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Discussions

A prototype framework for models of socio-hydrology: identification of key feedback loops with application to two Australian case-studies

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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 a model of socio-hydrology that posits a novel construct, a composite Community Sensitivity state variable, as a key link to elucidate the drivers of behavioural response in a hydrological context. The framework provides for both macro-scale contextual parameters, which allow it to be applied across climate, socioeconomic and political gradients, 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 different socio-hydrological case studies, taken from the Australian experience, are presented and discussed. 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 participate in the growing field of social-ecological systems modelling.

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

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century" (Annin, 2006). However, as Gleick (1993) highlighted, this phrase omits the critical point 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 fresh-water 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 ~~are~~ 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; Falkenmark, 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 ~~are able to~~ 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).

Integrated Water Resources Management (IWRM) has historically been the framework within which the interactions between human development and water resources

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have been explored, however it has been the subject of considerable debate in the literature over the past decade. It is defined as "a process which promotes the co-ordinated development and management of water, land and related resources, in order to maximise the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems" (GWP, 2000). It is essentially concerned with management actions and policy development with respect to the optimal allocation of water resources on a triple-bottom line basis. Proponents argue that the overarching philosophy and underlying principles of IWRM are integral in achieving a sustainable solution to the equitable allocation of an increasingly scarce water resource (Van der Zaag, 2005; Mostert, 2006; Leendertse et al., 2008). On the other hand, critics argue that the paradigm has yet to make the leap from theory to practice (Merrey et al., 2005; Jonker, 2007; Biswas, 2008; Saravanan et al., 2009), not least because there remains significant disparity as to what elements should be integrated under the banner of IWRM, with a review of the published literature yielding 41 different interpretations of those elements (Biswas, 2004a). The overriding impediment to the successful implementation of IWRM appears to be an inability to converge upon an operational definition and relevant measurement criteria and success metrics that adequately reflect and assess the functions of an integrated system. Despite its operational deficiencies to date, IWRM remains a key focus model for water resources management. A recent attempt at putting forward a framework for integrated models within this context concluded that "integrated modelling on coupled human-environment systems for decision support is still a poorly developed area... because it has yet to provide adequate representations of the dynamics concerning the complex interactions between human and environmental components" (Liu et al., 2008).

Accordingly, notwithstanding that the dynamic interconnection of human and natural systems has long been documented (e.g. Marsh, 1864; Thomas, 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). As a result,

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interdisciplinary research efforts have recently 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 interactions within the continually evolving coupled system, in terms of the feedbacks, non-linearities, thresholds, transformations and time lags. Indeed, notwithstanding the challenges of interdisciplinary efforts (Roy et al., 2013), a number of CHANS research initiatives have been launched internationally that seek to address such questions (set out in Liu et al., 2007a). With respect to SES research, which seeks to examine complex dynamics resulting from interactions and feedbacks between agents, resources and institutions on multiple levels (Berkes and Folke, 1998), the need for a prescriptive conceptual framework has been highlighted (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).

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; Montanari et al., 2013). Indeed, recent innovative socio-hydrology studies have proposed conceptualised models focusing on human-flood interactions (Di Baldassarre et al., 2013), urban water security (Srinivasan, 2013), and downstream use of glacier runoff (Carey et al., 2013). Socio-hydrology effectively tackles the holistic integration of the socioeconomic and environmental facets of hydrology from a different angle to IWRM – with the former focused on the exploration of fundamental scientific principles of interactions, feedbacks and co-evolution of human behaviour with the hydrological

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system, whilst the latter has tended to focus on policy-driven water management solutions imposed on the hydrological cycle. ~~The development of a~~ robust internationally-applicable theoretical framework is necessary, that has the capacity to guide the formulation of localised socio-hydrology models 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; Kandasmy et al., 2013), thus making it an ideal focus for the study of socio-hydrology.

This paper therefore, seeks to examine the coupled dynamics of an integrated agricultural human-hydrology catchment by proposing a conceptual framework that captures the workings of such agriculturally dominated 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

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catchment-specific aspects, which can be captured through locally tailored "closure functions". The paper concludes by demonstrating how such a framework would be applied to two site-specific Australian case studies, with a discussion on how the model parameters and closure functions can be characterised 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, non-linear interactions and 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).

In keeping with the triple bottom line approach taken in IWRM, social, economic and environmental components are all represented in an examination of the socio-hydrological system. Marginal changes in each of these components may be driven by exogenous drivers (e.g. climate, markets, politics) or internally by hydrological signatures within the catchment. 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. Although the logic of co-evolution amongst the two sub-systems is accepted wisdom in both the hydrology and social sciences (Boschma and Lambooy, 1999; Falkenmark, 2003; Martin and Sunley, 2007), and along the lines of the approach adopted in Di Baldassarre et al. (2013) and Srinivasan (2013), 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.

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As outlined above, the impacts of human forcings on the hydrological system, in terms of water balance, flows and quality, are presently well understood and modelled. However, the *drivers* of human forcings 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.

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. We know from general systems theory that complex systems, such as that described here, display highly non-linear 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. 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; readers are also referred to Schlüter et al. (2012) for a review of modelling considerations relevant to social-ecological systems.

2.1 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) population

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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 less tangible 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 act (Falkenmark, 1997; Folke et al., 2010) and for the population to drive the system towards a different state-space location that may be more or less sustainable. Likewise, the lower the sensitivity, the less likelihood that a change in hydrological variables will lead to meaningful action, and negative feedbacks and stable attractors within the system will promote stability. 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; Vaske and Donnelly, 1999; Armitage and Christian, 2003; Vanclay, 2004; 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" and they are described in more detail in subsequent sections.

It is important to note, however, that 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 overexploitation 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

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size, communication, reputation, time horizons) and the broader context (Anderies et al., 2011, 2013; Tavoni et al., 2012). 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-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.

In a socio-hydrology context, it is necessary to articulate community sensitivity in terms of its drivers in order to provide a logical link in a coupled context. Where does this sensitivity originate? 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

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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).

This paper thus puts forward community sensitivity as the composite driving variable. 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). 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 an emotive 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

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change, as opposed to reporting an absolute value, thus shifting the focus to the direction and relative magnitude of any movement.

2.2 Converging approaches in the literature

In defining the sensitivity variable, the framework draws on a number of concepts from vulnerability, resilience and sustainability sciences. The accepted definition of vulnerability is the degree to which a system is exposed to harm as a result of stressors or change. It is viewed as a dynamic characteristic, comprised of two parts: the degree of sensitivity and the adaptive or coping capacity of a system (Turner et al., 2003; Srinivasan et al., 2013). Dynamic vulnerability has been defined as "the extent to which environmental and economic changes influence the capacity of regions, sectors, ecosystems, and social groups to respond to various types of natural and socio-economic shocks" (Leichenko and O'Brien, 2002). There has been a great deal of emphasis in the literature on the vulnerability of communities to environmental hazards or perturbations, linking vulnerability directly to the sensitivity and coping capacity of the coupled human-environment system (Cutter, 2003; Adger, 2006; Eakin and Luers, 2006; Polsky et al., 2007). Despite extensive research on this matter, to date, lessons pertaining to the coupled human-environment system have tended to be predominantly based on selective qualitative case studies (Cutter et al., 2000; Turner et al., 2003; Wisner et al., 2005) with very few exceptions (Cutter and Finch, 2008), such that generalised principles and aggregate measures of vulnerability have remained elusive due to the complexity of the coupled system (Turner, 2010).

Another plausible way of interpreting vulnerability is in relation to a community's resilience, whereby vulnerability can be viewed as the antonym of resilience (Folke et al., 2002). 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, 2006; Forbes et al., 2009; Amundsen, 2012). In fact, Anderies et al. (2004) specifically highlight that resilience is useful when applied as

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a framework that enhances our ability to understand the workings of complex systems behaviour, rather than being a coherent body of theory in its own right. The collective resilience of a community has been defined as "the ability of a community to cope and adjust to stresses caused by social, political and environmental change and to engage community resources to overcome adversity and take advantage of opportunities in response to change" (Amundsen, 2012). This concept of community resilience, as applied to socio-ecological systems, is an emerging field within the resilience literature (Leichenko and O'Brien, 2008; Buikstra et al., 2010; Ross et al., 2010; Anderies and Janssen, 2011; Amundsen, 2012). As noted in Turner (2010, p. 573), "at their most fundamental level, vulnerability and resilience applied to the coupled human-environment system constitute different but complimentary framings". The former concept assesses the weakness of a system, while the latter focuses on the strength of the system. For our purposes, both concepts are associated with the sensitivity of the system, at opposite ends of the scale.

Yet another angle of examination is to employ the concept of sustainability, whereby the perceived sustainability of a community's quality of life is inversely related to its sensitivity. Numerous associations have been made between resilience and sustainability in the literature (Folke et al., 2004; Walker and Salt, 2006; Mäler, 2008; Derissen et al., 2011). Furthermore, there is support for 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. This 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; 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 below). It is worth noting that, despite such connections, these concepts are

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not interchangeable, in that the presence of one does not automatically equate with nor necessitate the presence of the other (Derissen et al., 2011).

From a model standpoint, the overall objective here 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. To this end, it is not the intention of this paper to fully quantify the resilience, vulnerability or sustainability of the system per se, merely to draw upon the concepts to develop a novel construct; a factor that enables an understanding of the coevolving components of the system. This focus on human perceptions in relation to natural resources is not new and has long been employed in ecological economics couched in terms of an individual's *willingness to pay* to resolve water issues for example (Poe and Bishop, 1999; Gregory and Wellman, 2001; Tanellari et al., 2009). However, to date there is little research regarding a community's *willingness to respond* to environmental changes (Gooch and Rigano, 2010), which is the crux of what our model seeks to examine. 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). 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.

2.3 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". The former is a reflection of observable quantifiable factors. 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.

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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 to the population effect, a rise in demand is also expected on account of increasing economic prosperity (even with a stable population). This is a development cycle characteristic, as more industrialised economies have increasingly sophisticated water needs. We would therefore expect to see increased water usage as communities move along the development scale, even as population size remains stationary.

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 water variables impede further growth. Water use efficiency measures would feed into the cycle to extend the life or economic productivity of these limiting variables. However, once all efficiency measures have been put in place, to the extent that water flows reduce, water quality deteriorates or land degrades, economic growth will naturally be constrained. As previously highlighted in the case of common pool resources, the resource that underpins development, in this case the freshwater resource, is often prone to overexploitation, which can ultimately lead to a deterioration in local social and economic conditions.

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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 possibly revegetating land. This is the first feedback loop that merits investigation.

The second loop centres around the Sensitivity state variable, which by its nature is more intangible and emotive, and therefore not as readily observable in empirical data. 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 that community's 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. However, as noted earlier, there are numerous preceding conditions that determine the extent to which such collective behaviour will emerge and spread (Anderies et al., 2011, 2013; Tavoni et al., 2012). Nonetheless, the hypothesis ties in broadly with the documented trajectory of development and management efforts in the Murrumbidgee basin (Kandasamy et al., 2013), the California Delta (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; Gober and Wheater, 2013; Savenije et al., 2013).

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Although community sensitivity generally makes relative shifts on a gradual basis, it is expected that baseline community sensitivity levels (i.e. the initial condition from which relativity is observed) will differ depending on that catchment's climate, water abundance, socioeconomic development stage and political regime. It is therefore essential to observe the Sensitivity state variable along each of these gradients.

Empirical evidence of the phenomena outlined above has indeed been documented by Kandasamy et al. (2013) in the Murrumbidgee River Basin in Australia. The evidence was interpreted 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. To illustrate how the Sensitivity state variable is expected to vary over time, its hypothetical trajectory for the Murrumbidgee River Basin is shown in Fig. 2.

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 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.

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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 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.

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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 allows 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:

$$\frac{dP_n}{dt} = (b - m + \mu)P_n \quad (1)$$

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 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.

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3.3 Economic function

Within the model framework the economics of the catchment, captured in its simplest form, can be made up of two key components, namely a land productivity component and an agricultural cost and water supply component. 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 two 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

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from agricultural productivity growth. Thus an economic function of the form:

$$E_c = \left[p_c A_C (1 - A_D) B_c\right] \tau_A - \left[(c_A A_C (1 - A_D)) + p_w \left(U_c + U_p\right)\right] \pm E_{ext} \quad (2a)$$

$$E_{pc} = \frac{E_c}{P_n} \quad (2b)$$

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_w is the price of water, 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.

It is important to note that such metrics are felt to sufficiently capture the economics of a predominantly agricultural catchment. To the extent the catchment in question has a strong fishing industry, manufacturing industry or hydropower plants, additional metrics would need to be considered for incorporation, such as profit per kg fished, profit per units produced or profit per kWh, along with their associated multipliers. E_{ext} is an optional variable added above to account for income generated within the catchment from sources independent of agriculture, and could be set to zero in the simplest case. To the extent that a more detailed catchment-specific economic model is available,

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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 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:

$$L_{ES} = f(W_Q) + f(Q_{ES}) + f(S_W) + f(A_N) \quad (3)$$

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

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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 (similar to Imberger et al., 2007). We acknowledge 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

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), making it difficult to quantify for the purposes of modelling. Further, we are precluded from doing this in the present context as we are attempting to capture phenomena and feedbacks over long periods of a century or more. Given it is only possible to canvass perceptions in this manner at a given point in time, it is necessary to employ proxies that can be objectively measured over historical periods. Further, it is our intention that this variable capture differences across climate, economic and political gradients in order that it can be used in a general way.

The Sensitivity function proposed here is comprised of six elements, three of which are national or regional in scale, and three of which pertain more specifically to the local catchment community. This approach of using both local dynamic variables, supplemented by the regional and global context in the examination of coupled human-nature systems, is supported in the literature (Liu et al., 2007a). The first macro-scale contextual parameter we introduce reflects the underlying climate regime experienced by the catchment, α, with drier catchments expected to display higher baseline 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

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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/PQ_{ET}). It is noteworthy that there is a marked observable difference between the climate regimes of developed vs. 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, 2004b).

The second macro-scale contextual parameter, β, reflects the influence of the socioeconomic regime of the catchment on perceived sensitivity levels. As catchments move along the scale from rural to transitional to industrialised, it is expected that 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). 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 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").

The third macro-scale contextual parameter to be captured is the 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

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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 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, these three macro-scale parameters can be set to define the catchment context and will constitute controls that serve to either amplify or dampen the feedback loops highlighted in the previous section.

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, $S_x = f(S_Q, S_{GW}, S_{US})$, 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

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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 rain-fed 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 multi-decadal 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,

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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:

$$\frac{dV}{dt} = \left[\left(\underbrace{-\widetilde{S_x}\gamma_s}_{\substack{\text{water}\\\text{availability}}} \underbrace{-\widetilde{L_{ES}}\gamma_{es}}_{\substack{\text{ecosystem}\\\text{services}}} \underbrace{-\widetilde{E_{pc}}\gamma_e(1+\delta)}_{\substack{\text{economic}\\\text{return}}} \right) \underbrace{(1-\alpha)}_{\substack{\text{climate}\\\text{context}}} \cdot \underbrace{(1-\beta)}_{\substack{\text{development}\\\text{context}}} \cdot \underbrace{(1-\varphi)}_{\substack{\text{political}\\\text{context}}} \right] V \qquad (4)$$

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 ($0 < \varphi < 1$), $\widetilde{L_{ES}} = \Delta L_{ES}/\overline{L_{ES}}$ is the relative change in ecosystem services of the catchment, $\widetilde{E_{pc}} = \Delta E_{pc}/\overline{E_{pc}}$ 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 $\widetilde{S_x}$, $\widetilde{L_{ES}}$ and $\widetilde{E_{pc}}$ 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 timesteps 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 calibratible 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

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a supplementary approach to account for the degree of reliance on agriculture in terms of local livelihoods.

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 non-linear dynamics. Thus, the defector fraction, $\omega = P_d/P_n$, 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 (refer to 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 anthropo-centric measures. In the simplest sense this can be composed as (Fig. 5):

$$\chi = f(V^*) - f(D_E) \quad (5)$$

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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 cooperativity within the community. Furthermore, it is important to note that whether and to what extent a community responds, are generally influenced by two variables: the magnitude of influence (derived from V) and the *capacity* of the community to respond (Chaskin,

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2008), modelled here as:

$$f(V^*) = \begin{cases} 0 & \text{for } V^* \le V^*_{\text{crit}} \\ \chi_{\text{maxV}} \left(\frac{(V^*)^\sigma}{(k_V^\sigma + (V^*)^\sigma)} \right) f(\varepsilon) & \text{for } V^* > V^*_{\text{crit}} \end{cases} \quad (6a)$$

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

$$V^* = \frac{\Delta V}{V_{\text{max}} - V_t} \quad (6b)$$

where V_{max} is an arbitrary constant reflecting the maximum sensitivity of the particular community, and the term $V_{\text{max}} - V_t$ scales the incremental change in sensitivity to increase χ as the baseline sensitivity approaches the maximum. In Eq. (6a), σ is a cooperativity 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 $\varepsilon = \frac{E_{\text{cN}}}{E_{\text{cS}}}$ 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

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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:

$$f(D_E) = \chi_{maxD}\left(\frac{D_E}{(k_D + D_E)}\right) \tag{7a}$$

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

$$D_E = \left[\frac{\Delta P_n}{P_n^t} + f(Z_C)\right]\left(1 - \frac{A_C}{A_T}\right)\left(1 - \frac{R_E}{S_T}\right)(1 + \zeta) \tag{7b}$$

where $\frac{\Delta P_n}{P_n^t}$ is the population growth rate (similar to Barbier's (2004) rural population growth rate), $f(Z_C)$ is a function of structural driving variables that could comprise agricultural export share, growth in agricultural value added, and 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_C/A_T) and water (R_E/S_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, $A_C/A_T \rightarrow 1$; or water limited, $R_E/S_T \rightarrow 1$). 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.

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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 ΔS_{max} is the annual change in storage capacity. Each of the management response equations would then take the form, for example, of:

$$\frac{dR_E}{dt} = \eta_{RE} f_{RE}(\chi), \tag{8a}$$

$$\frac{dA_C}{dt} = \eta_{AC} f_{AC}(\chi), \tag{8b}$$

$$\frac{dS_{max}}{dt} = \eta_{Smax} f_{Smax}(\chi), \tag{8c}$$

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. For an example of how each of these functions might evolve over time in a catchment such as the Murrumbidgee, the reader is referred to Fig. 4a for a depiction of storage evolution (i.e. S_{max} in this model), Fig. 4b which provides an illustration of the evolution of irrigation areas (i.e. A_C) and Fig. 4e which shows irrigation flow utilisation (i.e. U_C, which is a component of R_E) in Kandasamy et al. (2013).

4 The conceptual framework in practice

To demonstrate how the generic conceptual framework can be applied to analysing the evolution of different catchments, two distinct 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. 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

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these two catchments are to be captured through application of the conceptual framework and how parameterisation of the closure relationships is being pursued. 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.

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 N.S.W., has been described in detail by Kandasamy et al. (2013). 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., 2013). 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., 2013). Marked reductions were recorded in water bird and native fish populations in the Murrumbidgee basin as a result.

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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 river 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., 2013).

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 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).

4.2 Lake Toolibin catchment

The Toolibin catchment covers a much smaller area of some 48 000 ha, 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

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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 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 was 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, townsites), 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,

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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).

4.3 A final word on limitations

As can be seen from the distinctness of 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.

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. Such representation is intended to provide a stepping stone from which to begin to observe, analyse and compare real

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world coupled dynamics. 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. It is also important to note that the conceptual model presented in this paper seeks to represent general principles, to which there will undoubtedly be numerous examples that may stress the inherent assumptions. As an example, in the case of the Aral Sea the time lag between the appearance of negative ecosystem and economic impacts did not allow sufficient time for an effective response pattern that was capable of reversing such consequences. It is also possible that this basin operated within a threshold sensitivity band that was at or below critical response level, thereby precluding a significant change in the χ function.

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.

5 Conclusions

The sustainable management of global freshwater resources has never been a more pertinent or pressing issue for humanity, with the spotlight in recent years increasingly being placed upon the importance of understanding the complex dynamics of coupled human-hydrology systems. It is in this spirit that new effort is being placed by hydrologists in the field termed "socio-hydrology". This paper has sought to identify

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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 functions. 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.

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

Model component	Variable/ Function	Murrumbidgee Catchment	Toolibin Catchment
Water balance	S_{GW}, S_{US}, S_Q, S_W, Q_S, Q_{SS}, Q_{ES}, Q_{Rin}, Q_{Rout}	Simple water balance as in Farmer et al. (2003) and Fig. 4. S_Q is relatively large in this case as the catchment relies on irrigation water for farming, and catchment flows became increasingly diverted to storages over the past 100 yr. In the recent severe drought this led to the over allocation of water and consequent decline in wetland storages and provision of environmental flow.	This is a semi-arid catchment with a low runoff coefficient, a deep saline groundwater aquifer and ephemeral wetland system. A simple water balance as in Fig. 4 is appropriate and must account for recharge and surface runoff relative to the extent of land clearing. Note in this catchment 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. For the past two decades the wetland system has been managed by diverting saline water flows around the wetland system, however environmental flows are not used.
Population P_n	b, m, μ	Dynamic population models are available for the Murrumbidgee basin, based on birth rate, mortality rates, and "push" and "pull" factors associated with economic activity and environmental appeal.	Most population growth driven by government policies incentivising agricultural expansion. Population in this case can be imposed as a boundary condition using Australian Bureau of Statistics census data.
Economy E_c	p_c, B_c, p_w, A_C, A_D	p_c driven by global price of rice as the main crop. Irrigation farming: economic productivity is a function of area of cleared land (A_C), area of degraded land (A_D), the link between crop yield (B_c) and irrigation water applied. The cost of irrigation water (p_w), as well as technological advancements affecting agricultural productivity (e.g. mechanisation/pesticides) vary considerably over the past 100 yr.	p_c driven by global prices of wheat and wool. Dryland farming: economic productivity is a function of area of cleared land (A_C), area of degraded land (A_D) and given it is a rain-fed catchment the biomass (B_c) will evolve in line with soil moisture in the unsaturated zone. Technological advancements affecting agricultural productivity (e.g. mechanisation/pesticides) are also relevant over the past several decades.

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Table 1. Continued.

Model component	Variable/ Function	Murrumbidgee Catchment	Toolibin Catchment
Ecosystem Services L_{ES}	$f(W_Q)$ $f(Q_{ES})$ $f(S_W)$ $f(A_N)$	Loss of biodiversity and salinisation of land due to clearing of natural vegetation. Degradation of flora and fauna in river/seasonal wetlands due to the diversion of natural flows and subsequent algal blooms caused by nutrient runoff. Salt wedge intrusion due to diversion of natural flows. Water quality (W_Q) variables of relevance include: (i) nutrient runoff (P,N); and (ii) salt load. Water quantity measures to include: (iii) river flow (Q_{ES}) and (iv) wetland height (S_W); both derived from the water balance model. Land variables to include: (v) degraded/salinised land (A_D) and (vi) natural deep-rooted vegetation (A_N).	Land salinisation and secondary salinity in the lake (from saline surface water inflows and rising groundwater tables) is a major driver of ecosystem degradation. Lake Toolibin is a Ramsar listed wetland of international importance due to diversity and abundance of water birds that breed on lake. Flora and fauna at risk in and around lake. Remnant vegetation at risk due to land salinisation. Water quality (W_Q) variable of relevance: (i) salt load. Water quantity measure: (ii) lake storage (S_W). Land variables to include: (iii) degraded/salinised land (A_D) and (iv) natural deep-rooted vegetation (A_N).
Sensitivity $\frac{dV}{dt}$	$\widetilde{S_x}$	Community sensitivity in an irrigated catchment is predominantly a function of the amount of water available in storage (i.e. dams and weirs) – $f(S_Q)$. Thus periods of prolonged drought manifest in sensitivity changing in relation to dam storage levels.	Community sensitivity in a rain-fed 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 can also be considered since small farm dams service livestock and hence sheep farmers will be sensitive to water availability in local dams.
	$\widetilde{L_{ES}}$	As highlighted above, community sensitivity will be driven by the overall change in the main ecosystem services; see also specific drivers of change in Fig. 2.	As highlighted above, community sensitivity will be driven by the overall change in the main ecosystem services.
	$\widetilde{E_{pc}}$	Community sensitivity is known to respond to economic wellbeing per capita, which faced increasing pressure during the drought years when a decline in water availability drove up production costs.	Community sensitivity is known to respond to economic wellbeing per capita, which is closely linked to global commodity prices of wheat and wool and increasing costs of production.

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Table 1. Continued.

Model component	Variable/ Function	Murrumbidgee Catchment	Toolibin Catchment
Behavioural Response	$f(D_E)$	Driven by household water needs of resident population (P_n) and demand for agricultural land area (Z_C). Moderated by implementation of water and land usage efficiency measures (ζ) that have occurred over the past several decades.	Resident population water needs are supplied through Scheme Water, so the main driver is demand for agricultural land, Z_C, for which agricultural export share can be used as a proxy. Moderated by land usage efficiency measures (ζ).
χ Driver Functions	$f(V^*)$	Change in community sensitivity over time will be minor until the 1990's when environmental degradation motivated changes to water allocation. Community cohesion and cooperativity factor (σ) will be based on an agent based model using an "ostracism" factor since significant community dissension occurred following release of the Murray-Darling Basin Sustainable Diversion Limits. Societal development factor, ε, would be high based on high national GDP.	Change in community sensitivity over time is expected to exhibit an increase from the 1990s onwards as environmental damage became widespread and began to threaten agricultural productivity. Given the smaller size of the catchment, community cohesion and cooperativity factor (σ) may be based on the proportion of the population with memberships in social organisations. Societal development factor, ε, would be high based on high national GDP.
χ Response Functions	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 (refer to changes as outlined in Kandasamy et al., 2013).	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.

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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	m^3
S_W	Wetland storage	m^3
h_{WT}	Water table height	m
Q_S	Surface runoff	$m^3\,yr^{-1}$
Q_{SS}	Subsurface flow	$m^3\,yr^{-1}$
Q_{Rin}	Flow diverted to reservoirs	$m^3\,yr^{-1}$
Q_{Rout}	Flow released from reservoirs	$m^3\,yr^{-1}$
Q_{ES}	Environmental or residual flow	$m^3\,yr^{-1}$
Q_{OUT}	Flow to catchment outlet	$m^3\,yr^{-1}$
Q_{ET}	Evapotranspiration	$m^3\,yr^{-1}$
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	$m^3\,yr^{-1}$
R_{SW}	Water extracted from surface flow	$m^3\,yr^{-1}$
R_{GW}	Water extracted from groundwater store	$m^3\,yr^{-1}$
R_Q	Water extracted from reservoir storage	$m^3\,yr^{-1}$
U_c	Water usage directed to agricultural activities (i.e. irrigation and livestock)	$m^3\,yr^{-1}$
U_p	Water usage directed to household and non-agriculture related application within the catchment	$m^3\,yr^{-1}$
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

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Table A1. Continued.

Variable	Explanation	Units
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_w	Price of water	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	Cooperativity 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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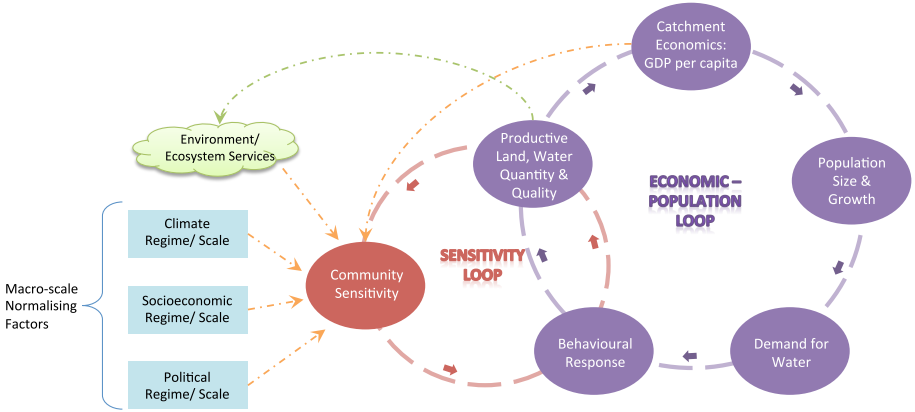

Fig. 1. The socio-hydrology model as two interconnecting feedback loops.

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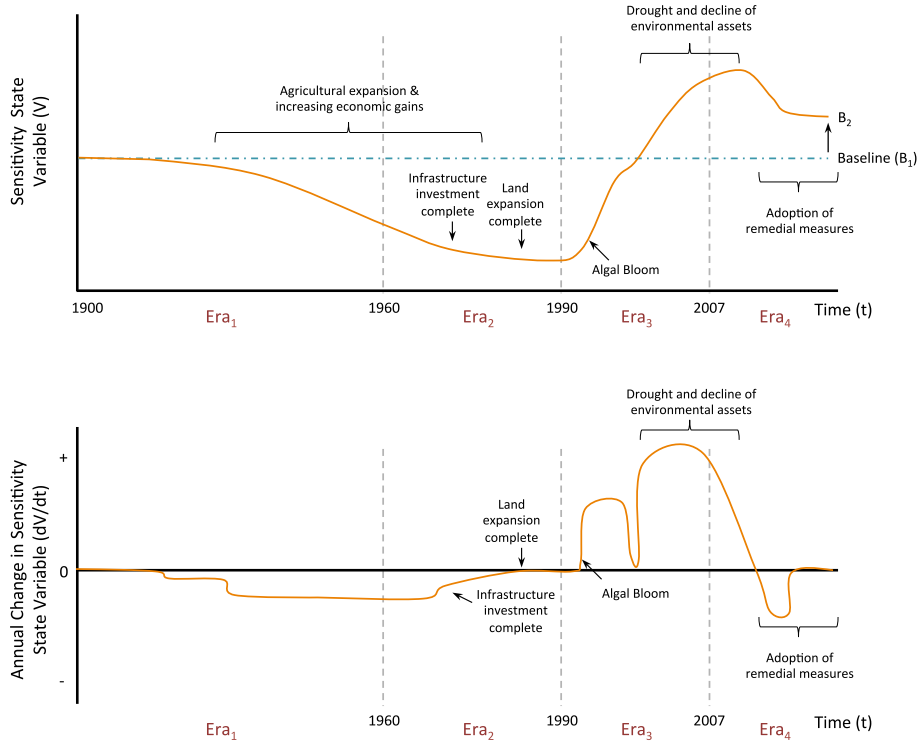

Fig. 2. An idealised sketch showing a hypothetical trajectory for the Sensitivity state variable in the case of an example catchment, such as the Murrumbidgee River Basin in Australia. The illustration adheres to the four eras represented in Kandasamy et al. (2013, Fig. 3). The reader is referred to Sect. 4.1 and to Kandasamy et al. (2013) for a more detailed discussion of the Murrumbidgee River Basin.

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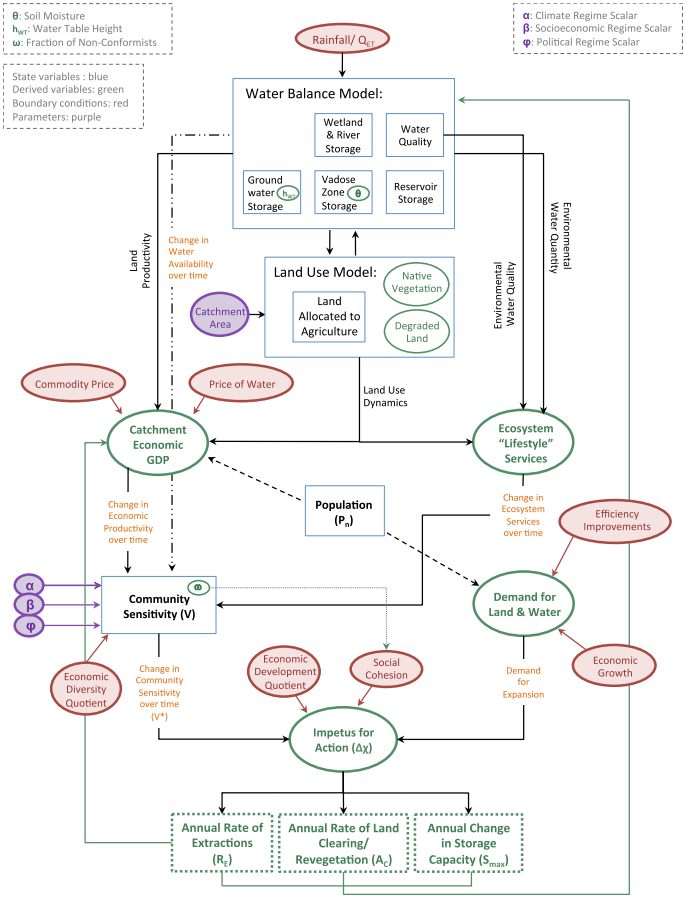

Fig. 3. A generic socio-hydrology conceptual framework for application to agricultural catchments.

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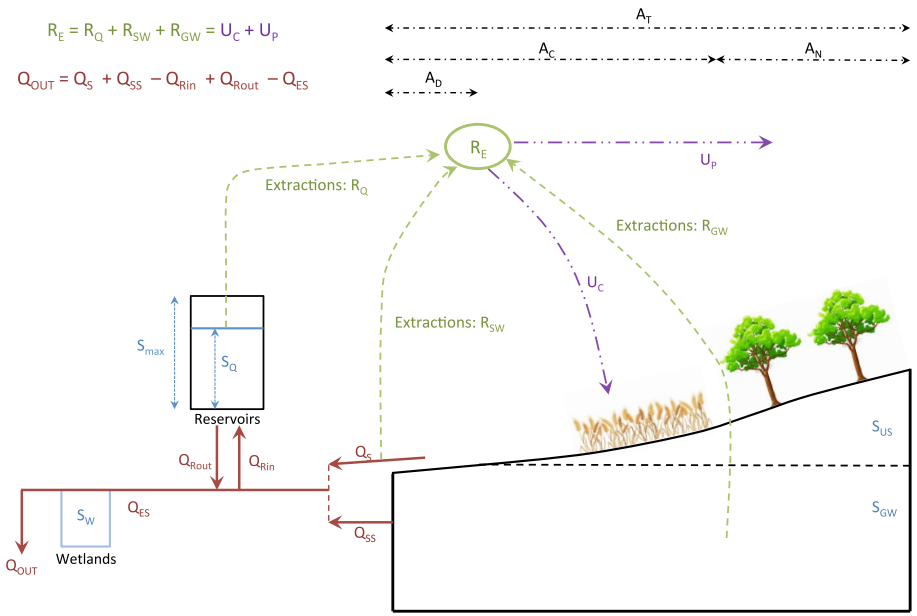

Fig. 4. A simple catchment water balance model that includes the minimum necessary components for application of the conceptual framework.

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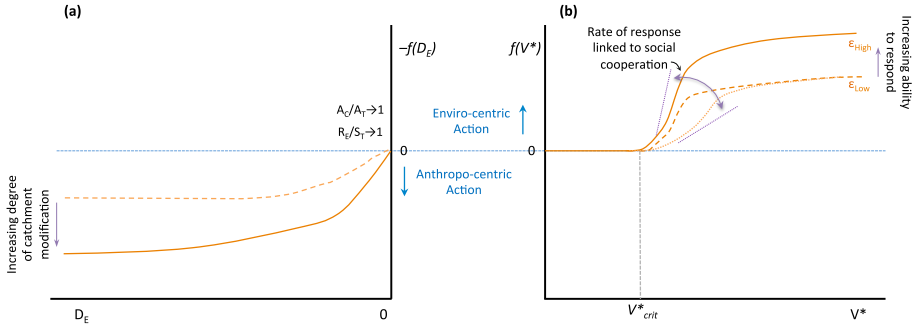

Fig. 5. A hypothetical illustration of how the Behavioural Response functions vary according to: **(a)** the change in catchment demand for expansion, D_E, and; **(b)** the change in collective community sensitivity, V^*. The functions can be customised according to factors such as community cooperation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

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