

Chapter 5

Current and Future Perspectives on Forest-Water Goods and Services

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5.1 Introduction

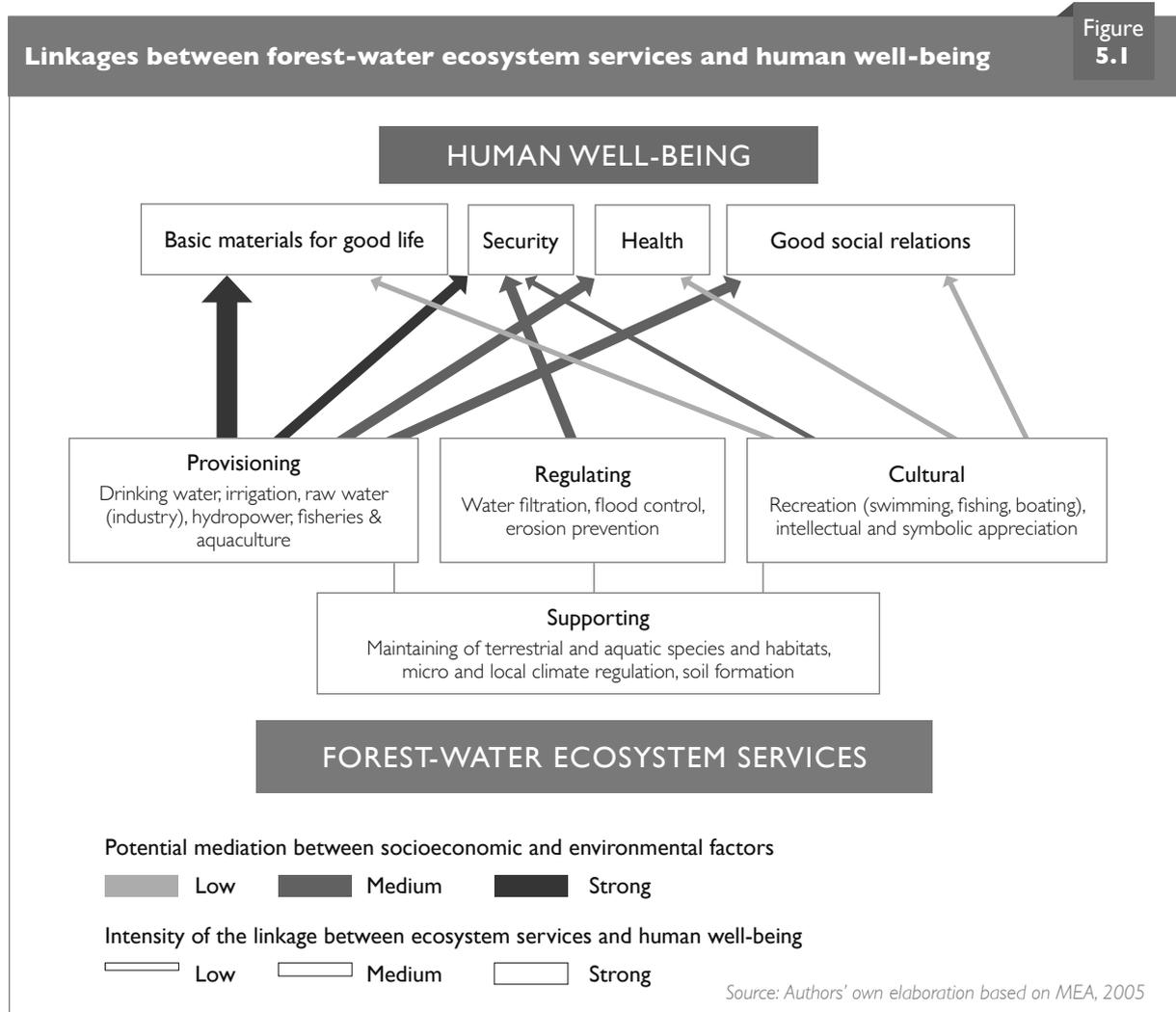
This chapter utilises the ecosystem services framework to understand the consequences of change in forest ecosystem functions and water-related implications. Using a scenario analysis, the chapter explores the likely changes in attributes of forest-water systems (and associated services) that will translate to exogenous impacts, and their consequences in the future. The narrative provides a foundation for the analysis of management options and policy responses that will be discussed in Chapters 6 and 7. Ultimately, these responses are likely to affect the drivers of change and thus highlight the interconnectedness of coupled forest-water systems.

5.2 Conceptualising Forest-Water Relationships in Terms of Ecosystem Services

5.2.1 Origins and Evolution

The dependence of human life and well-being on finite natural resources has long been acknowledged (Malthus, 1888; Meadows et al., 1972), and different

conceptualisations of human-nature relationships have emerged over time (Raymond et al., 2013). The term ecosystem services (ES) represents one such conceptualisation (Martin-Ortega et al., 2015). The ES concept was coined in the 1960s primarily to raise awareness among policymakers about the implications of biodiversity loss and environmental degradation by emphasising the benefits that nature freely provides to society (Gómez-Baggethun et al., 2010). The “Tragedy of the Commons” framed by Hardin (1968) triggered the debate about open access to natural resources. The natural processes of environmental degradation which have impacts on social-ecological systems, therefore, generate social change (Eckholm, 1975). Literature on ecosystem services grew exponentially from 1997 onwards, when Daily (1997) defined the term as “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life” and Costanza et al. (1997) estimated the total economic value of the planet’s ecosystem services at USD 33 trillion/year. Despite criticisms on methodological grounds (e.g., El Serafy, 1998), further publications consolidated this body of research (e.g., De Groot et al., 2002), until it firmly entered the policy arena when the UN Secretary-General Kofi Annan called for a global assessment of the world’s ecosystem services



(Millennium Ecosystem Assessment report, MEA, 2003, 2005). Ecosystem services were then defined as “the benefits that people obtain from ecosystems” (MEA, 2003) and the dominant classification scheme of ecosystem services was established. In this scheme, ES were divided as supporting (services required for the production of other ecosystem services), provisioning (products that can be directly obtained from the ecosystem), regulating (benefits that can be indirectly obtained from the regulation of ecosystem processes), or cultural services (non-material benefits that people obtain from ecosystems), which all directly or indirectly contribute to human well-being.

Further to the MEA, there has been a proliferation of ES frameworks and applications, including a multitude of novel research directions and refined definitions and classification of the ES domain (Ojea et al., 2012). A major difference between the ES frameworks is how intermediate ecosystem processes are treated. Some frameworks only include final services consumed or valued directly by humans (e.g., Hein et al., 2006; Haines-Young and Potschin, 2013), while others also include intermediate environmental processes that contribute indirectly to human-welfare (e.g., Boyd and Banzhaf, 2007). As ecosystems depend strongly on the water cycle, the complex inter-linkages between ecosystems and the water cycle make the classification of water-related services as supporting, regulating, or provisioning particularly complex (Ojea et al., 2012). For example, water flows can be regarded as supporting services for maintaining terrestrial and aquatic species and habitats, or micro and local climate regulation; or they can be regarded as regulating services for aquaculture production or as provisioning services for agriculture or drinking water supply; in a way that simultaneously affects different components of human well-being (Figure 5.1). Box 5.1 illustrates how an ecosystem services-based approach would apply to the understanding of water-related forest ecosystem services.

5.2.2 Valuing Ecosystem Services

Values and associated processes of valuation have been of interest to researchers and philosophers since ancient times, and the term has been ascribed a multiplicity of meanings (Schulz et al., 2017). On the one hand, values can be conceptualised as abstract guiding principles (fundamental or held values) that may inform preferences and decision-making. Examples are security, achievement, or self-direction. On the other hand, values can be understood as measurements of a certain quality or of importance (i.e. assigned values). The ecosystem service paradigm and environmental economics, which are rooted in neoclassical economics, are examples of strategies to describe assigned values. Human beings are seen as rational actors that aim to satisfy their substitutable preferences and maximise their personal utility through their choices (Pearce and Turner, 1990; Dietz et al., 2005). Value is then defined as “the change in human well-being arising from the provision of [an environmental] good or service” (Bateman et al., 2002). These welfare changes can be compared through conducting

Box
5.1

An ecosystem services-based approach to the understanding of water-related forest ecosystem services:

- ... recognises that structural changes to forests can influence several watershed processes (e.g., erosion rates, sediment load, water chemistry, peak flow levels, total flow, base flow, or groundwater recharge) – in different ways and that, in turn, these changes result in different kinds of impact on human well-being (e.g., increased costs of water purification, increased fertilisation of floodplain lands, decreased reservoir capacity due to siltation, flood damage, changes in agriculture), (Lele, 2009).
- ... requires the understanding of the biophysical processes that determine the way forest cover, forest structure, soil-vegetation dynamics, etc. affect the amount and quality of freshwater to the extent that it impacts on human well-being (through use or non-use) by the beneficiaries.
- ... combines knowledge of the service delivery processes that are based on natural sciences (e.g., plant physiology, ecology, hydrology) with information from social sciences (e.g., economics, psychology, political science) and (local) stakeholder knowledge (e.g., farmers, domestic water users, floodplain residents, hydropower companies, regulators) that jointly help to understand, for example, where benefits arise in relation to where ecosystem change takes place.
- ... requires at least some degree of quantification of changes in the final services delivered (e.g., increase of the flow of water associated with forest cover) coupled with a qualitative interpretation of the implications for human well-being, or the valuation of associated benefits through, for example, willingness to pay for increased water availability, so that these benefits can be incorporated into decision making (for example, on afforestation or the creation of protected forest areas) (Martin-Ortega et al., 2015).

monetary valuation studies that estimate relative values and people’s willingness to pay to achieve an environmental change, such as improved water quality from forest conservation practices.

While being the most widespread conceptualisation of (environmental) value, the neoclassical definition has also attracted a lot of criticism for epistemological and moral reasons (Gómez-Baggethun et al., 2010; Norgaard, 2010). Other critiques reflect misgivings about related concepts such as markets, capitalism, commodification and/or neoliberalism derived from monetisation (Brockington and Duffy, 2010) and have been rejected by those defending more eco-centric conceptualisations of human-nature relationships (Martinez-Alier, 2002; Schulz et al., 2017).

Another recent branch of literature on values focusses on shared and social values, which Kenter et al. (2015) present as those values that an individual holds on behalf of a community or group of which they are a part. More

recently, relational values, which are understood as ethical and moral principles that guide ‘good’ human-nature relationships and may differ across cultures (Chan et al., 2016), have emerged as a new conceptualisation more coherent with the pluralistic views promoted by new approaches to ecosystem services (Muradian, 2017), as advocated by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES).

Here, when we refer to the value of water services provided by forest, we refer to assigned values (i.e. the values that people derive from forest and the water services they provide) but recognise that these are underpinned by fundamental values that critically shape world views and the relationship between people and forests.

5.2.3 Criticisms and New Conceptualisations: Nature’s Contributions to Humans

The ecosystem services concept has arguably inspired novel avenues for environmental research, enhanced communication, debates and cooperation between scientists from diverse disciplines, policymakers, conservationists and practitioners. Beyond the MEA, the global TEEB initiative (The Economics of Ecosystem Services and Biodiversity; Kumar, 2010), and related national ecosystem assessments (e.g., the UK NEA, 2011) are testimony of the concept’s wide-ranging appeal.

Inevitably, the popularisation of the ES approach has also led to the emergence of new debates and criticisms. While not questioning it, some see gaps in the practical implementation of the conceptual advances made (Nahlik et al., 2012), such as deficient monitoring, and some see the risk of the concept of ecosystem services losing its original (or any) meaning as pre-existing environmental management approaches are simply relabelled. More critically, many point at the risk of oversimplification of ecological, economic and political processes (Norgaard, 2010). Ecological economists are critical of the neoclassical conceptualisation of environmental values and argue that some values cannot be measured with a single measurement unit such as money (Martinez-Alier et al., 1998). Ethical concerns have also been raised about the potential misuse of the ecosystem services concept for the commodification of nature where artificial markets are created for public environmental goods (Kosoy and Corbera, 2010; Peterson et al., 2010), as well as about the marginalisation and crowding-out of non-anthropocentric (often non-Western/utilitarian) ethical frameworks for nature conservation (Raymond et al., 2013).

The criticism is extended to the consideration of equality in the distribution of ecosystem services, and also to the interpretation of benefits in different socio-cultural contexts. The power, gender and labour relationships which mediate access and capability to manage ecosystem services need to be highlighted in an ecosystem service approach. The degree to which any individual benefits from ecosystem services thus depends on a complex range of mechanisms of access including natural and social capitals, both traditional as well as emerging and evolving rights to natural resources (Ribot and Peluso,

2003). Also, the ecosystem services approach often does not sufficiently take traditional ecological knowledge into account (Xu and Grumbine, 2014a,b). Some argue that a practical alternative to the problems of conventional valuation would be to make use of a multi-criteria approach, enabling the inclusion of a wider range of issues (Fontana et al., 2013). Others propose a less anthropocentric conceptualisation of values that encompasses other worldviews (such as those of indigenous communities). For example, in Australia, indigenous people believe that all of the environment is interlinked and they are part of that interlinkage (Altman and Branchut, 2008), having been created with forests and water, and all within them at the beginning of time, remaining as custodians of nature (Flannery, 1994; Skuthorpe and Sveiby, 2006). Even today they engage in living cultural landscapes and waterscapes, where water and forests are central to culture, spirituality and identity (Bark et al., 2011). A major challenge remains as to how such deep understandings can be incorporated into modern policy and institutional arrangements relating to the management of forests and water resources.



Uluru, also known as Ayers Rock, is a large sandstone rock formation in central Australia, Uluru-Kata Tjuta National Park. Uluru, including its rock caves, ancient paintings, and surrounding springs and waterholes, is sacred to the Pitjantjatjara Anangu, the Aboriginal people of the area

Photo © Pixabay:Wallula

To address some of these challenges and criticisms, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has recently developed a new framework that seeks to integrate different knowledge systems regarding human-nature interactions, including indigenous and local perspectives alongside western scientific models (Pascual et al., 2017). It consists of six interlinked elements constituting a social-ecological system that operates at various scales in time and space (Díaz et al., 2015). These are nature; nature’s benefits to people; anthropogenic assets; institutions and

governance systems and other indirect drivers of change; direct drivers of change; and good quality of life. While shifting the focus towards relational values, a good quality of life and cultural specificities, the IPBES framework essentially maintains the original anthropocentric perspective, but emphasises a less utilitarian philosophy and pluralistic values.

In this assessment we adopt the ecosystem services conceptualisation as the currently dominant way of expressing the relationship between humans and nature, so that any existing evidence can be integrated more effectively here. However, we acknowledge other visions and the fact that this is an evolving paradigm.

5.3 Consequences of Change

5.3.1 Consequences for the Delivery of Water-Related Forest Ecosystem Services

Forest ecosystems provide timber, energy, food, fodder and other goods while maintaining diverse ecosystem services and functions (see Section 5.2) that are relevant for human well-being.

Forests are spatially heterogeneous areas in which the trade-offs and synergies in the provision of goods and services are governed by complex interactions of environmental factors and processes, with social and economic forces operating at different spatial scales. The spatial pattern of land uses, management intensity, land use changes, climatic conditions, the resilience of forest ecosystems, and natural and anthropogenic stressors, such as droughts, extreme climatic events, wildfires, atmospheric pollution, or invasive species are the main factors affecting the provision of forest goods, and environmental services and functions (Lawler et al., 2014; Millar and Stephenson, 2015; Newbold et al., 2016; Castello and Macedo, 2016; Seidl et al., 2016). Changes in coupled forest-water systems can thus have significant impacts on biota, and ultimately on human-well-being. Yet the consequences of natural- and anthropogenic-driven changes on forest-water systems depend on their scale and intensity (see Chapter 3). Where forest and water are concerned, changes in land use and management would mainly affect water quality and quantity.

The most significant contribution forests make to water for all living beings is in maintaining its quality (FAO, 2008). The role of forests in filtering sediments and other pollutants from water before it reaches the stream has increased the interest in conserving forest and restoring riparian vegetation to protect water quality (Sweeney and Newbold, 2014). Brogna et al. (2017) found that forest cover has a positive effect on water quality, using a long-term and spatially distributed monitoring data set that covered more than half of Belgium's territory. The contribution of forest in protecting water quality can have economic implications. For example, Fiquepron et al. (2013) and Vincent et al. (2016) found that a higher forest cover can be translated into lower drinking water supply costs in France and Malaysia, respectively.

On the contrary, a decline in forest cover may have a negative effect on water quality. Large scale deforestation can affect the physicochemical properties of downstream waters (Dessie and Bredemeier, 2013). In studying the impacts of deforestation in Amazonia, Langerwisch et al. (2016) found that deforestation will decrease riverine particulate and dissolved organic carbon amount by up to 90% and the discharge of organic carbon to the ocean will be reduced by about 40% under a severe deforestation and climate change scenario. This will have local and regional consequences on the carbon balance and habitat characteristics in the Amazon Basin itself as well as in the adjacent Atlantic Ocean. Changes in forest structure can also affect water temperature, with the removal of riparian canopy, generally leading to increased energy loading to the stream and higher stream temperatures (Bladon et al., 2016). Likewise, forest management can affect water quality. Higher management intensities can raise concentrations of suspended sediment and nutrients following silvicultural operations (e.g., Eriksson et al., 2011; Laudon et al., 2011; Siemion et al., 2011). The effects of harvesting will be higher when timber and biomass extraction bares the soil surface, thereby increasing the erosion risk (FAO, 2008).

Where timber and water are concerned, researchers tend frequently to think in terms of trade-offs between timber and water provision. Those trade-offs may go beyond timber and water, as trade-off between carbon sequestration and water provision services have been also reported in areas with water scarcity problems (Chisholm, 2010; Ovando et al., 2017). Similarly, forests also provide non-timber products, which should be considered in evaluating options (see Box 5.2). Trade-offs can also involve erosion regulation and water yields, whereas afforestation can provide relevant erosion reduction benefits while reducing water yield (Dymond et al., 2012). Large scale forest plantations can control sediment and nutrient loads and protect water quality (depending on their management), but this can lead to conflicts between beneficiaries of upstream plantations and downstream water users, where there is a demand for irrigation water (e.g., Nordblom et al., 2012).

Changes in coupled forest-water systems have significant impacts on biota. For instance, Ricketts et al. (2004) found that forest-based pollinators increased coffee yields by 20% and improved coffee quality within 1 km of forests in Costa Rica. Similarly, coupled forest-water systems support a large variety of birds, with a high percentage being dependent on forest habitats. Among the benefits that birds provide are pollination, insect pest control, seed dispersal and nutrient cycling (Wenny et al., 2011), but they also add substantial value to the economy through tourism, with bird-watching being one of the faster growing subsectors of ecotourism (Callaghan et al., 2018). The direct dependence of aquatic biodiversity on water quality and quantity render it specifically vulnerable to change. One of the ways for citizens to support informed policy development and decision-making is through applying local and traditional knowledge to local solutions and feeding those through to policy and management domains (see Box 5.3).

Box
5.2**The importance of non-timber products for millions of forest dwellers and indigenous people**

Poor people throughout many parts of the world depend heavily on the direct use of natural capital for their livelihoods. The multi-functional use of natural capital can be exemplified by Non-Timber Forest Products (NTFPs) which are essential inputs to forest people. In the burgeoning discussion around ecosystem services, the safety net that NTFPs provide for subsistence households must not be forgotten (Vira et al., 2015).

In the development of more effective macroeconomic policies for forest and water management, it is vital that we recognise that contrary to conventional production theory (Simon and Khan, 1984; Beckerman, 1995), other forms of capital (finance, infrastructure) cannot infinitely be substituted for natural capital (Daly and Cobb, 1989; Daly, 1999). As such, in the development of forest and water policy, we must recognise the multitude of disparate values held by different social groups in different ecological contexts. In an example provided from a study of Amerindian communities in Guyana (Sullivan, 2002), the ways people use forest resources have been identified and quantified. The monetary value of this use is calculated using an income accounting approach, and these use-values demonstrate the importance of non-timber forest products and services to people in these subsistence communities.

Findings of this study show that depending on location, the value of forest inputs amounts to between 33% and 63% of productive values in these communities. Whereas benefits derived from the harvesting of timber products are limited by forest production, the livelihood support that can be generated from the use of non-timber forest products and services are not limited in the same way. This has significant implications for sustainability, and it is important that if the quality of this income stream is to be preserved for future generations, action must be taken to ensure that this dimension of forest resources is not depleted by the decisions and actions of either local residents or global policymakers.

Forest-water services are about hydrological dynamics, and the socially-constructed relationships that underpin humans and ecosystems; for example, the rules, infrastructure and access to benefits and substitutes. The ability for humans to receive ecosystem services varies from place to place and from time to time, with the social-ecological system or political economy often playing a role in shaping the distribution of benefits of ecosystem services (Ostrom, 2009). For instance, frequent extreme weather and floods cause more loss of human life and property due to poor land use practices, poor planning and urbanisation on flood-prone areas and the poor often suffer the most due to lack of protection (Agrawal et al., 2008). Since these impacts are more severe at downstream locations, such communities are often willing to pay more to upstream communities for forest/water services. Therefore, the impacts of change on ecosystem services are varied in space and time and case-specific due



Shoebill (*Balaeniceps rex*) in Mabamba swamp in Uganda. The shoebill attracts many bird-watchers

Photo © Marius Claassen

to the ecological relationship between forest and water, as well as the socioeconomic relationship between humans and nature. Water is perceived to be useful only when people have access to it or have the ability to benefit from it (Brauman et al., 2007), however, the indirect benefits of water to people go beyond this narrow view.

A limitation of the literature in this area is that many recent forest hydro-economic studies apply engineering-oriented bottom-up models based on a combination of empirical relationships and theoretical control factors, such as behavioural responses or hydro-ecological processes (e.g., Garcia-Prats et al., 2016; Susaeta et al., 2016; 2017; Ovando et al., in press). These have been used to estimate the response of a production function (and associated revenues and/or costs) to specific interventions (Brouwer and Hofkes, 2008). Such relatively simple input-output production relationships examined at the individual plot level provide only a static view of the supply or demand of water ecosystem services (Harou et al., 2009). Only a few studies explicitly analyse the effect of forest interventions on water supply and demand, water prices and their welfare effect on economic sectors competing for water use (e.g., Nordblom et al., 2012; Garcia-Prats et al., 2016).

In arid and semiarid areas, the substitution of natural grasslands, shrubs and croplands with fast growing plantations, is often associated with decreases in streamflows and groundwater recharge, leading to potential conflicts between upstream plantations and downstream water users (e.g., Nordblom et al., 2012). At the local scale, increased tree cover can also be associated with reduced streamflow (Chapter 3). However, depending on the baseline conditions, the opposite may be true. Reforestation of degraded agricultural lands with heavily compacted soils may raise dry season stream flows by increasing infiltration rates and soil water holding capacity (Garcia-Chavesich et al., 2017). The role of forests in filtering sediments and other pollutants from water before it reaches

the stream, has increased the interest in using forest to increase water quality, thus reducing drinking water supply costs (Abildtrup et al., 2013; Vincent et al., 2016). An analysis of the relative magnitude of ecosystem services provided by forests can inform decisions on species selection, forest density and management options in relation to the regulation of water flow and quality under changing climatic conditions (Box 5.4).

5.3.2 Consequences of Change at Different Scales

Forest-water related services are dynamic and complex across scale and time. Each service has attributes of quantity, quality, location and timing of flow (Brauman et al., 2007). Scale is still considered to be the unresolved problem in the relationship between forests and water (Malmer et al., 2010). Knowledge about hydrologic services from forests are often based at catchment scale (Bruijnzeel, 2004). Such forest and water relationships are varied or dynamic in terms of the time scale. While in some cases (particularly in tropical areas), forest cover may be restored in a relatively short time, many forest-related water services (such as reducing sediment and enhanced water quality) may take much longer to recover (Malmer et al., 2010).

The impacts of changes in coupled forest-water systems are sensitive to boundaries in biophysical systems as well as jurisdictional boundaries. Natural boundaries include climatic zones (defined by altitude, rainfall, temperature, ocean proximity, etc.), surface water catchments (defined by topography) and groundwater basins (defined by hydrology, geology and topography). Anthropogenic alterations to these boundaries include inter-basin water transfers and changes in land cover. Eco-regions, discussed in Chapter 2 represent the combined impact of the above boundaries. Societal boundaries include differences in cultures and practices, economic conditions and jurisdictional boundaries. Since social values and practices can vary within and between communities, the impacts of change in forest and water systems and related services will also vary between different societal contexts, whereas economic circumstances influence development priorities and options. Jurisdictional boundaries, which can be sub-national (e.g., districts), national (sovereign countries) or regional (e.g., regional economic commissions) are associated with different policy contexts, economic conditions and societal perspectives. The determinants of change (Chapter 3), changes to the forest/water system (Chapter 4) and response options (Chapters 6 and 7) are all sensitive to natural and societal boundaries. Temporal scale is another type of boundary, since the time scales of political, policy, economic and social process are not always aligned with the time frames of environmental (forest/water) impacts and responses.

The delivery of water-related forest ecosystem services is scale-dependent in terms of biophysical processes (Chapters 3 and 4), but also in terms of governance processes. More recent literature on forest-water interactions and dynamics suggests that the boundaries for the

Box
5.3

Toad's eye views and water quality

Science and technology as concepts, howsoever varied might be the understanding behind them, underlie any discussion of water and forest management. It is when deconstructing these concepts with questions like whose science? What kind of technology? Which methods? What inherent capacity for maintenance? etc. that the hegemony of particular technologies and scientific approaches expose themselves. Research methods can be classified as those primarily aimed at external learning ('extractive science'), those primarily supporting local learning, and methods that explicitly aim for both (Mehmood-UI-Hassan et al., 2017).

Methods that match local concerns over water and are yet understandable by scientists and forest management officials can play a substantial role in negotiations on clarifying sources of pollution and changes in flow regime linked to local land uses (Tomich et al., 2004; van Noordwijk et al., 2016). Biological water quality monitoring methods (Rahayu et al., 2013a, b) have been used to support local stakeholders in forest mosaic landscapes where land use patterns are contested.

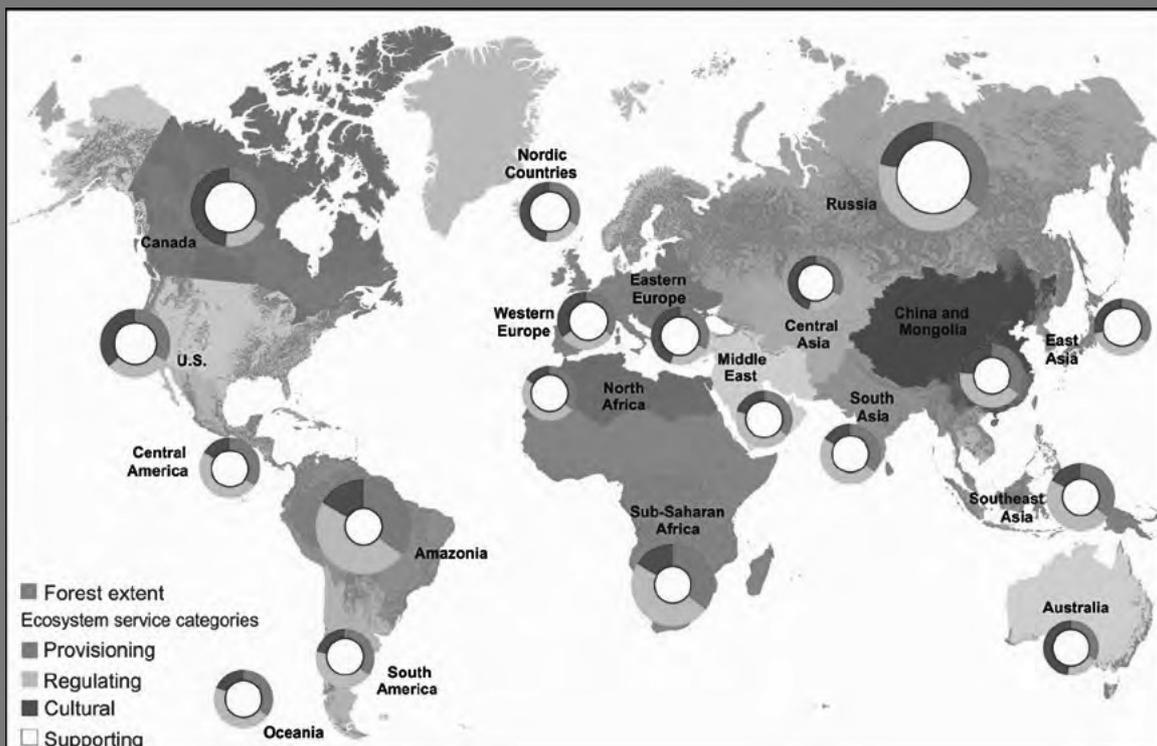
In water management, there is growing awareness that there are traditional technologies, some of them centuries old, which are perfectly well-suited to the hydro-ecology of the region in which they are found in as well as to the availability of local raw material and skills for their use. On the other hand, modern ferro-cement technologies or piped water systems may be efficient, but beyond the capacity of the local people or their raw material resource base to maintain or restore if damaged by a disaster. Brushwood dams still account for over two-thirds of actual irrigation in the Himalayas and the technology they deploy is dependent on the collective capacity of the local irrigation community. These dams are built at the start of the dry season to divert water to the fields and are washed away during the monsoon, only to be re-built in the next dry season. When they are replaced by modern ferro-cement dams, the modern technologies are very efficient as long as they operate, but when a major flood occurs that damages or washes them away, they are abandoned as un-restored relics and people revert back to their traditional technology (Gyawali, 2004). A similar story is unfolding in the semi-arid zones of western India with traditional water harvesting technologies as modern technologies fail to deliver (Agarwal and Narain, 1997)

If technology is defined as science which has commercial implications, the disjuncture between modern and traditional technologies is explained by the power and bias of the most powerful market players. The decision-making process remains dominated by investments backed by modern technologies involving cement, steel, petroleum-based plastics and other such powerful artefacts, while local communities are alienated and disempowered, and their traditional technologies marginalised. Ethnographic and anthropological studies of science and technology have tried to distinguish between 'toad's eye' or civic science and 'eagle eye' or modern western science to reveal this contrast in approaches to knowledge.

An attempt to quantify the value of forest-based ecosystem services

An approach to quantify service domains is to divide the world into regions and calculate the amount of tropical and temperate/boreal forest for each region. These estimates are then grouped into provisioning, regulating and cultural ecosystem services (ES), summed to estimate the total valuation per category, and then multiplied by the area of tropical and temperate/boreal forest in each region to produce an estimate of the total ecosystem service category value for each region. While the value of supporting services can also be calculated in this way, these services are deemed as intermediate services that are implicitly embedded in the final value of regulating, provisioning and cultural services (Hein et al., 2006).

The portfolio and relative magnitude of the different ES types, based on the above calculations and ES values from De Groot et al. (2012) are presented in Figure 5.2, with the supporting services being superimposed on the other three types of services to denote their intermediate nature. The figure illustrates the variation in different ecosystem services provided by forests in different regions of the world. Based on this approach, northern regions generally provide more cultural services than southern regions, which may reflect differences in sustainable forest management between these two regions (Fisher et al., 2009; Chan et al., 2012). These north/south differences may be attributed to the differing ecosystem service values for tropical and temperate/boreal forests, with tropical forests providing a larger amount of provisioning and regulating ecosystem services and temperate/boreal forests providing a larger amount of cultural and supporting ecosystem services. The loss or degradation of a forest in one region may have different consequences for water security than in another region. Forest provisioning and regulating ecosystem services – in particular – have important implications for water security. To mitigate the potential loss of specific ecosystem services, a portfolio of functions should be maintained on the landscape. A portfolio of function approach ensures that there exist forests in a specific region that provide low, medium and high levels of each of the categories of ecosystem services. This portfolio approach illustrates that the ecosystem can better buffer against change and ensure that forests provide a suite of ecosystem services to the landscape's inhabitants.



Source: Authors' own elaboration based on calculations from De Groot et al., 2012

Figure 5.2. Map illustrating the portfolio and relative magnitude of ecosystem services provided by forests. The relative magnitude of ecosystem services for each region is illustrated by the relative size of the circles, whereas the relative size of each segment represents the value of each service

political governance of these interactions need to be greatly extended in space (Ellison et al., 2017). Most cross-regional and international water management frameworks for negotiation consider only the catchment (watershed) boundaries and include actors situated at least in part within the catchments. However, in order to adequately

address water availability concerns and impacts with respect to forests and land use change, it is necessary to redesign these frameworks such that they can actually take into account the principal contributions from a much broader concept of hydrologic space. The 'precipitation-shed' approach is currently perhaps the best example of

this concept. Since land use practices, both upwind and within the given catchment, ultimately influence the total amount of water that is either consumed locally or re-distributed onto other downwind basins (Dirmeyer et al., 2009), it is of explicit interest to be able to harness these factors in the service of the larger framework of forest and water management strategies. Thus, both local, regional and larger forest and water management strategies and institutional systems need to find meaningful ways of not only incorporating and involving up- and downstream interests, but also of involving up- and downwind communities in the larger overall forest and water management framework.

5.3.3 Consequences for Human Well-Being

As described earlier, changes in forest status can lead to significant changes in hydrological functions, which in turn translate into changes in the provision of ecosystem services (Lele, 2009). Besides the biophysical repercussions, these changes have direct and indirect socio-economic consequences well beyond forests' boundaries (Gregory, 2006; Wang-Erlandsson et al., 2017). For example, in China controversial resettlement schemes have been a key instrument for the government to address poverty and environmental degradation in the past two decades, with up to 6 million of the 120 million internally-displaced people qualifying as environmental migrants (Myers, 2002). These schemes have been associated with both ecological and social consequences (Fan et al., 2015) with some cases showing that resettlement promotes ecosystem recovery by removing human pressures (notably from grazing livestock) and improving access to infrastructure, education, and health care. In other cases, however, there are also negative social and ecological impacts in newly resettled areas, including a disruption of the coupled social-ecological system among resettled communities.

One conceptualisation which helps to understand the well-being implications of changes in hydrological functions is derived from neoclassical economics, based on the measurement of welfare changes in monetary units (Pearce and Turner, 1990; Bateman et al., 2011). Under this conceptualisation, changes in well-being are directly linked to the value that humans attach to ecosystem services, measured through the monetary trade-offs that individuals are willing to undergo to secure the service. As an illustration, Box 5.5 provides current evidence of the monetary value of water ecosystem services delivered by forests, focusing on two forest systems of global relevance: tropical forests in Central and South America and mangrove forests in South East Asia. This evidence provides some basis for the general understanding of the welfare benefits that forest conservation provides in relation to water ecosystem services and, as a corollary, of the welfare loss associated with the decline in the state of ecosystems. However, it should be noted that this literature is very heterogeneous in purpose and approaches, providing a very fragmented view of the value of forest water services (Lele, 2009; Ojea et al., 2012).

Despite their limitations, a growing number of studies offer some insights into the economic implications of forest conservation and management for the provision of water ecosystem services. For example, some econometric

Box
5.5

Evidence of the monetary value of water services provided by forests

Tropical forests in South and Central America

Ojea and Martin-Ortega (2015) undertook a meta-analysis of 25 primary valuation studies of water services of tropical forests in Central and South America, which served to identify some factors that systematically influence forest values.

The review of this literature reveals how the definition and classification of water ecosystem services is highly inconsistent (Ojea et al., 2012), which can generate problems such as double counting (Fisher et al., 2009).

The meta-analysis shows that the relationship between the value and type of service is complex and is mediated by the type of beneficiary. Extractive water supply services (involving mostly agricultural and human water consumption) have, in general, relatively high values; although the value of flow-regulating services (in-stream water supply) when the beneficiary is an industrial user (i.e. mostly used for hydropower production) is significantly higher than when used for agricultural and human consumption (but not as high as extractive water supply generally).

There is much less consolidated evidence on the monetary value of damage mitigation and water cultural related benefits in comparison to provisioning services.

Mangroves in South East Asia

Brander et al. (2012) undertook a meta-analysis of 41 studies assessing the value of mangrove ecosystem services around the world to project values for South East Asia.

The range of ecosystem services represented in the collected studies includes provisioning services (fish, fuelwood, materials) and regulating services (coastal protection, flood prevention, water quality). Similarly to the case of tropical forest, the value of cultural ecosystem services is under-represented in the literature.

The type of service significantly affects its value, with water quality and fisheries having a positive and significant effect on this value. Mangrove value is also influenced by the existence of other mangrove forests in the area. This seems to indicate that fragmentation of mangroves and their surroundings (e.g., by road infrastructure) has a negative effect on the value of mangrove.

The median mangrove value in the sample is USD 239 per hectare per year (2007 prices).

Brander et al. 2012 also forecast the value change associated with a projected 2000 – 2050 scenario and estimate an annual value of lost ecosystem services from mangroves in South East Asia, which amounts approximately to USD 2.16 billion in 2050 (2007 prices).

studies provide empirical evidence on the positive effect of forest cover (thus conservation) in reducing the costs of drinking water supply (Ernst et al., 2004; Abildtrup et al., 2013; Figuepron et al., 2013; Vincent et al., 2016). Forest conservation is also expected to generate positive income and welfare effects by controlling dam sedimentation and increasing hydropower generation (Arias et al., 2011), or by reducing flooding damages for downstream farmers (Kramer et al., 1997). A number of other studies suggest trade-offs between the production of timber and water ecosystem services, implying an opportunity cost (revenues foregone) for landholders, in cases where no compensation schemes are in place (Eriksson et al., 2011; Kucuker and Baskent, 2015; Simonit et al., 2015; Garcia-Prats et al., 2016). A small number of studies are also starting to look at the economic implications of the trade-offs between water quantity and quality associated with forest practices, trying to integrate economic values associated with water ecosystem services into decision support systems (Keles and Baskent, 2011; Kucuker and Baskent, 2015; Mulligan et al., 2015; Garcia-Prats et al., 2016). These studies reveal that internalising water values leads to different optimal forest management decisions than are based on the single maximisation of timber net benefits, which highlights the need for advancing water ecosystem services valuation and integration into decision-making processes.

Further inspection of the literature also demonstrates how most of the existing evidence on the value of (water) ecosystem services provided by forests focuses on limited types of ecosystem services: predominantly provisioning and some of the regulating services; other regulating services and especially cultural ecosystem services, are limited in the monetary valuation literature. There are studies on the recreational value of forests (Chiabai et al., 2011 reviewed some of them) but the link to water ecosystem services is often unspecified which is consistent with the fact that less tangible services are harder to measure and hence tend to be ignored. This represents a critical limitation since the tendency to avoid services that are difficult to measure creates a bias in resultant policy choices. Moreover, it is increasingly argued that water-based ecosystem services provide benefits that go beyond what can be monetised. Even in the realm of human health alone, poor management of water and forest systems has been shown to result in increases in water borne diseases, increasing risk to humans from flooding and coastal inundation, and reducing food security (Corvalan et al., 2005).

5.3.4 Social Consequences and Distributional Considerations

The consequences of changes in forest-based water ecosystem services are not evenly distributed. While aggregate availability of water, as well as its quality, might be reflected in catchment level or system-wide analyses, the spatial distribution of this water, as well as the social and political context within which people have access to or are able to benefit from such services can be highly unequal

(Mollinga, 2008; Loftus, 2015). When considering the forest-water relationship in terms of impacts, it is thus important to be mindful of questions of distributional equity, fairness and justice (Sikor et al., 2014), the political economy of water allocation which underpins who gets

Box
5.6

Water allocation in the Murray Darling Basin, Australia: conflicts and consequences

The Murray Darling basin, located in southeastern Australia, covers over 1 million km² (14% of Australia's landmass) and contains over 30,000 wetlands. It provides water storage and multiple other ecosystem services across its vast floodplain. Throughout human history the river has played an important role, providing sustenance for Aboriginals and supporting national economic development (Bark et al., 2012). Today some 65% of all irrigated land within Australia lies within its boundaries, accounting for over 95% of total irrigation water withdrawals (Tan et al., 2012).

By the 1990s, threats to the basin's capacity to deliver water for multiple uses across its complex social and political boundaries led to recognition that the river had to be managed holistically, in a more integrated way. As a result, the Murray Darling Basin Authority (MDBA, 2012) was formed. Although at first these institutional developments were positively received at the state and local government levels, many farmers depending on irrigation continued to resist water allocation change. Exacerbated by extreme drought conditions in the first decade of the new millennium, and over-allocation of abstraction licences (Crossman and Overton, 2011), the federal government decided to implement a more stringent plan for water allocation, based on the concept of Sustainable Diversion Limits (SDLs). Key ecosystem functions such as sediment loads, nutrients, carbon exchange, habitat maintenance and connectivity (Falkner et al., 2009) were to be monitored to ensure compliance with the objectives of the plan, and a framework for transferable water rights through a water market were identified (Qureshi et al., 2009), based on water 'buy-backs' being strictly made on the basis of 'willing seller, willing buyer' agreements.

Unexpectedly for the government, when this plan was first released for public consultation in 2011, (MDBA, 2011) panic and chaos ensued among farming communities across the basin. This resulted in demands for a series of additional special studies (Bark et al., 2012; Mooney and Tan, 2012; Jiang and Grafton, 2012), and much debate about what the appropriate level for the SDLs should be (McKay, 2011; MDBA, 2012), causing long delays in the implementation of the plan. Even by early 2018, the debate over ecological water allocations remains unresolved. Making the situation worse, evidence has recently come to light of illegal water transfers being made in many areas of the basin, with national newspaper headlines drawing attention to 'water theft'. Meanwhile, many areas of the floodplain forests of iconic Red Gums continue to decline, conflicts between land and water users remain, and many forest and former wetland areas are consumed by the increasing number of bushfires occurring every year.

how much water, when and where, and to recognise that this is likely not to be equally available to all stakeholders across a landscape. Importantly, environmental flow requirements should be considered in allocation processes. A reflection on access and distribution is provided in Box 5.6.

5.4 Scenario Analysis: Consequences of Change in the Future

Anticipating changes under the ‘new normal’ is necessary in order to establish likely changes to the forest-water-climate-people system and to determine appropriate measures based on desired objectives. A scenario analysis helps to project into the future.

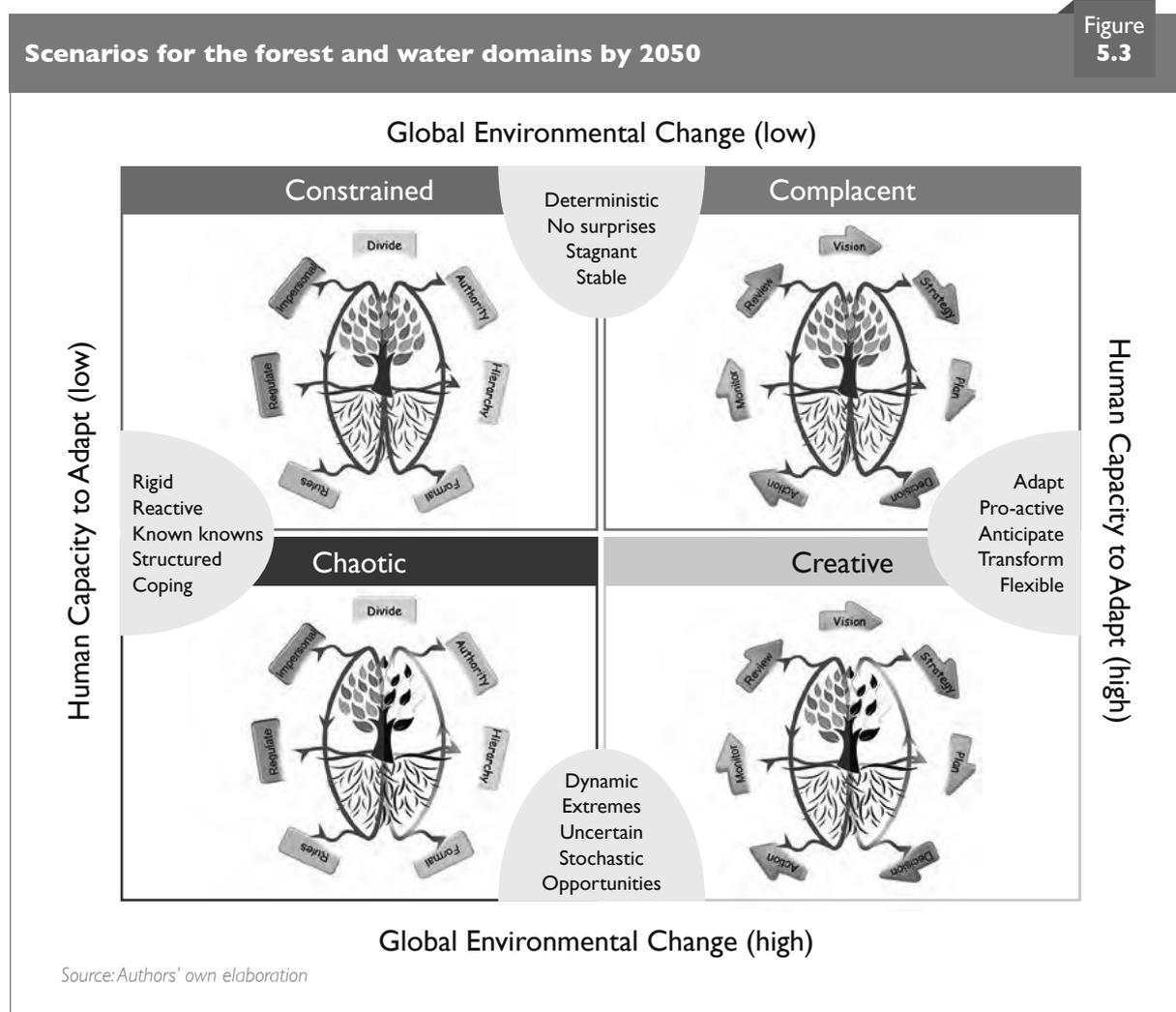
‘Normal’ generally refers to conditions that are similar to what they have been in the past (Hulme et al., 2009). The ‘new normal’ describes future conditions that are markedly different from the past (see Chapter 1). An example of such a ‘new normal’, is the anthropocene as a new geological epoch, where humans predominantly drive planetary changes (Zalasiewicz et al., 2010). The new normal in the context of impacts and consequences for changes in coupled forest-water systems will be characterised by greater complexity and uncertainty and shifts in risk perceptions. Such changes are generally viewed as

undesirable, but some changes can also translate into new opportunities. Response options can include preventative measures (to counter undesirable change) and mitigation measures to reduce the impacts of such change or measures to exploit the opportunities brought about by change.

5.4.1 Future Impacts and Consequences

Representations of future possibilities can be useful for long term strategy development, but also to direct actions in the short term to promote a desired future state (Funke et al., 2013). Scenario planning originates in military applications, with Sun Tzu acknowledging the importance of planning in the face of uncertainty 2,400 years ago (Giles, 1910), whereas contemporary applications of scenarios include the RAND Corporation that started to investigate the scientific use of expert opinion in planning for the future in the 1940s (Landeta, 2006) and Royal Dutch Shell that used scenario tools to good effect in the 1970s, leading to a competitive advantage that enabled them to act quickly during the oil price shock of 1973 (Daum, 2001; Wilkinson and Kupers, 2013).

The determinants of change discussed in Chapters 2 and 3 include societal dimensions and environmental dimensions. In this chapter we aggregated the drivers in two



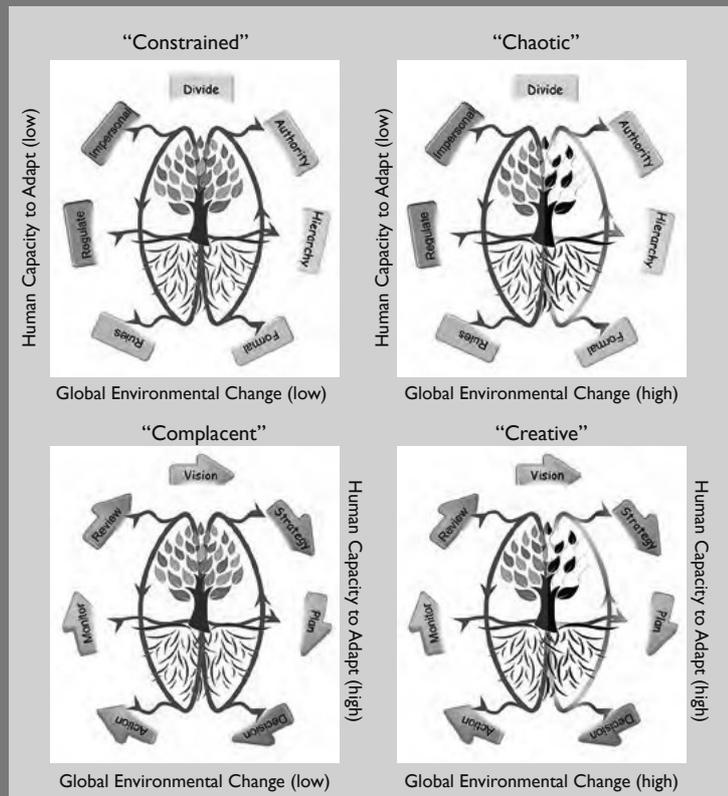
Impacts and consequences of coupled forest-water systems in relation to scenario contexts

In a 'Constrained' future, where environmental change is low and human capacity to adapt is low, there is a stable resource base but a constrained societal capability to service growing demands. The structure and function of the coupled forest-water system is variable, but with little directional change. Consequently, there is high confidence in the goods and services that the system can provide. The utilisation of these goods and services to meet societal development objectives is however constrained by limited innovation and adaptation and societal needs are not met in the future.

A 'Chaotic' future has high levels of global change and low human capacity to adapt. The structure and function, and associated goods and services, of the coupled forest-water systems change, as do the benefits that society obtain from those services. This can lead to societal losses, compromising livelihoods and affecting human well-being.

A 'Complacent' future combines low levels of environmental change with high levels of adaptive capacity. The goods and services that are provided at a high level of confidence allow for deliberation on the best portfolios of social and economic development opportunities. The adaptive capacity furthermore allows for learning by doing, thus ongoing adjustments to translate goods and services into development outcomes. However, unexpected impacts such as political dynamics, technological innovations or societal values can lead to unintended consequences, such as inequitable consumption and benefits at different temporal and spatial scales.

A 'Creative' future combines a high rate of environmental change with a high level of adaptive capacity. This can lead to continuously changing goods and services, and adaptation to the change through ongoing evolution of social and economic activities to harness the dynamic potential. Since the growing demand is met with dwindling resources, incremental improvement is generally not sufficient. Radical and disruptive innovation is needed to meet development aspirations while countering environmental change. This scenario can either spark creative thinking or lead to despair if the challenges are deemed too great to overcome.



higher order drivers: the rate of environmental change and human capacity to adapt, as juxtaposed axes for scenarios related to forest and water by 2050. The resultant scenarios are presented in Figure 5.3. Perspectives on the impacts and consequences of these scenarios are outlined in Box 5.7.

In the scenario analysis presented here, the environmental change components and the relation with demand for these resources are represented by a tree (forest system) and blue arrows (water cycle). Considering future scenarios of environmental change, two options are possible: low levels of change in the forest system, represented by a symmetric tree, or high levels of change, represented by an asymmetric tree. Likewise, in future scenarios where environmental change in water systems is low, the water cycle is shown as a symmetric blue colour, whereas high levels of change in the water systems are indicated by an asymmetric colour of the water cycle.

The human dimensions of change include societal perspectives and solutions. Where there is a low capacity

to adapt, the external circle is composed of rectangles, whilst when there is a high capacity to adapt, the circle is composed of arrows.

Under future scenarios with low levels of global environmental change, the coupled forest-water system can be expected to function as a dynamic system with natural variability, but within known ranges of variability. The flow regimes, linkages between climate and vegetation, and forested landscapes will be dynamic, but remain within boundaries that do not cause a major change in the coupled system.

Future scenarios with high levels of global environmental change, however, can cause disruptions in the natural systems and processes. Examples can include shifts in oceanic currents and sea surface temperatures with drastic effects on local climates beyond what can currently be predicted with any confidence or precision.

Variation along the other axis, human ability to adapt, will first of all depend on increased communication and synergy between the different knowledge systems

(including local knowledge and values, physical, biological and social science-based knowledge and current policies) that may currently compete. If the various knowledge-to-action chains can be connected, societal adaptation may keep up with the environmental change and avoid the passing of irreversible thresholds.

5.4.2 Governance Responses under Different Scenarios

Under conditions of ‘*Low human capacity to adapt*’, governance systems can be structured and efficient under stable conditions (thus ‘*Low global environmental change*’). A policy that assumed such a scenario is the US Endangered Species Act, which pursued a return of ecosystems to their ‘historical’ natural conditions and emphasised restoring habitat for single species, often to the exclusion of other species, but with increasing rates of global environmental change, those systems have been transformed beyond return, precluding more adaptive responses (DeCaro et al., 2017).

Under conditions of ‘*High global environmental change*’, the same ‘robust’ systems can be slow to adapt to changes, which can render seemingly good policies less effective. An example of such a situation is Lake Chad, which shrank by 90% over a period of 35 years, which is putting pressure on sustainable food production, wetland habitat conservation, water management in transboundary basins and adaptation to climate change (Zieba et al., 2017). This situation must be taken into account in the formulation of enabling framework policies for managing resources in the Lake Chad area.

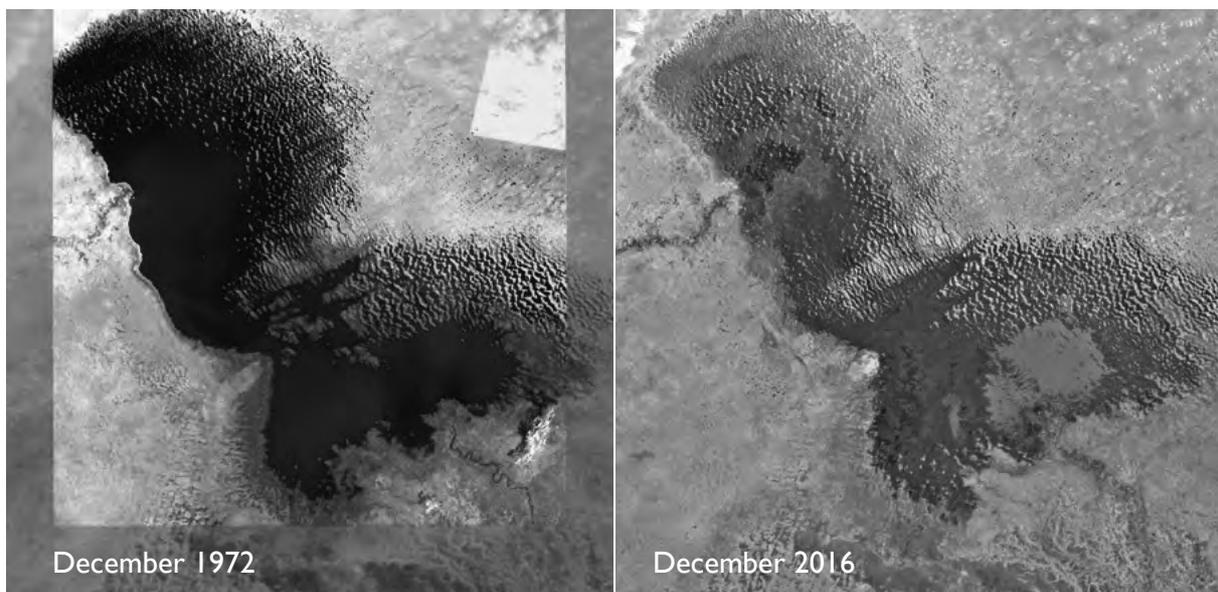
A scenario that seems to be a desired future is where ‘*Low global environmental changes*’ are prevalent and where there is ‘*High human capacity to adapt*’. Although there are fewer uncertainties about environmental conditions in this context, the opportunities brought about by

change are also limited. Policy options under these conditions will focus on sustainable practices on the supply side (forests and water) and greater efficiencies in the ever-increasing demand side (social and economic activities). There is a danger of complacency in this scenario, where environmental change may not be immediately apparent, such as the case in southeastern Spain, where intensive groundwater use and mining often exceed replenishment of supplies (Aldaya, 2017).

‘*High human capacity to adapt*’ is best demonstrated in conditions with ‘*High global environmental change*’. In such contexts, the ideal policies are framework policies that enable adaptive approaches and are supported by rapid feedback loops and learning systems. Ultimately, adaptive governance consists of a range of interactions between actors, networks, organisations, and institutions emerging in pursuit of a desired state for social-ecological systems (Chaffin et al., 2016).

The drivers of change are relevant at global, regional, national and local spatial scales, however, their manifestation would be different at each scale. Environmental change may be driven by global systems but has significant implications for local conditions. Likewise, the capacity to adapt to change can be facilitated through policies and processes at scale, but also depend on local capacity for action. These dimensions emphasise the need for cooperation across scales to mitigate change and increase adaptive capacity.

The implication of this scenario approach for water and forest interlinkage lies in its reframing of the social response to risk and uncertainties and in viewing policy as not just something within the government domain but also within that of markets and civic movements (Gyawali and Thompson, 2016). Such a triad understanding of power and policy is also what has been described by other schools of thought (e.g., Karl Polanyi (1944) with his concept of exchange, redistribution and reciprocity,



Lake Chad, straddling Cameroon, Chad, Niger and Nigeria, has shrunk by 90% over a period of 35 years
Photo © Mapdata: Google; NASA, U.S. Geological Survey, Landsat/Copernicus

as well as Lukes (1974) with his triad understanding of power). Such a framing will also have implications for management and governance (discussed in Chapters 6 and 7): firstly by defining policy as not just an action by governments but also by markets and civic movements; and secondly by bringing uncertainty and surprise – and the plural social response to them – to the centre stage.

5.5 Data Needs and Knowledge Gaps

- There is still much to learn about the ways eco-hydrological and socioeconomic processes can be integrated into forest and water resources management and planning strategies.
- More recognition of the shortcomings of current knowledge of biophysical processes is needed, along with relationship of these to the generation of ecosystem services and their values.
- We need further integration of biophysical information into the design of valuation scenarios, including new and innovative epistemological approaches for integration which can cope both with biophysical uncertainty and human ‘ambiguities’ (Byg et al., 2017). i.e. the agenda on valuation should be driven by a better representation of both the biophysical and social complexities (rather than necessarily on instrumental sophistication) (Martin-Ortega et al., 2017).
- Also, new efforts should be directed towards integrating monetary and non-monetary values, and operationalising these and other forms of value into decision making including relational values.
- While there is expertise concerning integrating monetary and non-monetary values at lower geographical scales, challenges remain in scaling up the analyses to regional and global scales.
- There is a need to build more sophisticated forest hydro-economic models based on integrated frameworks, to guide optimal resource allocation between forests (and other land uses) and water ecosystem services. Such models would need a detailed representation of forest functionality and its explicit relationship to watershed-based ecosystem services and their values (Ferraz et al., 2014)

5.6 Conclusions

Linkages between coupled forest-water systems and benefits to people are generally well understood but there are some limitations, specifically across spatial and temporal scales. The ability to attach values to these benefits is often lacking in terms of monetary metrics and even more so for non-monetary metrics.

The lack of a systematised approach to the valuation of water ecosystem services provided by forests hinders their incorporation into mainstream decision-making. Coupled forest-water systems’ interactions are characterised by great complexity and uncertainty across space and time, in which trade-offs and synergies of goods and services are governed by complex environmental and management factors and interactions. Those environmental

and management interactions are magnified when linked to the complex socio-economics and political boundaries given the multiple human well-being dimensions that can be affected by forest-water related ecosystem services. That leads to a recognition that complex socio-ecological forest and water interactions need to be managed holistically and in a more integrated way.

Changes to the underlying structure and function of coupled forest-water systems will affect available goods and services and consequent development options. While these linkages are conceptually well-understood, we need to improve our ability to characterise the relationships to support choices about management and policy options. Under future scenarios with low levels of global change, the coupled forest-water system can be expected to function as a dynamic system with natural variability, but within known ranges of variability.

The ‘new normal’ in the context of impacts and consequences for changes in coupled forest-water systems is characterised by greater complexity and uncertainty and shifts in risk perceptions. Such changes are generally viewed as undesirable, but some changes can also translate into new opportunities. However, consequence of changes in forest-based water services are not evenly distributed, affecting unequally people’s rights and responsibility. Social justice and institutional arrangements need to be examined within the particular political and historical settings.

Responses under different future scenarios incorporate state, market and civic domains. For the coupled complex system to evolve towards sustainability, it is necessary for all these voices (including those of women and other marginalised groups) to be heard and responded to in a spirit of constructive engagement.

Current knowledge suggests that there is well established evidence on the fact that changes in the structure and functions of forests result in changes in the delivery of water ecosystem services, and these have consequences for the benefits people can obtain from forests. However, substantial levels of uncertainty remain in elaborating the details of the direction and magnitude of these relationships, but methods for improving our understanding of these consequences are rapidly developing. These methods are improving at the local/lower levels (e.g., catchment or lower levels), which means that the evidence they provide is quite solid, although still limited to specific places where data and monitoring systems are in place. However, there is more work to be done in terms of expanding this understanding to ‘data scarce’ locations. Much more needs to be done in terms of understanding and bringing this up to broader and global scales.

References

- Abildtrup, J., Garcia, S. and Stenger, A., 2013. The effect of forest land use on the cost of drinking water supply: A spatial econometric analysis. *Ecological Economics*, 92, pp.126-136.
- Agarwal, A. and Narain, S. eds., 1997. *Dying Wisdom: Rise, fall and potential of India's traditional water harvesting systems*. Delhi: Centre for Science and Environment.
- Agrawal, A., McSweeney, C. and Perrin, N., 2008. *Local Institutions and Climate Change Adaptation*. Washington DC: World Bank.
- Aldaya, M.M., 2017. Environmental science: Eating ourselves dry. *Nature*, 543(7647), p.633.
- Altman, J.C. and Branchut, V., 2008. *Fresh Water in the Maningrida Region's Hybrid Economy: Intercultural Contestation Over Values and Property Rights*. Canberra: Centre for Aboriginal Economic Policy Research.
- Arias, M.E., Cochrane, T.A., Lawrence, K.S., Killen, T.J. and Farrell, T.A., 2011. Paying the forest for electricity: a modelling framework to market forest conservation as payment for ecosystem services benefiting hydropower generation. *Environmental Conservation*, 38(4), pp.473-484.
- Bark, R., MacDonald, D. H., Connor, J., Crossman, N., Jackson, S. 2011. Water values. In: *Water Science and Solutions for Australia*, Prosser, I. (ed.). Collingwood: CSIRO Publishing.
- Bark, R.H., Garrick, D.E., Robinson, C.J. and Jackson, S., 2012. Adaptive basin governance and the prospects for meeting Indigenous water claims. *Environmental Science & Policy*, 19, pp.169-177.
- Bateman, I.J., Carson, R.T., Day, B., Hanemann, M., Hanley, N., Hett, T., Jones-Lee, M., et al., 2002. *Economic Valuation with Stated Preference Techniques: A Manual*. Cheltenham, UK/ Northampton, MA: Edward Elgar Publishing.
- Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G. and Turner, K., 2011. Economic analysis for ecosystem service assessments. *Environmental and Resource Economics*, 48(2), pp.177-218.
- Beckerman, W., 1995. *Small is stupid: Blowing the whistle on the Greens*. London: Duckworth.
- Bladon, K.D., Cook, N.A., Light, J.T. and Segura, C., 2016. A catchment-scale assessment of stream temperature response to contemporary forest harvesting in the Oregon Coast Range. *Forest Ecology and Management*, 379, pp.153-164.
- Boyd, J. and Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics*, 63(2-3), pp.616-626.
- Brander, L.M., Wagtendonk, A.J., Hussain, S.S., McVittie, A., Verburg, P.H., de Groot, R.S. and van der Ploeg, S., 2012. Ecosystem service values for mangroves in Southeast Asia: A meta-analysis and value transfer application. *Ecosystem Services*, 1(1), pp.62-69.
- Brauman, K.A., Daily, G.C., Duarte, T.K.E. and Mooney, H.A., 2007. The nature and value of ecosystem services: an overview highlighting hydrologic services. *Annu. Rev. Environ. Resour.*, 32, pp.67-98.
- Brockington, D. and Duffy, R., 2010. Capitalism and conservation: the production and reproduction of biodiversity conservation. *Antipode*, 42(3), pp.469-484.
- Brogna, D., Míchez, A., Jacobs, S., Dufrière, M., Vincke, C. and Dendoncker, N., 2017. Linking forest cover to water quality: A multivariate analysis of large monitoring datasets. *Water*, 9(3), p.176.
- Brouwer, R. and Hofkes, M., 2008. Integrated hydro-economic modelling: Approaches, key issues and future research directions. *Ecological Economics*, 66(1), pp.16-22.
- Bruijnzeel, L.A., 2004. Hydrological functions of tropical forests: not seeing the soil for the trees? *Agriculture, Ecosystems & Environment*, 104(1), pp.185-228.
- Byg, A., Martin-Ortega, J., Glenk, K. and Novo, P., 2017. Conservation in the face of ambivalent public perceptions—The case of peatlands as ‘the good, the bad and the ugly’. *Biological Conservation*, 206, pp.181-189.
- Callaghan, C.T., Slater, M., Major, R.E., Morrison, M., Martin, J.M. and Kingsford, R.T., 2018. Travelling birds generate eco-travellers: The economic potential of vagrant birdwatching. *Human Dimensions of Wildlife*, 23(1), pp.71-82.
- Castello, L. and Macedo, M.N., 2016. Large-scale degradation of Amazonian freshwater ecosystems. *Global Change Biology*, 22(3), pp.990-1007.
- Chaffin, B.C., Garmestani, A.S., Gunderson, L.H., Benson, M.H., Angeler, D.G., Arnold, C.A., Cosens, B., et al., 2016. Transformative environmental governance. *Annual Review of Environment and Resources*, 41, pp.399-423.
- Chan, K.M., Satterfield, T. and Goldstein, J., 2012. Rethinking ecosystem services to better address and navigate cultural values. *Ecological Economics*, 74, pp.8-18.
- Chan, K.M., Balvanera, P., Benessaiah, K., Chapman, M., Díaz, S., Gómez-Baggethun, E., Gould, R., et al., 2016. Opinion: Why protect nature? Rethinking values and the environment. *PNAS*, 113(6), pp.1462-1465.
- Chiabai, A., Travisi, C.M., Markandya, A., Ding, H. and Nunes, P.A., 2011. Economic assessment of forest ecosystem services losses: cost of policy inaction. *Environmental and Resource Economics*, 50(3), pp.405-445.
- Chisholm, R.A., 2010. Trade-offs between ecosystem services: water and carbon in a biodiversity hotspot. *Ecological Economics*, 69(10), pp.1973-1987.
- Callaghan, C.T., Slater, M., Major, R.E., Morrison, M., Martin, J.M. and Kingsford, R.T., 2018. Travelling birds generate eco-travellers: The economic potential of vagrant birdwatching. *Human Dimensions of Wildlife*, 23(1), pp.71-82.
- Corvalan, C., Hales, S. and McMichael, A., 2005. *Ecosystems and human well-being: health synthesis. A report of the Millennium Ecosystem Assessment*. Geneva: WHO.
- Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'neill, R.V., Paruelo, J. and Raskin, R.G., 1997. The value of the world's ecosystem services and natural capital. *Nature*, 387(6630), p.253.
- Crossman, N. and Overton, I., 2011. *Returning water to the environment for multiple ecosystem service benefits, Murray-Darling Basin, Australia. Water for a healthy Country*. National research flagships CSIRO ESP Conference, 2011, 4th October.
- Daly, H., 1999. *Ecological Economics and the Ecology of Economics*. Cheltenham: Edward Elgar.
- Daly, H., and Cobb, J., 1989. *For the Common Good*. Boston, MA: Beacon Press.
- Daily, G. C., 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Washington, DC: Island Press.
- Daum, J.H., 2001. *How scenario planning can significantly reduce strategic risks and boost value in the innovation value chain*. The New Economy Analyst Report.
- DeCaro, D., Chaffin, B., Schlager, E., Garmestani, A. and Ruhl, J.B., 2017. Legal and institutional foundations of adaptive environmental governance. *Ecology and Society*, 22(1).
- Dessie, A. and Bredemeier, M., 2013. The effect of deforestation on water quality: A case study in Cienda micro watershed, Leyte, Philippines. *Resources and Environment*, 3(1), pp.1-9.
- De Groot, R., Brander, L., Van Der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., et al., 2012. Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services*, 1(1), pp.50-61.
- De Groot, R.S., Wilson, M.A. and Boumans, R.M., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, 41(3), pp.393-408.

- Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., et al., 2015. The IPBES Conceptual Framework—connecting nature and people. *Current Opinion in Environmental Sustainability*, 14, pp.1-16.
- Dietz, T., Fitzgerald, A. and Shwom, R., 2005. Environmental values. *Annu. Rev. Environ. Resour.*, 30, pp.335-372.
- Dirmeyer, P.A., Brubaker, K.L. and DelSole, T., 2009. Import and export of atmospheric water vapor between nations. *Journal of Hydrology*, 365(1-2), pp.11-22.
- Dymond, J.R., Ausseil, A.G.E., Ekanayake, J.C. and Kirschbaum, M.U., 2012. Tradeoffs between soil, water, and carbon—a national scale analysis from New Zealand. *Journal of Environmental Management*, 95(1), pp.124-131.
- Eckholm, E.P., 1975. The deterioration of mountain environments. *Science*, 189(4205), pp.764-770.
- El Serafy, S., 1998. Pricing the invaluable: the value of the world's ecosystem services and natural capital. *Ecological Economics*, 25(1), pp.25-27.
- Ellison, D., Morris, C.E., Locatelli, B., Sheil, D., Cohen, J., Murdiyarto, D., Gutierrez, V., et al., 2017. Trees, forests and water: Cool insights for a hot world. *Global Environmental Change*, 43, pp.51-61.
- Eriksson, L.O., Löfgren, S. and Öhman, K., 2011. Implications for forest management of the EU Water Framework Directive's stream water quality requirements—A modeling approach. *Forest Policy and Economics*, 13(4), pp.284-291.
- Ernst, C., Gullick, R. and Nixon, K., 2004. Conserving forests to protect water. *Am. Water W. Assoc.*, 30, pp.1-7.
- Falkner, I., Whiteway, T., Prezeslawski, R. and Heap, A.D., 2009. *Review of ten key ecological features (KEFs) in the northwest marine region*. Canberra: Geoscience Australia Record, Canberra, Australia, p.117.
- Fan, M., Li, Y. and Li, W., 2015. Solving one problem by creating a bigger one: The consequences of ecological resettlement for grassland restoration and poverty alleviation in Northwestern China. *Land Use Policy*, 42, pp.124-130.
- Ferraz, S.F., Ferraz, K.M., Cassiano, C.C., Brancalion, P.H.S., da Luz, D.T., Azevedo, T.N., Tambosi, L.R. and Metzger, J.P., 2014. How good are tropical forest patches for ecosystem services provisioning? *Landscape Ecology*, 29(2), pp.187-200.
- Fiquepron, J., Garcia, S. and Stenger, A., 2013. Land use impact on water quality: valuing forest services in terms of the water supply sector. *Journal of Environmental Management*, 126, pp.113-121.
- Fisher, B., Turner, R.K. and Morling, P., 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics*, 68(3), pp.643-653.
- Flannery, T., 1994. *The Future Eaters: An Ecological History of the Australasian Lands and People*. Port Melbourne: Reed Books.
- Fontana, V., Radtke, A., Fedrigotti, V.B., Tappeiner, U., Tasser, E., Zerbe, S. and Buchholz, T., 2013. Comparing land-use alternatives: Using the ecosystem services concept to define a multi-criteria decision analysis. *Ecological Economics*, 93, pp.128-136.
- Food and Agriculture Organization of the United Nations (FAO), 2008. *Forests and water. FAO Forestry paper 155*. Rome: FAO.
- Funke, N., Claassen, M. and Nienaber, S., 2013. Development and uptake of scenarios to support water resources planning, development and management: examples from South Africa. pp 1-27. In: *Water Resources Planning, Development and Management*, Wurbs R. (ed.). Rijeka: Intech publications.
- Garcia-Chevesich, P.A., Neary, D.G., Scott, D.F., Benyon, R.G. and Reyna, T. (eds.), 2017. *Forest Management and the impact on water resources : Forest management and the impact on water resources : a review of 13 countries. IHP - VIII / Technical document N° 37*. Montevideo: UNESCO.
- Garcia-Prats, A., del Campo, A.D. and Pulido-Velazquez, M., 2016. A hydroeconomic modeling framework for optimal integrated management of forest and water. *Water Resources Research*, 52(10), pp.8277-8294.
- Giles, L., 1910. *Translation of "The art of war" by Sun Tzu*. Portland, OR: The Puppet Press.
- Gómez-Baggethun, E., De Groot, R., Lomas, P.L. and Montes, C., 2010. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. *Ecological Economics*, 69(6), pp.1209-1218.
- Gregory, K.J., 2006. The human role in changing river channels. *Geomorphology*, 79(3-4), pp.172-191.
- Gyawali, D., 2004. Water, sanitation and human settlements: crisis, opportunity or management. *Water Nepal*, 11(2), pp.7-20.
- Gyawali, D. and Thompson, M., 2016. Restoring Development Dharma with Toad's Eye Science. *IDS Bulletin Special 50th Anniversary Issue*, 47(2A), pp. 179-190.
- Haines-Young, R. and Potschin, M., 2013. *Common International Classification of Ecosystem Services (CICES, Version 4.3)*. Nottingham: Report to the European Environment Agency.
- Hardin, G., 1968. The Tragedy of the Commons. *Science*, 162(3859), pp. 1243-1248.
- Harou, J.J., Pulido-Velazquez, M., Rosenberg, D.E., Medellín-Azuara, J., Lund, J.R. and Howitt, R.E., 2009. Hydro-economic models: Concepts, design, applications, and future prospects. *Journal of Hydrology*, 375(3-4), pp.627-643.
- Hein, L., Van Koppen, K., De Groot, R.S. and Van Ierland, E.C., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics*, 57(2), pp.209-228.
- Hulme, M., Dessai, S., Lorenzoni, I. and Nelson, D.R., 2009. Unstable climates: Exploring the statistical and social constructions of 'normal' climate. *Geoforum*, 40(2), pp.197-206.
- Jiang, Q. and Grafton, R.Q., 2012. Economic effects of climate change in the Murray–Darling Basin, Australia. *Agricultural Systems*, 110, pp.10-16.
- Keleş, S. and Başkent, E.Z., 2011. Joint production of timber and water: a case study. *Water Policy*, 13(4), pp.535-546.
- Kenter, J.O., O'Brien, L., Hockley, N., Ravenscroft, N., Fazey, I., Irvine, K.N., Reed, M.S., et al., 2015. What are shared and social values of ecosystems? *Ecological Economics*, 111, pp.86-99.
- Kosoy, N. and Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. *Ecological Economics*, 69(6), pp.1228-1236.
- Kramer, R.A., Richter, D.D., Pattanayak, S. and Sharma, N.P., 1997. Ecological and economic analysis of watershed protection in Eastern Madagascar. *Journal of Environmental Management*, 49(3), pp.277-295.
- Küçüker, D.M. and Baskent, E.Z., 2015. Evaluation of Forest Dynamics Focusing on Various Minimum Harvesting Ages in Multi-Purpose Forest Management Planning. *Forest Systems*, 24(1), pp. 1–10.
- Kumar, P., (ed.), 2010. *The economics of ecosystems and biodiversity TEEB: ecological and economic foundations*. London and Washington, DC: Earthscan.
- Landeta, J., 2006. Current validity of the Delphi method in social sciences. *Technological Forecasting and Social Change*, 73(5), pp.467-482.
- Langerwisch, F., Walz, A., Rammig, A., Tietjen, B., Thonicke, K. and Cramer, W., 2016. Deforestation in Amazonia impacts riverine carbon dynamics. *Earth System Dynamics*, 7(4), pp.953-968.
- Laudon, H., Sponseller, R.A., Lucas, R.W., Futter, M.N., Egnell, G., Bishop, K., Ågren, A., et al., 2011. Consequences of more intensive forestry for the sustainable management of forest soils and waters. *Forests*, 2(1), pp.243-260.
- Lawler, J.J., Lewis, D.J., Nelson, E., Plantinga, A.J., Polasky, S., Withey, J.C., Helmers, D.P., et al., 2014. Projected land-use change impacts on ecosystem services in the United States. *Proceedings of the National Academy of Sciences*, 111(20), pp.7492-7497.

- Lele, S., 2009. Watershed services of tropical forests: from hydrology to economic valuation to integrated analysis. *Current Opinion in Environmental Sustainability*, 1(2), pp.148-155.
- Loftus, A., 2015. Water (in) security: securing the right to water. *The Geographical Journal*, 181(4), pp.350-356.
- Lukes, S., 1974 (2005 second edition). *Power: A Radical View*. London: Palgrave Macmillan
- Malmer, A., Ardö, J., Scott D. Vignola R. and Xu, J., 2010. *Forest water and global water governance*. In: *Forests and Society—Responding to Global Drivers of Change*. Mery, G., Katila, P., Galloway, G., Alfaro, I., Kanninen, M., Lobovikov, M. and Varjo, J. (eds.). Vienna: IUFRO World Series No. 25.
- Malthus, T. R. 1888. *An essay on the principle of population*. London: Reeves and Turner.
- Martinez-Alier, J., 2002. *The Environmentalism of the Poor: A Study of Ecological Conflicts and Valuation*. Cheltenham, UK/ Northampton, MA: Edward Elgar.
- Martinez-Alier, J., Munda, G. and O'Neill, J., 1998. Weak comparability of values as a foundation for ecological economics. *Ecological Economics*, 26(3), pp.277-286.
- Martin-Ortega, J., Ferrier, R.C., Gordon, I.J. and Khan, S. (eds.), 2015. *Water ecosystem services: a global perspective*. Paris, France and Cambridge, UK: UNESCO and Cambridge University Press.
- Martin-Ortega, J., Glenk, K. and Byg, A., 2017. How to make complexity look simple? Conveying ecosystems restoration complexity for socio-economic research and public engagement. *PLoS one*, 12(7), p.e0181686.
- McKay, J.M., 2011. Australian water allocation plans and the sustainability objective—conflicts and conflict-resolution measures. *Hydrological Sciences Journal*, 56(4), pp.615-629.
- MDBA (Murray-Darling Basin Authority), 2011. *The proposed “environmentally sustainable level of take” for surface water of the Murray-Darling Basin: Method and Outcomes*. Canberra: Murray-Darling Basin Authority.
- MDBA (Murray-Darling Basin Authority), 2012. *Hydrologic modelling to inform the proposed Basin Plan: Methods and results*. Canberra: Murray-Darling Basin Authority.
- Meadows, D. H., Meadows, D. L., Randers, J., Randers, M. and Behrens III, W.W., 1972. *Limits to Growth*. New York: Universe Books.
- Mehmood-Ul-Hassan, M., van Noordwijk, M. and Namirembe, S., 2017. Partnering and capacity development with local stakeholders in ecosystem service management. In: *Co-investment in ecosystem services: global lessons from payment and incentive schemes*. Namirembe, S., Leimona, B., van Noordwijk, M. and Minang, P., (eds.) Nairobi: World Agroforestry Centre (ICRAF).
- Millar, C.I. and Stephenson, N.L., 2015. Temperate forest health in an era of emerging megadisturbance. *Science*, 349(6250), pp.823-826.
- Millennium Ecosystem Assessment (MEA), 2003. *Ecosystems and human well-being, a framework for assessment*. Washington, DC: Island Press.
- Millennium Ecosystem Assessment (MEA), 2005. *Ecosystems and human well-being: general synthesis*. Washington, DC: Island Press.
- Mollinga, P.P., 2008. Water, politics and development: Framing a political sociology of water resources management. *Water Alternatives*, 1(1), pp.7-23.
- Mooney, C. and Tan, P.L., 2012. South Australia's River Murray: social and cultural values in water planning. *Journal of Hydrology*, 474, pp.29-37.
- Mulligan, M., Benitez-Ponce, S., Lozano-V, J. S., and Sarmiento, J. L., 2015. Policy support systems for the development of benefit-sharing mechanisms for water-related ecosystem services. In: *Water Ecosystem Services: A Global Perspective*. Ferrier, R.C., Gordon, I.J. and Khan, S. (eds.) Cambridge University Press, Cambridge.
- Muradian, R., 2017. *The limits of the ecosystem services paradigm and the search for alternative ways of conceiving human-nature relations*. 12th Conference of the European Society of Ecological Economics, Budapest 20-23 June, 2017.
- Myers, N., 2002. Environmental refugees: a growing phenomenon of the 21st century. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 357(1420), pp.609-613.
- Nahlik, A.M., Kentula, M.E., Fennessy, M.S. and Landers, D.H., 2012. Where is the consensus? A proposed foundation for moving ecosystem service concepts into practice. *Ecological Economics*, 77, pp.27-35.
- Newbold, T., Hudson, L.N., Arnell, A.P., Contu, S., De Palma, A., Ferrier, S., Hill, S.L., et. al., 2016. Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. *Science*, 353(6296), pp.288-291.
- Norgaard, R.B., 2010. Ecosystem services: From eye-opening metaphor to complexity blinder. *Ecological Economics*, 69(6), pp.1219-1227.
- Nordblom, T.L., Finlayson, J.D. and Hume, I.H., 2012. Upstream demand for water use by new tree plantations imposes externalities on downstream irrigated agriculture and wetlands. *Australian Journal of Agricultural and Resource Economics*, 56(4), pp.455-474.
- Ojea, E. and Martin-Ortega, J., 2015. Understanding the economic value of water ecosystem services from tropical forests: A systematic review for South and Central America. *Journal of Forest Economics*, 21(2), pp.97-106.
- Ojea, E., Martin-Ortega, J. and Chiabai, A., 2012. Defining and classifying ecosystem services for economic valuation: the case of forest water services. *Environmental Science & Policy*, 19, pp.1-15.
- Ostrom, E., 2009. A general framework for analyzing sustainability of social-ecological systems. *Science*, 325(5939), pp.419-422.
- Ovando, P., Caparrós, A., Díaz-Balteiro, L., Pasalodos, M., Beguería, S., Oviedo, J.L., Montero, G. and Campos, P., 2017. Spatial Valuation of forests' environmental assets: an application to andalusian silvopastoral farms. *Land Economics*, 93(1), pp.87-108.
- Ovando, P., Beguería, S. and Campos, P., 2018. Carbon sequestration or water yield? The effect of payments for ecosystem services on forest management decisions in Mediterranean forests. *Water Resources and Economics*, In press.
- Pascual, U., Balvanera, P., Díaz, S., Pataki, G., Roth, E., Stenseke, M., Watson, R.T., Dessane, E.B., Islar, M., Kelemen, E. and Maris, V., 2017. Valuing nature's contributions to people: the IPBES approach. *Current Opinion in Environmental Sustainability*, 26, pp.7-16.
- Pearce, D.W. and Turner, R.K., 1990. *Economics of Natural Resources and the Environment*. Hemel Hempstead, UK: Harvester Wheatsheaf.
- Peterson, M.J., Hall, D.M., Feldpausch-Parker, A.M. and Peterson, T.R., 2010. Obscuring ecosystem function with application of the ecosystem services concept. *Conservation Biology*, 24(1), pp.113-119.
- Polanyi, K., 1944. *The Great Transformation: the Political and Economic Origins of Our Time*. Boston: Beacon Press.
- Qureshi, M.E., Shi, T., Qureshi, S.E. and Proctor, W., 2009. Removing barriers to facilitate efficient water markets in the Murray-Darling Basin of Australia. *Agricultural Water Management*, 96(11), pp.1641-1651.
- Rahayu, S., Widodo, R.H., van Noordwijk, M. Verbist, B. 2013a. Participatory water monitoring (PaWaMo). In: *Negotiation-support toolkit for learning landscapes*. van Noordwijk, M., Lusiana, B., Leimona, B., Dewi, S. Wulandari, D. (eds) Bogor: World Agroforestry Centre (ICRAF) Southeast Asia Regional Programme.
- Rahayu, S., Widodo, R.H., van Noordwijk, M., Suryadi, I., Verbist, B. 2013b. *Water monitoring in watersheds*. Bogor: World Agroforestry Centre (ICRAF) Southeast Asia Regional Programme.

- Raymond, C.M., Singh, G.G., Benessaiah, K., Bernhardt, J.R., Levine, J., Nelson, H., Turner, N.J., et al., 2013. Ecosystem services and beyond: Using multiple metaphors to understand human–environment relationships. *BioScience*, 63(7), pp.536-546.
- Ribot, J.C. and Peluso, N.L., 2003. A theory of access. *Rural Sociology*, 68(2), pp.153-181.
- Ricketts, T.H., Daily, G.C., Ehrlich, P.R. and Michener, C.D., 2004. Economic value of tropical forest to coffee production. *PNAS*, 101(34), pp.12579-12582.
- Schulz, C., Martin-Ortega, J., Glenk, K. and Ioris, A.A., 2017. The value base of water governance: A multi-disciplinary perspective. *Ecological Economics*, 131, pp.241-249.
- Seidl, R., Spies, T.A., Peterson, D.L., Stephens, S.L. and Hicke, J.A., 2016. Searching for resilience: addressing the impacts of changing disturbance regimes on forest ecosystem services. *Journal of Applied Ecology*, 53(1), pp.120-129.
- Siemion, J., Burns, D.A., Murdoch, P.S. and Germain, R.H., 2011. The relation of harvesting intensity to changes in soil, soil water, and stream chemistry in a northern hardwood forest, Catskill Mountains, USA. *Forest Ecology and Management*, 261(9), pp.1510-1519.
- Sikor, T., Martin, A., Fisher, J. and He, J., 2014. Toward an empirical analysis of justice in ecosystem governance. *Conservation Letters*, 7(6), pp.524-532.
- Simon, J. and Khan, H., (eds.), 1984. *The Resourceful Earth*. Oxford: Blackwell.
- Simonit, S., Connors, J.P., Yoo, J., Kinzig, A. and Perrings, C., 2015. The impact of forest thinning on the reliability of water supply in Central Arizona. *PLoS one*, 10(4), p.e0121596.
- Skuthorpe, T. and Sveiby, K.E., 2006. *Treading Lightly: The Hidden Wisdom of the World's Oldest People*. NSW: Allen and Unwin.
- Sullivan, C. A., 2002. *Using an income accounting framework to value non-timber forest products*. In: *Valuation Methodologies*. Pearce, D. (ed.). Cheltenham: Edward Elgar.
- Susaeta, A., D. Adams, Gonzalez-Benecke, C. and Soto, J., 2017. Economic Feasibility of Managing Loblolly Pine Forests for Water Production under Climate Change in the Southeastern United States. *Forests*, 8(3), p.83.
- Susaeta, A., Soto, J. R., Adams, D. and Allen, D. L., 2016. Economic Sustainability of Payments for Water Yield in Slash Pine Plantations in Florida. *Water*, 8(382), pp.1–16.
- Sweeney, B.W. and Newbold, J.D., 2014. Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: a literature review. *JAWRA Journal of the American Water Resources Association*, 50(3), pp.560-584.
- Tan, P.L., Bowmer, K.H. and Baldwin, C., 2012. Continued challenges in the policy and legal framework for collaborative water planning. *Journal of Hydrology*, 474, pp.84-91.
- Tomich, T.P., Lewis, J., Timmer, D. (eds.). 2004. *Participatory development of methods that local groups can use to monitor and interpret changes in their environment can empower communities to manage their natural resources more effectively*. *ASB Policy Brief 7*. Nairobi: World Agroforestry Centre.
- UK NEA, [UK National Ecosystem Assessment], 2011. *The UK National Ecosystem Assessment: synthesis of the key findings*. Cambridge: UNEP-WCMC.
- van Noordwijk, M., Kim, Y.S., Leimona, B., Hairiah, K. and Fisher, L.A., 2016. Metrics of water security, adaptive capacity, and agroforestry in Indonesia. *Current Opinion in Environmental Sustainability*, 21, pp.1-8.
- Vincent, J.R., Ahmad, I., Adnan, N., Burwell, W.B., Pattanayak, S.K., Tan-Soo, J.S. and Thomas, K., 2016. Valuing water purification by forests: an analysis of Malaysian panel data. *Environmental and Resource Economics*, 64(1), pp.59-80.
- Vira, B., Wildburger, C and Mansourian, S., 2015. *Forests, Trees and Landscapes for Food Security and Nutrition. A Global Assessment*. Vienna: IUFRO.
- Wang-Erlandsson, L., Fetzer, I., Keys, P.W., van der Ent, R.J., Savenije, H.H.G., Gordon, L.J., 2017. Remote land use impacts on river flows through atmospheric teleconnections. *Hydrology and Earth System Sciences*, pp.1–17.
- Wenny, D.G., Devault, T.L., Johnson, M.D., Kelly, D., Sekercioglu, C.H., Tomback, D.F. and Whelan, C.J., 2011. The need to quantify ecosystem services provided by birds. *The Auk*, 128(1), pp.1-14.
- Wilkinson, A. and Kupers, R., 2013. Living in the futures. *Harvard Business Review*, 91(5), pp.118-127.
- Xu, J. and Grumbine, R.E., 2014a. Integrating local hybrid knowledge and state support for climate change adaptation in the Asian Highlands. *Climatic Change*, 124(1-2), pp.93-104.
- Xu, J. and Grumbine, R.E., 2014b. Building ecosystem resilience for climate change adaptation in the Asian highlands. *Wiley Interdisciplinary Reviews: Climate Change*, 5(6), pp.709-718.
- Zalasiewicz, J., Williams, M., Steffen, W. and Crutzen, P., 2010. The new world of the anthropocene. *Environmental Science and Technology*, 44(7), pp.2228–2231.
- Zieba, F.W., Yengoh, G.T. and Tom, A., 2017. Seasonal Migration and Settlement around Lake Chad: Strategies for Control of Resources in an Increasingly Drying Lake. *Resources*, 6(3), p.41.