, are a dryness/aridity index, the HDI and the CPI, respectively. Together these can be set to define the catchment context and will constitute controls that serve to either amplify or dampen the feedback loops highlighted in Sect. 2.

The remaining three factors that make up the Sensitivity state variable are inherently part of the dynamic workings of the catchment community. The water quantity and quality variables influence sensitivity in two ways. Firstly, there is a direct relationship between the "available" amount of water in the catchment for consumption, $S_x = f\left(S_{\mathrm{Q}}, S_{\mathrm{GW}}, S_{\mathrm{US}}\right)$, and the perceived level of threat. It follows that as S_x decreases, the community's perceived threat to their quality of life will increase. Conversely, an increase in S_x would be expected to be associated with a decrease in sensitivity levels as water is becoming more bountiful. It is worth highlighting that, depending on the local context, this function could simply be the sum of all water sources, or a weighted sum with the most socially relevant sources given greater weighting (e.g. S_{Q} for an irrigated catchment or S_{US} for a rainfed catchment where soil moisture is pertinent to productivity), as the case may be. Note that S_x is determined by anthropogenic drivers (i.e. population size and water management decisions) as well as changes in climate parameters.

The second way in which the catchment water balance impacts a community's sensitivity is through the effect on lifestyle-related ecosystem services, L_{ES}, provided by the catchment as outlined above. There is substantial evidence that flow alterations and/or a decline in water quality negatively impact ecosystems services (Walker and Thoms, 1993; Cullen and Lake, 1995; Bunn and Arthington, 2002; Arthington and Pusey, 2003; Vörösmarty et al., 2005; Tolun et al., 2012). As ecosystem services deteriorate (whether due to decreased flora and fauna, algal blooms, worsening water quality, a decline in aesthetic or recreational value, increased water-borne diseases etc.) a community's sensitivity level is expected to rise (Odum, 1989; Daily, 1997; Vörösmarty et al., 2005; Bunch et al., 2011; Steffen et al., 2011). This is a reflection of a growing threat that has a direct and observable impact on the community's quality of life.

Finally, the catchment community's GDP per capita, E_{pc}, will influence its perceived vulnerability and resilience. It is important to note that this metric can change in spite of the overall socioeconomic regime remaining the same. For instance, a catchment may be located in Australia, which is considered a developed and industrialised first-world country. However, even though national movement along the socioeconomic development scale takes place on a multidecadal basis, community E_{pc} can rise and fall multiple times during several economic cycles in the process. The more prosperous a community, the higher its perceived resilience level and lower its perceived sensitivity level (Folke, 2003; Briguglio et al., 2009; Sherrieb et al., 2010). In an appraisal of land use case studies from around the world, Lambin et al. (2001) concluded that economic circumstances were the chief determinant of community and societal response. Thus it is hypothesised that a direct inverse relationship exists between E_{pc} and sensitivity, whereby an increase (decrease) in E_{pc} will be associated with a corresponding decrease (increase) in a community's sensitivity level. This is in response to a change in the net wealth of the community, and hence its ability to enjoy an enhanced (diminished) quality of life.

Accordingly, the change in the Sensitivity state variable, V, over a period hypothetically illustrated in Fig. 2, may be estimated as

$$\frac{\mathrm{d}V}{\mathrm{d}t} = \left[\left(\underbrace{-\widetilde{S_x}\gamma_{\mathrm{s}}}_{\substack{\text{water}\\\text{availability}}}\ \underbrace{-\widetilde{L_{\mathrm{ES}}}\gamma_{\mathrm{es}}}_{\substack{\text{ecosystem}\\\text{services}}}\ \underbrace{-\widetilde{E_{\mathrm{pc}}}\gamma_{\mathrm{e}}(1+\delta)}_{\substack{\text{economic}\\\text{return}}}\right)\right.$$
$$\left.\underbrace{f(1-\alpha)}_{\substack{\text{climate}\\\text{context}}}\cdot\ \underbrace{f(1-\beta)}_{\substack{\text{development}\\\text{context}}}\ \cdot\underbrace{f(1-\varphi)}_{\substack{\text{political}\\\text{context}}}\right]V, \tag{4}$$

where α is the climate regime scalar ($0 < \alpha < 1$), β is the socioeconomic regime scalar ($0 < \beta < 1$), φ is the political regime scalar ($0 < \varphi < 1$), $\widetilde{L_{\mathrm{ES}}} = \Delta L_{\mathrm{ES}}/\overline{L_{\mathrm{ES}}}$ is the relative change in ecosystem services of the catchment, $\widetilde{E_{\mathrm{pc}}} = \Delta E_{\mathrm{pc}}/\overline{E_{\mathrm{pc}}}$ is the relative change in economic gain per head of capita for the catchment population, $\widetilde{S_x} = \Delta S_x/\overline{S_x}$ is the relative change in water availability within the catchment, and δ is a GDP concentration metric that captures agricultural production as a percentage of GDP (i.e. an economic location quotient analysis around agricultural productivity). Each of $\widetilde{S_x}$, $\widetilde{L_{\mathrm{ES}}}$ and $\widetilde{E_{\mathrm{pc}}}$ are normalised by a mean or reference value to calculate the relative change over the interval $t-n{:}t$, where n is the number of time steps used to calculate the relative change and can be used to define a lag time between change and response. The change in any one of these local sensitivity drivers may disproportionally contribute to the resultant community sensitivity and therefore the three γ factors are introduced as calibratable parameters. It is worth highlighting that the proposed approach could be extended, for example, by adding an additional employment concentration factor (i.e. the percentage of the catchment population employed in the agriculture industry) as a supplementary approach to account for the degree of reliance on agriculture in terms of local livelihoods.

Figure 2 provides a visual demonstration of the idealised trajectory of the Sensitivity variable over time (as discussed in Sect. 2.1). Figure 2c in particular depicts the effect of the macro-scale contextual parameters, should all other factors be equal. In this example, the regional climate regime is used for illustrative purposes, whereby three idealised catchments distinguished only by the level of water abundance (α) are examined. As can be seen, it is expected that the more arid the catchment (i.e. the lower the α) the greater will be the amplitude of oscillation away from the baseline state, as the relative change in V is amplified as α decreases. Thus the magnitude of any of the macro-scale parameters (in this case, α) will serve to amplify or subdue the degree of oscillation, thereby reflecting the extent to which different regional/ national regimes translate change in water balance drivers from a feedback point of view.

The Sensitivity state variable, as defined, represents the average community sensitivity. However, as noted earlier, there are numerous conditions under which collective norm adoption and action occur (Kinzig et al., 2013). The use of a "social ostracism" agent-based model has been demonstrated by Tavoni et al. (2012) and Lade et al. (2013), which allows for a departure from collective co-operation at a socially optimal level, by a subset of "defectors" that seek to maximise self-interest. Tavoni et al. (2012) show that the level of ostracism displayed towards defectors can play an important role in shaping the nonlinear dynamics. Thus, the defector fraction, $\omega = P_{\mathrm{d}}/P_n$, may be incorporated as a state variable within a model that acts to modify the degree of collective sensitivity within the catchment community.

3.6 Behavioural response (χ) function

Within the model framework the two key drivers of the χ function are the Sensitivity (V) and Demand (D_{E}) variables (see Fig. 1). The drivers effectively determine the degree and direction of overall impetus for action. This impetus then potentially translates into behavioural change in each of three components, namely the rate of water extraction (R_{E}), the area of land cleared for the purposes of economic development (A_{C}), and the amount of storage due to engineering structures such as dams and weirs ($S_{\max}$). These variables are all supported by the literature as signifying human induced change on watersheds (Falkenmark, 1979; Vörösmarty et al., 2005; Gregory, 2006) and would feed directly into the hydrology model as appropriate.

The χ response function that determines the overall impetus for action is designed to have a positive value to indicate a stimulus towards more enviro-centric measures, and a negative value to denote a drive towards more anthropocentric measures. In the simplest sense this can be composed as (Fig. 5)

$$\chi = f\left(V^{*}\right) - f\left(D_{\mathrm{E}}\right), \tag{5}$$

where V^{*} is a normalised sensitivity metric developed below. As a general premise, decreasing sensitivity levels would be expected to be associated with higher annual rates of water extraction, land clearing and dam building, to a point, while the converse is expected to hold true for increases in sensitivity levels. The sensitivity–response link has been made in the literature previously (Leichenko and O'Brien, 2002) and is broadly consistent with what has been observed in the development trajectories of river basins outlined earlier. Although this deals with the direction of an expected shift in the χ function, a number of further hypotheses are put forward in terms of the timing and magnitude of such shifts. Firstly, it is believed that upward (i.e. positive) movements will be observably more "sticky" and demonstrate a greater time lag in response when compared with downward (i.e. negative) movements in χ, as the former seeks to "reverse" behaviour. Secondly, it is expected that a catchment's baseline sensitivity levels will affect the magnitude and timing of management action. For instance, catchments operating at generally higher levels of the sensitivity scale (e.g. arid rural catchments) that experience an increase in sensitivity level over a period might be expected to show a more immediate and severe management response, relative to catchments operating at the lower end of the sensitivity scale experiencing the same absolute increase in sensitivity level. Finally, it is expected that there will be points at both ends of the sensitivity scale beyond which there will be no observable change in management action.

A number of studies have also found evidence of a link between social networks (i.e. memberships of churches, sports clubs, volunteer organisations, political groups) and response (Buikstra et al., 2010; Sherrieb et al., 2010; Smith et al., 2012). Thus this function must consider the degree of social interaction and co-operativity within the community. Furthermore, it is important to note that whether and to what extent a community responds, is generally influenced by two variables: the magnitude of influence (derived from V) and the *capacity* of the community to respond (Chaskin, 2008),

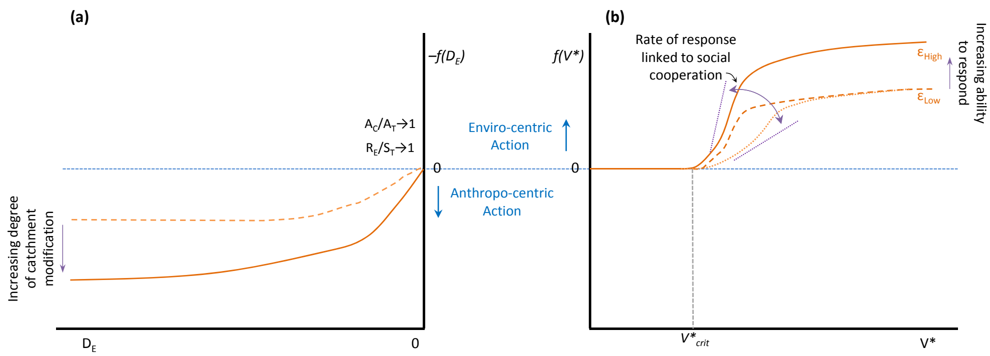

Figure 5. A hypothetical illustration of how the Behavioural Response functions vary according to: **(a)** the change in catchment demand for expansion, D_E, and; **(b)** the change in collective community sensitivity, V^*. The functions can be customised according to factors such as community cooperation or technological capability for enacting modifications to the catchment as indicated by the different lines.

modelled here as

$$f\left(V^*\right) = \begin{cases} 0 & \text{for} \quad V^* \leq V^*_{\text{crit}} \\ \chi_{\max V}\left(\frac{(V^*)^\sigma}{\left(k_\mathrm{V}^\sigma + (V^*)^\sigma\right)}\right) f(\varepsilon) & \text{for} \quad V^* > V^*_{\text{crit}}, \end{cases} \tag{6a}$$

where χ is proposed to follow a sigmoidal response function based on V^*, calculated at time t as

$$V^* = \frac{\Delta V}{V_{\max} - V_t}, \tag{6b}$$

where $V_{\max}$ is an arbitrary constant reflecting the maximum sensitivity of the particular community, and the term $V_{\max} - V_t$ scales the incremental change in sensitivity to increase χ as the baseline sensitivity approaches the maximum. In Eq. (6a), σ is a co-operativity function used to modify the rate at which χ will change (Schwarz and Ernst, 2009). It is intended to be related to the degree to which the community will collectively respond to a change in sensitivity levels, and can be calculated based on the defector fraction within the community, ω, or other relevant proxy, such as the percentage of the catchment population holding memberships in social organisations. The term $f(\varepsilon)$ captures the propensity for action based upon the national capacity to act in terms of financial and technological resources (based on the country's level of development, whereby $\varepsilon = \frac{E_{\mathrm{cN}}}{E_{\mathrm{cS}}}$ such that it reflects the national rate of development beyond a baseline subsistence economy).

The second driver of the χ function can be thought of as the degree of inducement for agricultural expansion (D_E). It is composed of two primary driving components (population growth and the relative importance and growth of agriculture in the economy) which may act independently or in tandem, limiting variables relating to the available land and water resources, and a moderating variable reflecting efficiency improvements in resource utilisation. Such an approach is similar to that found by Barbier (2004) to adequately reflect the rate of land-use change in favour of agriculture in developing economies. The population will thus be motivated to change their interaction with the catchment land surface and water balance in response to the demand for agricultural development as follows:

$$f\left(D_\mathrm{E}\right) = \chi_{\max\mathrm{D}}\left(\frac{D_\mathrm{E}}{\left(k_\mathrm{D} + D_\mathrm{E}\right)}\right), \tag{7a}$$

where the Monod equation above is proposed to reflect the response function based on D_E. This is calculated at time t as

$$D_\mathrm{E} = \left[\frac{\Delta P_n}{P_n^t} + f\left(Z_\mathrm{C}\right)\right]\left(1 - \frac{A_\mathrm{C}}{A_\mathrm{T}}\right)\left(1 - \frac{R_\mathrm{E}}{S_\mathrm{T}}\right) f(\zeta), \tag{7b}$$

where $\frac{\Delta P_n}{P_n^t}$ is the population growth rate (similar to Barbier's (2004) rural population growth rate) and $f(Z_\mathrm{C})$ is a function of structural driving variables that could comprise agricultural export share, growth in agricultural value added, and/or agricultural crop yield (Barbier, 2004). The extent of development is mitigated by the extent of "capacity usage" of underlying natural resources within the catchment, namely land ($A_\mathrm{C}/A_\mathrm{T}$) and water ($R_\mathrm{E}/S_\mathrm{T}$) resources. The capacity usage factor is included as management decisions are progressively less likely to acquiesce to expansion pressures as usage levels approach the capacity (i.e. land limited, $A_\mathrm{C}/A_\mathrm{T} \rightarrow 1$; or water limited, $R_\mathrm{E}/S_\mathrm{T} \rightarrow 1$). The variable ζ is a composite efficiency metric that captures the improvement in existing land and water utilisation as a result of implementing efficiency measures (e.g. rainwater harvesting or agricultural intensification through the application of more efficient farming technologies). It therefore acts to mediate demand for the underlying resources by enabling a degree of expansion that is not reliant on further resource exploitation. This term is thus driving humans to more actively modify the catchment water balance in favour of development, and will slow down as opportunities for further development reduce.

The overall behavioural response (in terms of magnitude and direction) is then able to drive each of the components of management action that make up the response model: ΔR_E determines the change in annual rate of extractions, ΔA_C reflects the change in annual land clearing, and $\Delta S_{\max}$ is the annual change in storage capacity. Each of the management response equations would then take the form, for example, of

$$\frac{\mathrm{d}R_{\mathrm{E}}}{\mathrm{d}t} = \eta_{\mathrm{RE}} f_{\mathrm{RE}}(\chi) \tag{8a}$$

$$\frac{\mathrm{d}A_{\mathrm{C}}}{\mathrm{d}t} = \eta_{\mathrm{AC}} f_{\mathrm{AC}}(\chi) \tag{8b}$$

$$\frac{\mathrm{d}S_{\mathrm{max}}}{\mathrm{d}t} = \eta_{\mathrm{Smax}} f_{\mathrm{Smax}}(\chi), \tag{8c}$$

which then each feed into the hydrology model. In the above equations, η is the translation factor that captures the extent to which χ manifests in this particular water management action. The closure relationships used in Eq. (8) will be highly specific to any given context, thus each of these must be defined upon local catchment conditions, and are therefore left for practitioners to determine on a case-by-case basis. By way of example, $f_{\mathrm{AC}}(\chi)$ in Eq. (8b) might be parameterised as $f_{\mathrm{AC}}(\chi) = \chi^{\mathrm{a}}$ for case study site A, whereas it may take the form $f_{\mathrm{AC}}(\chi) = 2\chi^{1/\mathrm{b}}$ for case study site B, due to the distinctness of circumstances and response patterns between the sites.

4 The conceptual framework in practice

To demonstrate how the generic conceptual framework can be applied to analysing the evolution of different catchments, two agricultural catchments located in Australia have been selected for further illustration: the Murrumbidgee catchment in New South Wales and the Toolibin catchment in Western Australia. The case studies have been chosen for illustration purposes based on differences in size and water balance drivers in each agricultural catchment. The Murrumbidgee catchment is examined as a large-scale irrigated river basin catchment, whilst the Lake Toolibin catchment provides a contrasting small-scale rainfed lake catchment. In light of these differences, case-study-specific manifestations of the generic conceptual framework are made possible, and tailored application of the model to unique catchment histories can be explored. Prior to full implementation of the model for these case studies, this paper outlines the approach to parameterisation of the above framework, and in particular the necessary closure relationships described above in general terms. Table 1 summarises how the differences between these two catchments are to be captured through application of the conceptual framework and how parameterisation of the closure relationships could be pursued. A stylised model that incorporates several of the above components of the framework is run for a 110-year timescale (1900–2010) for each catchment using various simplifying assumptions and some generic parameterisations to demonstrate the approach to application of the model.

4.1 Murrumbidgee catchment

The trajectory of the Murrumbidgee catchment, an area of 8.4 million hectares located within the greater Murray–Darling River basin in southern New South Wales, has been described in detail by Kandasamy et al. (2014). The nation's capital city, Canberra, is located within the catchment, along with numerous other regional towns and inland cities. The Murrumbidgee River, at 1600 km long, supports a diverse range of fish and bird species, along with numerous seasonal wetlands, nature reserves and riparian vegetation. The catchment is predominantly used for grazing and irrigated crop farming. The advent of increasingly extensive environmental problems in recent decades, including the adverse impacts of nutrient runoff and salinisation, has prompted concerted remedial efforts at local, regional and state levels. It therefore presents a compelling case study for the implementation of the socio-hydrology framework on a large-scale area.

In addition to the extensive clearing of native vegetation to make way for agricultural expansion, humans vastly altered natural flow regimes throughout the catchment as a consequence of the large-scale development of dams and weirs for irrigation farming, which occurred up to 1970 (Kandasamy et al., 2014). A number of environmental problems began to appear in the latter half of the 20th century, and became progressively more serious. The considerable reallocation of water to irrigation led to the first major environmental crisis facing the sustained health of riverine and wetland ecosystems, with the diversion of up to 90 % of the Murrumbidgee river's natural flow to irrigation causing a sharp decline in residual flows to the environment (Kandasamy et al., 2014). Marked reductions were recorded in water bird and native fish populations in the Murrumbidgee basin as a result.

The second major issue to arise pertained to the excessive discharge of nutrients from sewage treatment plants and farming practices into the Murray River, causing a sharp decline in water quality and threatening riverine ecosystems. In fact, one of the worst blue-green algal blooms anywhere on record occurred along more than 1000 km of the Murray–Darling rivers in the summer of 1991–1992. Furthermore, the widespread replacement of deep-rooted native vegetation with shallow-rooted agricultural crops caused a rise in groundwater tables throughout the catchment, thereby dissolving salts stored in the soil profile and transporting them to the surface. This led to the third key issue – land salinisation – which threatened agricultural productivity, local livelihoods and the useful life of existing infrastructure, as well as having detrimental impacts on riverine ecology. This predicament was exacerbated by the use of irrigation, which created pervasive waterlogging (Kandasamy et al., 2014).

All three of these issues acted as management response levers with varying degrees of severity, albeit with a time lag. Escalating concern for the health of the environment over the 1990s and 2000s spurred remedial action at the highest levels of government. A number of measures are being

Table 1. Application of the conceptual framework: comparison of two Australian catchments.

Model component	Variable/ Function	Irrigated farming catchment (1900–present)	Dryland farming catchment (1900–present)
Summary		We have chosen to present an idealisation of the Murrumbidgee catchment in eastern Australia, based on the data and narrative presented in Kandasamy et al. (2014). Please see Sect. 4.1 for further context and detail.	We have chosen to present an idealisation of an example catchment in the Wheatbelt of Western Australia (Lake Toolibin catchment) based largely on the figures and narrative presented in Allison and Hobbs (2004). Please see Sect. 4.2 for further context and detail.
Water balance	S_{GW}, S_{US}, S_Q, S_W $S_{max}, Q_S, Q_{SS},$ $Q_{ES}, Q_{Rin}, Q_{Rout}$	S_Q is relatively large as the catchment relies heavily on irrigation water for farming, and catchment flows became increasingly diverted to storages over the past 100 years. In the recent severe drought this led to the over allocation of water and consequent decline in wetland storages and provision of environmental flow. For the purpose of this illustrative example, a full water balance has not been constructed. Instead, storage capacity data (S_{max}) has been sourced directly from Kandasamy et al. (2014) (Fig. 6c). For simplicity, water availability in a given year is calculated using the ratio of actual annual discharge to average discharge. To the extent that this ratio is greater than 1 (i.e. wet year), $S_Q = S_{max}$, however where the ratio is less than 1 (i.e. dry year), S_{max} is reduced by half the shortfall to account for cumulative storage from previous years (Fig. 6e).	This is a semi-arid catchment with a low runoff coefficient, a deep saline groundwater aquifer and ephemeral wetland system. The natural woodland vegetation, crops and pasture are all supported by moisture in the vadose zone. The rise of the water table into the root zone caused by increased recharge will induce tree mortality and land degradation. S_Q is relatively small as a minor amount of surface water is diverted to farm dam storages to support livestock. In lieu of a water balance model, growing season (May–Sep) rainfall data has been sourced from the Bureau of Meteorology (Fig. 7c). Water availability in the vadose zone is then calculated using as a proxy the inverse of the soil moisture deficit (i.e. long-term mean growing season potential evapotranspiration less annual growing season rainfall) (Fig. 7e).
Population	P_n	Population numbers have been sourced from Kandasamy et al. (2014) (Fig. 6a).	Population numbers have been obtained from Australian Bureau of Statistics census data using the total of Shire of Narrogin and Town of Narrogin data as proxies for the catchment (Fig. 7a).
Economy Eq. (2a, b) E_C, E_{pc}	p_c, B_c, p_{wc}, A_C, A_D	Kandasamy et al. (2014) use the main crop, rice, as the economic proxy. We use their rice production data as a proxy for agricultural output in our model, $A_C\,(1-A_D)\,B_c$ (Fig. 6b). In a full model, p_c would be driven by the annual global price of rice, however for the purposes of this simplistic illustration p_c is assumed constant at USD 200. Furthermore, for simplicity we have assumed τ_A is 1, E_{ext} is zero, and the cost component, $\left[(c_A A_C\,(1-A_D)) + (p_{wc}U_c + p_{wp}U_p)\right]$, is constant at 50 % of gross revenue, thus foregoing a delineation of costs at this stage. In our illustration therefore, the evolution of E_{pc} is driven by rice production and population (Fig. 6f). In a full run of the model economic productivity would be a function of the annually evolving area of cleared land (A_C), degraded land (A_D), crop yield (B_c) relative to irrigation water applied, the cost of irrigation water (p_{wc}), as well as technological advancements affecting agricultural costs and productivity (e.g. mechanisation/pesticides).	Wheat is the main crop farmed in this region. In a full model, p_c would be driven by the annual global price of wheat, however for consistency across cases in this instance p_c is assumed constant at AUD 450, the long-term average price/tonne at 2004 prices. Wheat production data is calculated using $A_C\,(1-A_D)\,B_c$ (Fig. 7b), with data for both A_C and A_D sourced from Fig. 7 in Allison and Hobbs (2004) and tonne/hectare wheat yields (B_c) sourced from the Australian Bureau of Statistics. B_c begins at 0.5 tonne ha^{-1} in 1900, rises to 1.0 tonne ha^{-1} in the early 1940s, breaks the 1.5 tonne ha^{-1} mark in the early 1980s, remaining above this mark to the present. For consistency across cases, we assume τ_A is 1, E_{ext} is zero, and the cost component, $\left[(c_A A_C\,(1-A_D)) + (p_{wc}U_c + p_{wp}U_p)\right]$, is constant at 50 % of gross revenue. The evolution of E_{pc} is thus driven by wheat production and population (Fig. 7f).
Ecosystem Services Eq. (3) L_{ES}	$f(W_Q)$ $f(Q_{ES})$ $f(S_W)$ $f(A_N)$	Although there are numerous environmental issues in this catchment, the primary ones relate to the loss of biodiversity due to the diversion of natural flows and significant increases in nutrient runoff levels which have resulted in severe algal blooms. Data for Q_{ES} has been sourced from Kandasamy et al. (2014) using the annual ratio of upstream to downstream flow (Fig. 6d). Water quality (W_Q) indicator has been assumed to start at 1 and deteriorate to a low of 0.05 at the height of the algal bloom in the early 1990s, followed by a slight recovery up to 0.10–0.14 in the 2000s. A lag has been assumed such that a measurable impact on the water quality indicator is not detected until 1960. The closure relationships are then defined as follows: $f(Q_{ES}) = \begin{cases} 1 & Q_{ES} \geq 0.6 \\ {Q_{ES}}^{0.5} & Q_{ES} < 0.6 \end{cases}$ $f(W_Q) = W_Q^{0.6}$ Finally, L_{ES} is calculated as an equally weighted average of the moving 10 year averages of each of $f(Q_{ES})$ and $f(W_Q)$ (Fig. 6g).	Land clearing, consequent salinisation and secondary salinity in the lake (from saline surface water inflows and rising groundwater tables) are the overriding drivers of ecosystem degradation. Data for A_C has been obtained from Fig. 7 in Allison and Hobbs (2004), and A_N has been calculated as $1-A_C$ (Fig. 7d). To capture the increasing salt loads into the lake, a water quality (W_Q) indicator has been assumed to start at 1 and deteriorate to a low of 0.05 immediately prior to the diversion of saline flows into the lake, followed by a slight recovery up to 0.10–0.14 in the 2000s. A lag has been assumed such that a measurable impact on the water quality indicator is not detected until 1960. The closure relationships are then defined as follows: $f(W_Q) = \begin{cases} 1 & Q_{ES} \geq 0.6 \\ {W_Q}^{0.5} & Q_{ES} < 0.6 \end{cases}$ $f(A_N) = \begin{cases} 1 & Q_{ES} \geq 0.75 \\ {A_N}^{0.6} & Q_{ES} < 0.75 \end{cases}$ Finally, L_{ES} is calculated as an equally weighted average of the moving 10 year averages of each of $f(W_Q)$ and $f(A_N)$ (Fig. 7g).
Sensitivity Eq. (4)	α, β, φ	The UNEP Aridity Index was used to calculate an α of 0.545 for the Murrumbidgee catchment (using annual rainfall of 600 mm and annual potential ET of 1100 mm, *Australian Bureau of Meteorology*). The HDI for Australia is currently 0.864 (β) and the CPI for Australia is currently 85 out of a maximum of 100 (φ). These three values were set as constants for the period for simplicity, however a full model run could use time series data.	The UNEP Aridity Index was used to calculate an α of 0.342 for the Lake Toolibin catchment (using annual rainfall of 410 mm and annual potential ET of 1200 mm, *Australian Bureau of Meteorology*). The HDI and CPI are consistent across cases. These three values were set as constants for the period, however a full model run could use time series data.
	$-\widetilde{S}_x\gamma_s$	Community sensitivity in an irrigated catchment is predominantly a function of the amount of water available in storage (i.e. S_Q). Thus storage infrastructure development would be expected to decrease sensitivity levels, while periods of prolonged drought and declining water stocks would increase sensitivity (Fig. 6h). The change in available stored water (ΔS_Q) has been calculated using the difference in the present period's 10 year average value and the previous time step's 10 year average. This approach is taken as community sensitivity would begin to reflect the impacts of a new reservoir for instance for some time prior to its inauguration. The reference value ($\overline{S_Q}$) is calculated as follows: $\overline{S_Q} = \begin{cases} 100\,\text{yr ave } S_Q & \text{pre} - 1965 \\ 20\,\text{yr rolling ave} S_Q & \text{post} - 1965 \end{cases}$ Finally, γ_s has been set at 100 for the whole period.	Community sensitivity in a rainfed catchment is primarily a function of crop productivity, which is linked to water storage in the vadose zone – $f(S_{US})$. To a lesser extent, dam storage could also be considered since small farm dams service livestock and hence sheep farmers will be sensitive to water availability in local dams, however this has been ignored for simplicity in the present example. As expected, sensitivity levels oscillate with reasonable frequency depending on rainfall (Fig. 7h) The change in soil moisture (ΔS_{US}) has been calculated using the difference in the present period's 10 year average value and the previous time step's 10 year average. The reference value ($\overline{S_{US}}$) is calculated as: $\overline{S_{US}} = -(\overline{\mathrm{ET_P}} - \overline{P})$ over the growing season for the entire period under investigation. Finally, γ_s has been set at 40 for the whole period.
	$-\widetilde{L_{ES}}\gamma_{es}$	A marked deterioration in ecosystem services that are sufficiently observable so as to detract from the community's quality of life would be expected to increase sensitivity levels (Fig. 6h). ΔL_{ES} has been calculated as $L_{ES}^{t} - L_{ES}^{t\text{-}1}$. $\overline{L_{ES}}$ is calculated as the 20 year moving average of L_{ES}. Finally, γ_{es} is set at 400 for the whole period, as environmental issues in this catchment, and the wider Murray–Darling Basin of which it is a part, have been elevated to matters of national importance and security as the basin currently provides half of Australia's rice production (Kandasamy et al., 2014).	As with the case for the irrigated catchment, a large deterioration in ecosystem services that are sufficiently observable so as to detract from the community's quality of life would be expected to increase sensitivity levels (Fig. 7h). ΔL_{ES} has been calculated as $L_{ES}^{t} - L_{ES}^{t\text{-}1}$. $\overline{L_{ES}}$ is calculated as the 20 year moving average of L_{ES}. Finally, γ_{es} is set at 250 for the whole period.
	$-\widetilde{E_{pc}}\gamma_e(1+\delta)$	Economic prosperity within the catchment was increasing at various rates up to the 2000s when prolonged drought caused a sharp decline in economic well-being. Thus we would expect the economic component to cause a general downward pull on community sensitivity levels up to the point at which the turnaround occurred, at which stage sensitivity levels should feel upward pressure from economics (Fig. 6h). ΔE_{pc} is calculated as the difference in the current time step's 5 year average value and the previous time step's 5 year average. $\overline{E_{pc}}$ is calculated using a 10 year moving average. γ_e is set at 100 for the whole period. The GDP concentration metric for agriculture (δ) has been set to vary between 20 and 35 % in the first half of the 20th century (peaking at 35 % in 1950) and declining steadily to a its current value of 3 %, in line with the narrative in Kandasamy et al. (2014).	Economic prosperity occurred within the catchment in three large bursts related to land clearing policies and incentives in place in the State of Western Australia. The large decrease in prosperity in the 2000s was largely due to the significant land salinisation that became more widespread as a delayed impact of a legacy of land clearing (Fig. 7h). The calculation of ΔE_{pc} and $\overline{E_{pc}}$ as well as assumptions related to γ_e and δ have been held constant across both cases.
	V $\frac{\mathrm{d}V}{\mathrm{d}t}$	The starting value for V has been arbitrarily set at 100. $\frac{\mathrm{d}V}{\mathrm{d}t}$ has been calculated using 10 year averages of each of the underlying catchment-scale components outlined above. The macro-scale contextual closure relationships have in this case been defined as follows: $f(1-\alpha) = 1-\alpha,$ $f(1-\beta) = 1-\beta,$ $f(1-\varphi) = 1-\varphi$ The trajectories of V and $\frac{\mathrm{d}V}{\mathrm{d}t}$ appear in Fig. 6i. Change in community sensitivity over time displays a generally decreasing trend until the 1990s when widespread environmental degradation was observed. This would be expected to be exacerbated by drought conditions causing a simultaneous worsening in economic conditions and water security, which is observable in Fig. 6i.	All assumptions and calculation approach in relation to V and $\frac{\mathrm{d}V}{\mathrm{d}t}$ is held constant across cases. Their trajectories appear in Fig. 7i. As expected, change in community sensitivity over time exhibits a sharp decline early in the period due to the onset of expansion activities, levels out and then shows an increase from the 1990s onwards as environmental damage became widespread and began to threaten agricultural productivity.
Behavioural Response χ *Driver Functions* Eqs. (6a, 6b) Eqs. (7a, 7b)	$f(V^*)$ $f(D_E)$	To calculate V^*, V_{max} has been set to 120 and V_t is calculated using the average of V for the previous 10 time steps. $f(V^*)$ is calculated based on the following assumptions: V^*_{crit} is set at 0.001; χ_{maxV} is 100; k_V is 1; σ is assumed to remain constant at 2; and $f(\varepsilon)$ has been set to 1 for simplicity. To calculate D_E, the following assumptions have been made: Z_C is proxied by agricultural export share, which has been set at an initial value of 80 % in 1912, declining to 70 % in 1950 and assumed to decline steadily to its current value of 21%; $\frac{A_C}{A_T}$ and $\frac{R_E}{S_T}$ are assumed to decline from a starting value of 0 to their present value of 0.9; and ζ is assumed to be zero for simplicity. The closure relationship $f(Z_C)$ is defined as $\frac{1}{Z_C}$. For simplicity and consistency, $f(D_E)$ is then calculated by setting χ_{maxD} to 100 and k_D to 1. We would expect expansion-driven demand to be high in the first half of the 20th century reflecting the aggressive expansion that occurred due to the importance of agriculture to the national economy, gradually slowing over time as agriculture declined in importance, and the underlying resources approached capacity. This trajectory is observable in Fig. 6j.	All assumptions and calculation approach in relation to $f(V^*)$ and its components are held constant across cases. It is worth noting that revegetation efforts have been underway during the 2000s throughout the Wheatbelt, which is reflected in the recovery in ecosystem services in this period. The ratio $\frac{A_C}{A_T}$ is calculated based on the data sourced from Fig. 7 in Allison and Hobbs (2004). All other assumptions and calculation approach in relation to $f(D_E)$ and its components are held constant across cases. The trajectories for both $f(V^*)$ and $f(D_E)$ are observable in Fig. 7j.
χ *Response Functions* Eqs. (8a, 8b, 8c)	η_{RE} η_{AC} η_{Smax}	All action functions are relevant: η_{RE}, η_{AC} and η_{Smax}, since catchment modification has involved land clearing, water extraction and major water infrastructure expansion, thus these would each be set to 0.33.	The predominant catchment modification has been via land clearing, such that the translation factor η_{AC} will be close to 1. The amount of surface water being stored, as governed by η_{Smax}, is minor, and household and other water needs are provided by scheme water, such that the factor η_{RE} is also negligible.

examined and gradually implemented, including projects aimed at increasing water usage efficiency, economically motivated trading mechanisms, more stringent restrictions on water licences, the relocation of storage infrastructure further downstream to allow for greater inundation of wetlands and riparian areas along the length of the river, and policies designed to reallocate water in favour of the environment (Horne, 2012).

Kandasamy et al. (2014) interpreted this historical trajectory in terms of a pendulum swing between people and the environment. It is hypothesised that in fact this pendulum swing is indicative of a gradual change in the community's Sensitivity state variable over time. As the adverse impacts of development and land clearing manifested themselves throughout the catchment, the community's sensitivity to an imminent decline in its quality of life increased, which drove the shift in response function components. Table 1 explains the assumptions used for a preliminary examination of this catchment. Figure 6a–d reflect the inputs used in the model (based on information provided in Kandasamy et al., 2014), whilst Fig. 6e–h show the evolution in each of the underlying components of the Sensitivity state variable. Figure 6i–j demonstrate the evolution of the Sensitivity and Response variables over the time period.

4.2 Lake Toolibin catchment

The Toolibin catchment covers a much smaller area of some 48 000 hectares, located in the Blackwood River basin in Western Australia's "Wheatbelt" region. The Wheatbelt was subject to large-scale development over the course of the 20th century, the most drastic of which occurred throughout the 1949–1969 period (Allison and Hobbs, 2004). Such rapid expansion led Conacher (1986) to comment that "no other area in the world as large as the Wheatbelt has been cleared of its native vegetation over so short a period of time". The Toolibin catchment reflected this rate of growth, with more than 90 % of the catchment cleared of native vegetation by the early 1970s for dryland farming purposes, predominantly sheep and grain farming. Lake Toolibin, located within the catchment, is a Ramsar-listed wetland of international importance due to its diversity of water birds, many of which breed on the lake, now recognised as a "threatened ecological community" (Munro and Moore, 2005). In much the same trend as that observed in the Murrumbidgee catchment, the Toolibin catchment began to exhibit signs of increasingly severe environmental degradation, in the form of rising groundwater levels and dryland salinisation, which endangered both agricultural productivity and biodiversity. In response to the threat of widespread environmental deterioration, the State government and community began to take remedial action in the 1990s in an attempt to halt, and potentially reverse, the adverse impacts of development. The Lake Toolibin catchment thus presents an ideal case study for the application of the socio-hydrology framework on a small scale.

The fundamental environmental issue to arise throughout the catchment is that of land and water salinisation. As Hatton et al. (2003, p. 342) note, "while the impacts of agricultural clearing through salinisation extend across the continent, they are particularly severe and extensive in the Wheatbelt... with up to 8.8 million hectares (33 %) at risk by 2050". The "at risk" land area for the Toolibin catchment translates to 24 %, with 8 % already salt-affected (George et al., 2005). Such a deterioration would result in extensive damage to infrastructure (roads, rail, town sites), remnant vegetation, plant species, wetlands and river systems. Unique features of the landscape in the Wheatbelt, including the Toolibin catchment, are the exceptionally low land and hydraulic gradients. Historically, the pre-clearing hydrogeology, climate and native vegetation characteristics of the region produced hydrological systems with relatively deep groundwater tables (> 30 m), remarkably high rates of evaporation, very low surface flows, and a build-up of salts stored in the unsaturated root zone. Pervasive clearing throughout the region caused a drastic shift in these defining characteristics, triggering a significant rise in groundwater tables and consequent mobilisation of stored salts, a sharp decrease in evaporation rates, frequent waterlogging due to degraded soils, and substantial increases in surface runoff leading to the discharge of saline water into rivers and lakes (Hatton et al., 2003). These impacts have been further compounded by the high variability in the amount and spatial distribution of annual rainfall. Furthermore, post-clearing hydrological equilibrium has yet to be reached, with groundwater levels continuing to rise in the majority of the Wheatbelt.

A number of remedial mechanisms have been considered and applied within the catchment, in an attempt to combat the unfavourable aspects of a legacy of development. Such measures include the installation of a gate to divert saline surface water around the lake and control the inflow of freshwater to the lake, continuous groundwater pumping to maintain groundwater tables below the lakebed to a maximum of 1.5 m, the installation of shallow interceptor drains and the revegetation of native plant species (George et al., 2005; Hatton et al., 2003).

Table 1 outlines the assumptions used in a preliminary investigation of this catchment. Figure 7a–d reflect the inputs used in the model (based on information provided in Allison and Hobbs, 2004), whilst Fig. 7e–h show the evolution in each of the underlying components of the Sensitivity state variable. Figure 7i–j demonstrate the evolution of the Sensitivity and Response variables over the time period.

4.3 Discussion of preliminary case study results

It is important to note that the fully parameterised dynamic coupled model has not been applied at this stage and further work is required to link the behavioural response driver (χ) to water balance actions (R_{E}, A_{C}, $S_{\max}$) in each case. However, the results of a stylised model that uses simplifying

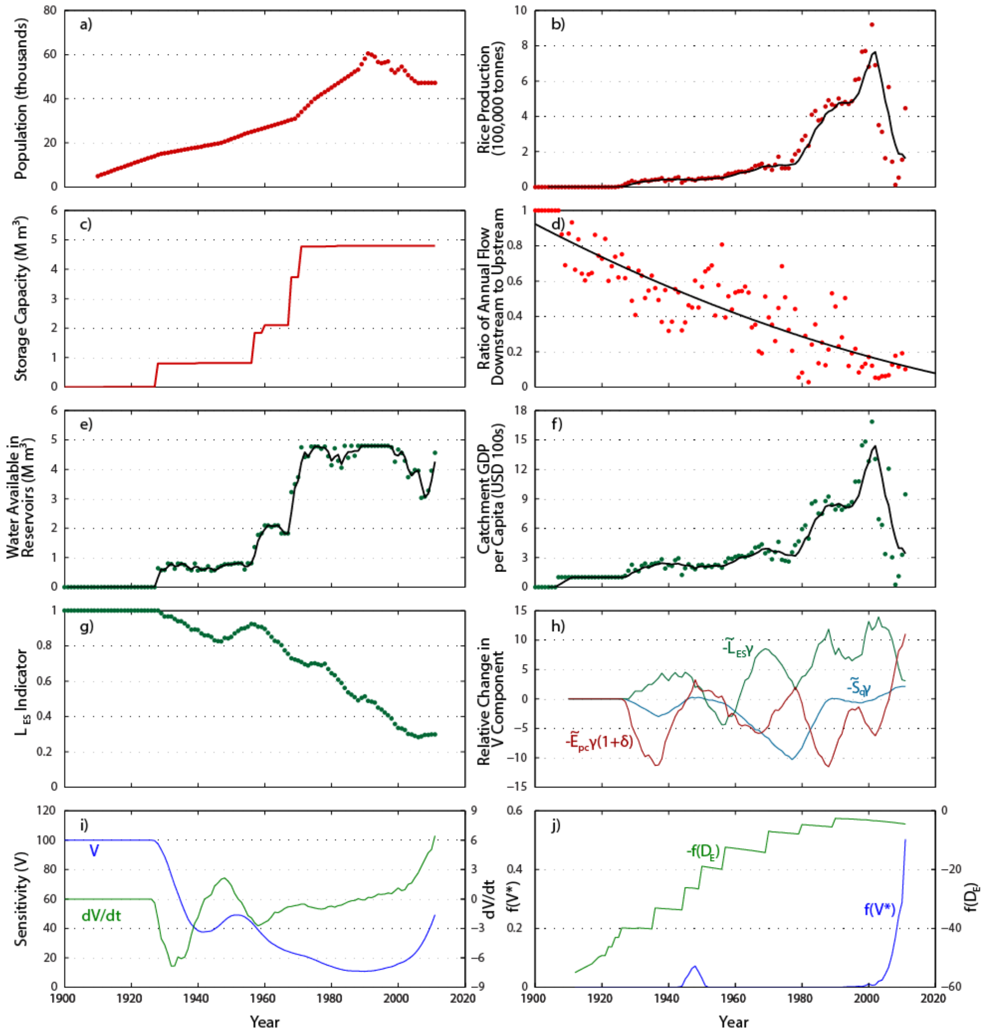

Figure 6. Stylised model data and results for a semi-idealised irrigated catchment, e.g. Murrumbidgee Catchment. **(a)** Catchment population growth (P_n). **(b)** Rice production. **(c)** Development of dam and reservoir storage infrastructure ($S_{\max}$). **(d)** Environmental flow (Q_{ES}), taken as the ratio of annual flow observed at Balranald to that observed at Wagga Wagga. **(e)** Water availability stored in dams and reservoirs (S_{Q}) (data for **(a–e)** sourced from Fig. 4 in Kandasamy et al., 2014). **(f)** Catchment GDP per capita (E_{pc}). **(g)** Lumped "lifestyle" ecosystem services indicator (L_{ES}). **(h)** Evolution of the three underlying catchment-component drivers of Sensitivity (V) over time. **(i)** Evolution of Sensitivity (V) and $\mathrm{d}V/\mathrm{d}t$ over time. **(j)** Evolution of behavioural response component functions, $f(V^*)$ and $f(D_{\mathrm{E}})$, over time.

assumptions and generic parameterisations provide a preliminary indication of the trajectories of certain key framework components. Both case studies show similar patterns in population growth and crop production over the time period (Figs. 6a–b, 7a–b). These patterns reflect growth rates in both agricultural expansion and intensification up to a point, whereupon environmental degradation and climate factors constrained further growth. Both catchments thus demonstrate similar GDP per capita trajectories (Figs. 6f, 7f), which are in line with the theory outlined in the "Economic-Population Loop" discussed in Sect. 2.1. A similar trajectory is also evident in lifestyle ecosystem services (Figs. 6g, 7g) although the mix of proxies used to calculate the index differ in each case according to contextually relevant issues, with for instance, environmental (or residual) flow being an appropriate indicator in a river basin catchment such as the Murrumbidgee, whilst percentage of natural deep-rooted vegetation is the more appropriate indicator in the Lake Toolibin catchment (Figs. 6d, 7d). A key point of difference between the cases is in relation to the supply of water for agriculture, with the Murrumbidgee catchment reliant on irrigation and reservoir storage (Fig. 6c, e) while the Lake Toolibin catchment relies on rainfall and soil moisture during the growing season (Fig. 7c, e). By altering the appropriate measure driving sensitivity to the water balance, it can be seen how the model is able to adapt to context-specific circumstances. It is worth noting that the framework proposed in this paper is general; however the way it will manifest in various

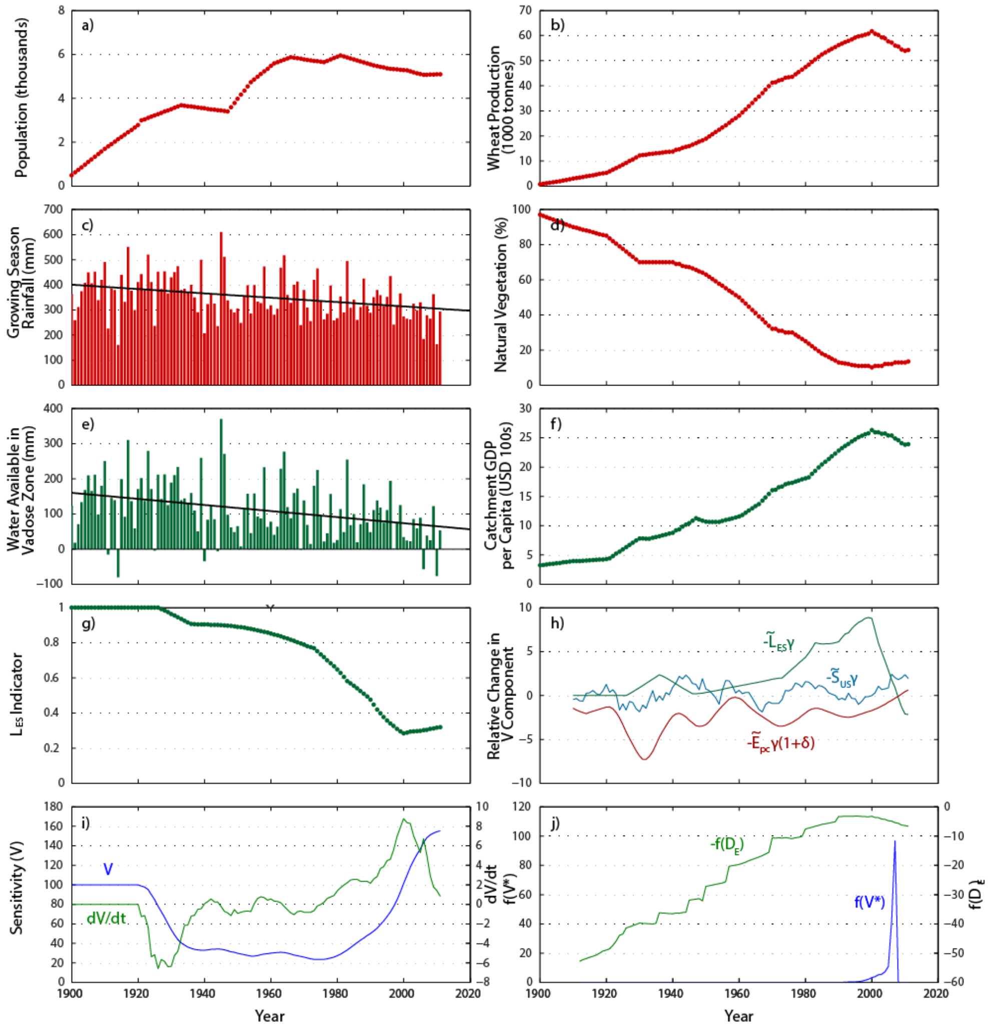

Figure 7. Stylised model data and results for an idealised rainfed catchment e.g. Lake Toolibin catchment. **(a)** Catchment population growth (P_n). **(b)** Wheat production. **(c)** Growing season rainfall (May–Sep). **(d)** Percentage of Natural deep rooted vegetation cover (A_N). **(e)** Growing season water availability stored in the vadose zone (S_{US}). **(f)** Catchment GDP per capita (E_{pc}). **(g)** Lumped "lifestyle" ecosystem services indicator (L_{ES}). **(h)** Evolution of the three underlying catchment-component drivers of Sensitivity (V) over time. **(i)** Evolution of Sensitivity (V) and dV/dt over time. **(j)** Evolution of behavioural response component functions, $f(V^*)$ and $f(D_E)$, over time (all data sources outlined in Table 1).

sites will depend upon local environmental conditions, such as whether surface or groundwater is exploited by humans and for what purpose. Accordingly, although groundwater extractions do not comprise a significant component of either of the specific case studies examined, the model incorporates the functionality required to appropriately account for more significant groundwater depletion in sites where it is a more crucial component of the cycle.

Despite the contextually specific inputs used in each case, the theoretical framework has the functionality to translate these into comparable component drivers of community sensitivity (Figs. 6h, 7h). It can be seen in both cases that environmental factors have an increasingly upward pull on sensitivity levels, whereas economic factors have a predominantly downward pull on sensitivity levels for most of the period (as economic prosperity increases) with a reversal towards the end of the period as environmental factors begin to impact upon economic yields. Sensitivity to water availability shows a much stronger downward pull on sensitivity levels in an irrigated catchment where the water balance is being more actively modified, relative to a rainfed catchment where sensitivity levels oscillate about the mean with high frequency. As hypothesised in Fig. 2, the same broad trajectory in V and $\frac{dV}{dt}$ is perceptible in both cases (Figs. 6i, 7i). Whilst we note the simplification inherent in the assumptions employed in these model runs, the Expansion, Contraction and Recession phases illustrated in Fig. 2 are evident in the results. Finally, behavioural response components also exhibit a similar pattern over time, despite these responses manifesting in different ways (Figs. 6j, 7j). $f(D_\mathrm{E})$ displays a strong impetus for agricultural expansion in the early part of the period, gradually decreasing over time to approach 0. In the case of the Murrumbidgee catchment, development was achieved by way of both land clearing and the construction of storage infrastructure that actively diverted the natural flow regime, while expansion activity in Lake Toolibin was primarily achieved through land clearing. $f(V^*)$ meanwhile stays at 0 for the majority of the period, with a sharp increase towards the end of the period, propelling the enviro-centric measures instituted in each of these catchments from the mid-1990s onwards. Remediation efforts in the Murrumbidgee catchment are largely focused on increasing environmental flows, in contrast with remedial efforts in the Lake Toolibin catchment, which have focused predominantly on revegetation, drainage and diversion of saline water.

4.4 A final word on limitations

As can be seen from the two case studies outlined above, the generic framework provides a flexible basis from which to investigate context-specific case studies. In conjunction with differences in the macro-scale contextual factors depending on the climate, socioeconomic and political context of study sites, this framework presents a workable compromise between accounting for context-specific idiosyncrasies and system-scale dynamics, in order to observe centurial trends across various geographic locations. We have put forward a conceptual framework that seeks to reconcile a number of theories and findings from diverse research streams, and in doing so, aimed to introduce novel components in the way the coupled system may be viewed and analysed.

It is important to note that there are in reality myriad feedbacks within and amongst the human and hydrological systems. Conceptualising a model that is a comprehensive reflection of all these dynamics is not the intention of this paper, if such an endeavour is even feasible. The model presented in this paper has only sought to represent and capture the most vital high-level features of the coupled system as a starting point to explore dynamics at the catchment system scale. In this regard it is a stepping stone, with the potential to be refined in future iterations as diverse case studies are examined globally and we acknowledge the specific nature of the parameterisations we have used may indeed evolve over time. As our knowledge of socio-hydrology, and indeed psychology, advance, this model will undoubtedly be revised and enhanced, and there is scope for case-study-specific innovations to be applied through careful parameterisation of the closure functions.

Furthermore, Srinivasan et al. (2013) illustrate how any coupled human–nature system is comprised of different temporal and spatial scales. Incorporating interactions between fast and slow processes, as well as between micro and macro variables, renders the examination of integrated adaptive system behaviours extremely complex (Liu et al., 2007b). Whether it be climate change, ecosystem degradation, socioeconomic development, or changes to the catchment's hydrological signature, such shifts generally occur gradually over decadal to centurial scales. Having said that, certain large-scale drastic events can and do occur (such as a political coup, a market crash, a widespread algal bloom or natural disaster) which act as external shocks that alter Sensitivity levels on an immediate scale.

The authors have drawn on recent concepts and findings in the system dynamics and SES literature (e.g. Lade et al., 2013; Schlüter et al., 2012) where idealised models relevant to SES issues have been used to explore theoretical state-space relationships and response trajectories. We acknowledge that numerical calibration of a model built in line with this framework will be a challenging undertaking; however, we believe that over time sufficient case study examples will emerge which could cover a range of gradients, and slowly provide confidence in the more complex parameterisations. Further, the model framework is presented in completeness to provide a larger vision and guide to hydrology modellers, but when it comes to specific implementations aspects of the model can be simplified or removed from the key components to make it more tractable or subject to systems analysis techniques.

5 Conclusions

The sustainable management of global freshwater resources remains an urgent challenge, with the spotlight in recent years increasingly being placed upon the importance of understanding the complex dynamics of coupled human–hydrology systems. This paper has sought to identify the fundamental drivers of one of the key socio-hydrology feedback loops, termed the "Sensitivity Loop", with the ultimate goal of understanding what drives human behaviour and management decisions in the hydrological context. A generic conceptual modelling framework has been put forward which posits a novel construct, a composite Community Sensitivity state variable, as the crucial driver of behavioural response in the human system. The six basic components of the framework are outlined in detail, and two Australian case studies are examined to illustrate how the generic framework would be tailored to specific contextual applications by way of localised closure relationships. Furthermore, by including a number of macro-scale contextual parameters, the framework has the capacity to be applied across climate, socioeconomic and political gradients globally. Indeed, the model is intended to normalise along each of these gradients.

The model framework is now being formally developed on each of the two case studies highlighted, with the aim of adding further international case studies to stress test the basic assumptions of the model and refine closure relationships to the extent that more universal principles are found to apply. The conceptual framework presented in this paper is seen as a step towards illuminating our knowledge of the workings of these complex coupled systems. It will no doubt be refined through empirical application and future iterations, and as additional research comes to light in the underlying disciplines (e.g. psychology, ecological economics) that can more fully inform various aspects and components of the model.

Acknowledgements. The authors would like to thank Associate Professor Jaya Kandasamy at the University of Technology, Sydney, for making available the data used in Kandasamy et al. (2014). The authors are also sincerely grateful to the reviewers for their suggestions and feedback, which were instrumental in significantly improving the paper. Finally, Yasmina Elshafei would like to acknowledge the PhD scholarship funding provided by the Australian Postgraduate Award and the University of Western Australia.

Edited by: H. H. G. Savenije

Appendix A

Table A1. Table of variables.

Variable	Explanation	Units
S_{max}	Total man-made water storage capacity in catchment (including dam and reservoir storage)	m^3
S_T	Total available water in catchment (made up of groundwater storage, vadose zone storage and reservoir storage)	m^3
S_{GW}	Water stored in groundwater store	m^3
S_{US}	Water stored in the vadose zone	m^3
S_Q	Water stored in reservoirs (derived from re-routing of surface/river flows or groundwater pumping)	m^3
S_W	Wetland storage	m^3
h_{WT}	Water table height	m
Q_S	Surface runoff	m^3 year^{-1}
Q_{SS}	Subsurface flow	m^3 year^{-1}
Q_{Rin}	Flow diverted to reservoirs	m^3 year^{-1}
Q_{Rout}	Flow released from reservoirs	m^3 year^{-1}
Q_{ES}	Environmental or residual flow	m^3 year^{-1}
Q_{OUT}	Flow to catchment outlet	m^3 year^{-1}
Q_{ET}	Evapotranspiration	m^3 year^{-1}
θ	Soil moisture parameter	Dimensionless
A_T	Total catchment area	m^2
A_C	Area of catchment land cleared for agriculture	m^2
A_D	Degraded land factor	Fraction
A_N	Fraction of landscape covered by deep-rooted natural vegetation	Fraction
R_E	Total quantity of water extracted from the catchment	m^3 year^{-1}
R_{SW}	Water extracted from surface flow	m^3 year^{-1}
R_{GW}	Water extracted from groundwater store	m^3 year^{-1}
R_Q	Water extracted from reservoir storage	m^3 year^{-1}
U_c	Water usage directed to agricultural activities (i.e. irrigation and livestock)	m^3 year^{-1}
U_p	Water usage directed to household and non-agriculture related application within the catchment	m^3 year^{-1}
P_n	Population size	Number
b	Annual birth rate	Number yr^{-1}
m	Annual mortality rate	Number yr^{-1}
μ	Annual net migration rate	Number yr^{-1}
E_c	Total economic gain for catchment	Dollars
E_{pc}	Economic gain per head of catchment population	Dollars per person
p_c	Global commodity price	Dollars per tonne
B_c	Crop biomass	Tonnes per m^2
τ_A	Economic multiplier of agriculture	Dimensionless
c_A	Non-water related costs of the relevant agricultural crop or enterprise (e.g. fertiliser, machinery, livestock feed, labour etc.)	Dollars per m^2
p_{wc}	Price of water supplied for irrigation purposes	Dollars per m^3
p_{wp}	Price of water supplied for household and other uses	Dollars per m^3
E_{ext}	Catchment income generated from non-agricultural sources	Dollars
L_{ES}	Lumped indicator for the state of lifestyle-related Ecosystem Services within catchment	Dimensionless
W_Q	Lumped water quality indicator (including P, N, salt loads etc.)	Dimensionless
V	Collective community sensitivity	Dimensionless
V_{max}	Maximum value on the community sensitivity scale	Dimensionless
α	Scalar for climate regime within which catchment operates $(0 < \alpha < 1)$	Dimensionless
β	Scalar for socioeconomic development regime within which catchment operates $(0 < \beta < 1)$	Dimensionless
φ	Scalar for political regime within which catchment operates $(0 < \varphi < 1)$	Dimensionless
δ	Proportion of agriculture production as a percentage of national GDP	Dimensionless
ω	Fraction of defectors that depart from the collective action of the community (P_D/P_n)	Dimensionless
χ	Impetus for behavioural response	Dimensionless
D_E	Demand for economic expansion	Dimensionless
Z_C	Structural variables driving expansion demand, including agricultural export share, agricultural value added and crop yield	Dimensionless
ε	Development factor reflecting level of development relative to a subsistence economy	Dimensionless
ζ	Efficiency metric reflecting annual improvement in utilisation of land and water resources	Percentage
σ	Co-operativity function seeking to capture social cohesion and cooperation within catchment community (calculated with respect to ω and/or the percentage of catchment population with memberships in social organisations)	Dimensionless

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