Chapter 6
Management Options for Dealing with Changing Forest-Water Relations

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6.1 Introduction

This chapter addresses potential forest and water management strategies based on our understanding of the 'new normal', the challenges imposed, in particular, by climate change and human population growth, and our evolving knowledge of forest-water interactions. It further considers forest and water management strategies when water is prioritised over other forest-related goals (such as biomass accumulation or the sequestration of carbon in standing forests). Explicitly prioritising water in forest management attempts to reset our priorities toward more sustainable strategies for long-term forest health and human welfare. This reordering of priorities does not necessarily compromise other forest-related goals but provides a much-needed emphasis on water as a key contribution to both planetary and human health.

Forests have long been considered a valued natural resource. Timber, wild game, fuelwood, recreation and more recently carbon sequestration are all products associated with forests. Clean, abundant water is an ecosystem service provided by forests. Depending on the location, meteorological conditions, size of the forest and time of year, forest water may be flowing, stagnant, a trickling seep, a clear running or silt laden brook or a cascading river. However, some form of flowing water from these ecosystems seems as natural as the trees that surround them for good reason. Leaf litter, tree roots and animal burrowing allow a high level of soil permeability for precipitation. Once water enters the forest floor, high concentrations of organic matter retain the moisture for plant use. Water in excess of soil storage capacity is slowly drained through the soil toward lower elevations that converge to form brooks and streams, rivers and potentially aquifers. Hydrologic studies have found that once saturated, forest soils can provide a constant supply of water for over four months after the soil profile was sealed and no additional precipitation was added to the column (Hewlett and Hibbert, 1963).

Water is very seldom considered first in forest management perhaps because the co-occurrence of forest and water are so common. However, as global climate air temperatures and climate variability continue to increase, the relationship between forests and water flow may be changing. Studies have shown that incoming precipitation is first used by vegetation with the excess used to then saturate the soil column (Sun et al., 2011). Only after these two conditions are met does water then begin to drain from the forest ecosystem as streamflow (Sun et al., 2011; Caldwell et al., 2015). As air temperature increases, so does potential evaporation. Therefore, if precipitation is constant, and air temperature rising, evapotranspiration will increase while ground water and streamflow will decrease (Caldwell et al., 2015). Furthermore, if changing climatic patterns reduce precipitation, streamflow may be even further reduced compared to historic conditions. However, some reductions may be moderated if forest mortality reduces plant water demand, but the evidence for this impact is uncertain (Biederman et al., 2015).

In addition to changing climate, global population increases and a demographic shift towards equatorial regions are further stressing historic water supplies. The time has come to begin considering some forests primarily for their water resource value instead of a by-product of some other natural resource objective. Considering forests first and foremost for water, is not a simple task. Trade-offs between tree water use to maintain forest structure and function (including soil permeability), while maximising water flow during critical times of need is a complex issue. Meeting annual water volume demands is of little use if the majority of the water comes during a period of reduced resource need (e.g., winter months). Forest managers and owners might have to change their management objectives and consider some of their forests primarily for their ability to supply water for both environmental stability and anthropogenic use.

There are important considerations of scale, management levels and responsibilities which affect decision making for both forests and water. Forest management decisions are usually made by very diverse landowners, forest authorities, leaseholders, communities and organisations at local scales (often the stand or management unit, or property), while public authorities are often primarily responsible for the delivery of water resources, typically operating at catchment, landscape, watershed or precipitationshed levels. Forest managers, working at more micro scales, might not integrate objectives for water quantity or quality into their management decision systems, and forest management practices might be very diversified at catchment level. This chapter builds the case for greater harmonisation across these scales, management units and the integration of private and public responsibilities for the delivery of improved strategies for managing forest-water interactions.

Section 6.2 takes a more traditional status quo understanding of the interactions between forests and water and focuses on the catchment as the typical unit of analysis, primarily targeting up- and downstream hydrologic flows. Section 6.3 then adopts a much larger multi-basin (precipitationshed) perspective and considers the ways in which forests and water contribute to up- and downwind dynamics in precipitation and subsequent impacts on hydrologic flows. If forests use water from the basin perspective, from the larger regional and continental scale perspective, they are dynamic contributors to the hydrologic cycle, rainfall and the availability of water. Both of these contrasting scale perspectives yield important potential forest management strategies that ultimately need to be considered in concert. Section 6.4 considers the social and institutional responses, typically at catchment scales, outlining a range of ways that mutual interdependence of stakeholders across landscapes can be mobilised to better manage forest-water interactions. Section 6.5 develops these institutional mechanisms further, with a specific focus on incentive- and reward-based mechanisms for managing interdependence and reciprocity in forest-water systems. Section 6.6 looks forward to a more integrated, water-sensitive approach to forest management, focusing on the identification of critical water zones, and

mechanisms for the management of reciprocity across key stakeholders. The chapter concludes with a brief discussion of research and data needs and knowledge gaps.

6.2 Management at Catchment Scale

As the scale and intensity of forest management increase so does the impact of humans on the natural ecosystem (Keenan and Kimmins, 1993; Sullivan and O'Keeffe, 2011). There is a wide range of forest management options at the catchment scale but seldom are practices conducted across the entire catchment. Lessons learned from large-scale clear-cutting in Canada (Buttle et al., 2005), the United States (USDA, 2001), Australia (Bradshaw, 2012) and Indonesia (Tsujino et al., 2016) demonstrated the ecosystem degradation of these practices. Although reduced, catchment scale clear-cut harvesting still occurs in parts of the world with continued high levels of land degradation (Asner et al., 2006).

There are many degrees of forest management ranging from passive or low to intensive (Duncker et al., 2012). The level of forest management is a function of both biogeographical conditions and societal demands (Duncker et al., 2012). Although often not considered as such, the decision to do no management (passive) is actually a form of management in which natural forces (e.g., disturbance, growth and regeneration) dominate the future direction of forest structure and function in catchments. National parks and other protected areas are often managed in this way. All other forms of forest management fall between clear-cutting and no management. Management practices range from selective cutting, to group cutting (in which groups of trees are removed to promote early successional tree species regeneration).

Management approaches depend on the objectives for the catchment. In catchments where timber production is a priority, all factors that would reduce growth or increase forest mortality are minimised. Examples of such activities would include the removal or control of insect pests and disease to prevent the spread to healthy trees. Increased timber, pulpwood and fuel productivity may cause reduced streamflows. With some exceptions such as cloud forests where fog condenses on leaves and is a significant contributor to the total hydrologic budget (Marzol-Jaén, 2010), as forest productivity increases, so does forest water use.

As described in Chapters 2 and 3, leaf area index (LAI) is a common term used to predict both forest water use and forest productivity. Management practices that reduce or increase LAI also increase or reduce catchment annual water yield. Controlled burning may be used to reduce the growth of non-commercial woody and herbaceous living and dead material, reducing LAI, and therefore possibly increasing forest annual water yield (Hallema et al., 2017). Forest thinning and eventually harvesting for income generation or wood use also increases annual water yield at the catchment scale (Hibbert, 1965; Downing, 2015; Yurtseven et al., 2017).

Plantation forestry is the most intensive form of forest management and represents approximately 7% of the total forest area (Payn et al., 2015). Forest plantations are almost always planted in rows to optimise tree growth and harvesting, and therefore increase LAI, and as a result decrease forest water yield (Brown et al., 2005). Additionally, the majority of plantations are rapidly growing monocultures of exotic species with less biodiversity compared to natural stands (Brockerhoff et al., 2008). This type of forestry can increase water demands by the trees (Scott et al., 2004) as well as increase the risk of episodic insect and disease outbreaks, or fire that can threaten the health of the entire stand (Mitchell et. al., 1983; McNulty et al., 2014). While complete stand or catchment mortality can significantly increase streamflows, tree mortality may also decrease water quality (Hibbert, 1965; Swank et al., 2001).

Eucalyptus plantation and indigenous forest in South Africa
Photo © Mark Gush

Aside from production forestry, there are **other objectives for forest management** such as recreation, biodiversity, cultural heritage, specialty crops; each of these practices has hydrologic impacts. For example, controlled burning is used to minimise forest ground cover and reduce wildfire risk (Outcalt and Wade, 2004). This also may increase soil nutrients for trees (DeBell and Ralston, 1970; White et al., 2008), reduce soil water competition (Haase, 1986) and promote tree seedling regeneration (Sackett, 1984). On shallow slopes, controlled burning has a negligible impact on stream water quality (Vose et al., 2005). However, both controlled burning and wildfires can have negative impacts on stream water quality on forests located on steeper slopes (Wright et al., 1976; LaPoiat, 1983; Pierson et al., 2002). Other mitigation measures such as soil bunding and brush barriers can be used to reduce the amount of soil that reaches the stream. Soil bunding has the effect of slowing down the rate of runoff from the forest floor, while brush barriers are often constructed of tree branches and other smaller debris that is a by-product of the cutting operation (McNulty and Sun, 1998). This material is placed on the down

slope side of areas susceptible to erosion (e.g., denuded soils on steep slopes). As overland flow runs off of the exposed soil surface, sediment is trapped in the brush while water passes through to the stream. Brush barriers are effective in capturing coarse sand, but finer material (e.g., clays and silts) remain suspended in the flow.

Increasing biodiversity may require various forms of forest management. For example, many mammals (e.g., deer) and birds (e.g., hawks) prefer recently cut stands (Hunter and Schmiegelow, 2011). The regenerating seedlings after a cut provide a ready food source for herbivores. Mice and other small animals that are drawn to these openings then become potential prey for predator species (e.g., owls and hawks). If the objective is to maximise species that inhabit newly cut areas, then the forest plan should be to routinely harvest patches of forest to maintain these openings. As trees are cut, and the LAI is reduced, water yield increases (Bosch and Hewlett, 1982). Conversely, if the objective is to increase animal species (e.g., bear and turkey) which prefer old growth forest, then little or no cutting of the forest is required. In this case, LAI and forest evapotranspiration would be higher while streamflow would be lower compared to the cut stands. Between the two extremes of total cut and no cut, lie many other forest management options (e.g., shelter-wood cut, individual tree cut, seed-tree cut) with intermediate impacts on forest hydrology.

Similarly, the **maintenance of culturally or historically important areas** requires forest management. Although now heavily forested, much of the northeastern US was cleared for agriculture in the 18th and 19th centuries, and the southeast mountains were cleared for farms until the 20th century (Yarnell, 1998). Most of these areas have reverted back to forest cover, but some areas are retained as farms. Conversely, in many countries old growth or virgin forests have cultural significance so they are less likely to be harvested. As with biodiversity objectives, the impact on water quality and quantity will be dependent on the degree of cutting needed to maintain the cultural or historical objective.

Riparian vegetation is an important factor influencing the aquatic environment. It plays an important role in the prevention of nutrient and sediment pollution, the stabilisation of river banks and fish habitat, the perpetuation of the microbial food loop, and the control of flooding (Dosskey et al., 2010). The importance of the **protection of riparian ecosystems** may depend on the size of streams, topography and existing disturbance regimes (Likens and Bormann, 1974; Hughes et al., 1986). As such, riparian zone protection and management often include the identification and establishment of a vegetated buffer-strip of a prescribed width which is incorporated as an important component of watershed management strategies. However, in many areas, the current management strategy may apply a constant size of buffer-strip, which may not effectively serve its purpose of stream water protection (Boggs et al., 2015; Boggs et al., 2016). As an alternative, an approach incorporating variable buffer-strip widths depending on local conditions has been proposed (Belt et al., 1992).

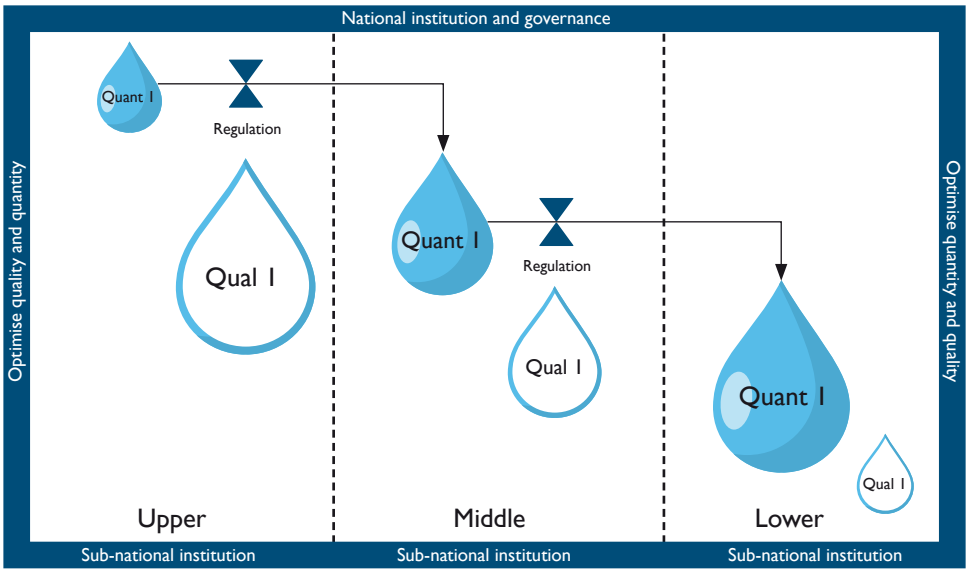
The timing of water flow is important to proper aquatic zone structure and function. Increases in annual water flow may not have beneficial impacts on aquatic populations if there is a reduction in seasonal water flow despite an overall annual increase. For example, protection of salmon populations in British Columbia (Canada) requires consideration of the magnitude, timing, frequency, duration and variability of flow reqimes (Poff and Zimmerman, 2010; Zhang et al., 2016). Consideration of the factors influencing streamflow is further complicated as climate change and other anthropogenic stresses are increasingly impacting efforts to maintain and restore aquatic ecosystems (Ukkola et al., 2015; Hjalten et al., 2016).

Riparian vegetation and landscape in Mongolia
Photo © Alexander Buck

Catchment water can be derived from within the catchment through precipitation or originate outside the catchment as an inflow. Therefore, regulation of catchment water quality and quantity requires environmental regulation. Options for such regulation must be openly planned and discussed with all the relevant land, forest and water stakeholders, and must take account of prevailing legislation. This is particularly relevant when infrastructure to regulate environmental flows is being put in place. There is no 'one size fits all' in the context of biophysical conditions and socio-economic-cultural settings, and many approaches have been designed to identify the level of environmental flow requirements (Tharme, 2003). The extent to which environmental flows have been implemented in different countries varies widely. Some countries, including parts of the US, Australia, New Zealand and countries of the European Union (Acreman and Ferguson, 2010) together with South Africa, have accepted the need to develop and implement catchment water resource plans that include environmental flows. Also, in these countries where environmental flows have now been mainstreamed into water resource planning, there is an acceptance that the concept of environmental flows should be extended to groundwater as well as to estuaries and even near-shore regions; this can have potential future implications for management of floodplain forests or coastal forests.

Cascading water regimes that demonstrate a common feature of decreasing quality with increasing quantity as environmental flows should be governed and dynamically optimised across the jurisdiction at various levels

Figure 6.1



Source: Daniel Murdiyarso (author's own elaboration)

Figure 6.1 shows that the governance system plays a key role in regulating the water regime to ensure optimised water quality and quantity encompassing upper, middle and lower catchment areas. The extent to which an increase in water quantity within a catchment affects water quality depends on the nature of flows, sediment transport and pollutants within the system. While an increase in the extent and speed of surface flows is likely to increase sediment loads, negatively impacting water quality, an increase in the volume of water is likely to dilute pollutants and nutrients within the system, but necessarily improve water quality if total nutrient load increases. The relative balance of these two effects is likely to be very context specific, but there is an important need for institutions across this gradient to be aware that there are both quantitative and qualitative effects to be considered while determining an appropriate response at each scale of intervention, while also being aware that these impacts have a cascading effect down the catchment.

All forest management strategies, however well designed, have to contend with some well-known challenges and problems associated with the delivery of well-intentioned interventions which can constrain their overall effectiveness. These include:

Technical and capacity problems: Lack of trained local personnel with skills in forest maintenance and management, poor understanding of species' viability in differing conditions and inadequate number and poor quality of seedlings hamper effective forest management. There is also a poor understanding of the long-term impact of exotic species (Little et al., 2009) and the need for improved equipment design, especially for small scale operations.

Economic problems: Lack of capital to cover start-up costs, labour shortages in suitable planting areas and poor understanding of opportunity costs of forest operations and income potentials can be a challenge. The high cost of planting material, transport and heavy equipment costs, and long time periods before returns are realised influence management practices. There is a real need for better operational data measurement techniques to support financial decisions (Rönnqvist et al., 2015).

Social and institutional problems: In areas where reforestation is potentially viable, there may be problems of trade-offs and conflicts between agricultural and forestry activities. Variable definitions of forest cover create data disharmony, and there are often problems with clarity over jurisdictional responsibilities, especially in agroforestry contexts (Mentis, 2015). Land tenure restrictions, particularly on tenanted or leasehold land, can act as a barrier to tree planting, and there is some reluctance to take up new techniques and innovations. If increases in forest cover are to be achieved at a pace appropriate to achieving specific Sustainable Development Goals (and other associated global commitments, such as under the Convention on Biological Diversity's Aichi targets and the Bonn Challenge on Forest Landscape Restoration), there is a need for these challenges to be overcome. To this end, donor agencies and national governments need to work towards a more streamlined and integrated approach to forest operations, and to recognise the political economy context in which interventions are implemented.

6.3 New Management Options in the Context of the New Normal

Under the 'new normal', water storage and timing distribution are changing. For example, southern California relies on snow melt from the Sierra Mountains for potable water, but due to changes in winter weather patterns, the snowpack has been more variable. The spring 2017 snowpack was the largest in 19 years while the previous years were some of the smallest (NASA, 2017). Combined with an ongoing drought, this unpredictability of the water supply forces managers to prepare for the worst possible scenario to assure that vital water needs are being met. However, 'new normal' water regulations must also be flexible to allow for removal of restrictions when annual water flows provide sufficient water to optimise productivity (Nagourney and Lovett, 2016). Flexibility in water management regulations is likely to be more effective than large scale engineering projects designed to transport water from one basin to another due to cost and the shifting nature of climatic patterns under the 'new normal'.

The complexity of forest-water interactions defy broad generalisation and therefore it is important to approach the water dimensions of forest management in an adaptive framework particularly in the context of the 'new normal' (Pahl-Wostl et al., 2007). Decisions must be made continuously, but the more the outcomes of forest management choices can be monitored and evaluated, the more likely better choices will be made now and also in the future when forest ecosystem services are likely to be in even greater demand. Box 6.1 illustrates the risks of simplistic management responses based on unfounded assumptions about eco-hydrological processes and the social and behavioural contexts in which people make decisions, which have led to almost two decades of misguided interventions in the Himalayas.

A focus on catchment - level interactions between forests and water does not recognise the potential for water to be both imported into the catchment in the form of atmospheric moisture, a very large component of which is produced by upwind evapotranspiration, and also exported downwind in the form of evapotranspiration. Though the general paired-catchment basin literature clearly highlights the atmospheric moisture production of forests (Bosch and Hewlett, 1982; Jackson et al., 2005; Filoso et al., 2017; Zhang et al., 2017), this literature typically neglects to provide any explanation of what happens to the water which is evapo-transpired from within the basin and, to the extent to which it is, or is not, recycled locally, and does or does not contribute to local, within-basin streamflow or groundwater recharge. This water is currently unaccounted for in the water balance. But it is clearly exported from the basin as atmospheric moisture and thus has relevance for downwind, receiving basins, ecosystems and communities. Only when we move beyond the catchment to consider genuine water provisioning relationships at the landscape scale is it possible to understand the full impact of forests on water availability.

Up- and downwind forest-water relationships are likely to matter for the cross-continental supply of atmospheric moisture, and thus for the supply of available rainfall at the individual catchment basin level, even far from the basins where specific management actions are taken (Van der Ent et al., 2010; Keys et al., 2016; Ellison et al., 2012, 2017; Wang-Erlandsson et al., 2017). Since continental- and landscape-scale land use practices

Box 6.1

Larger landscape effects of forests and water – The Theory of Himalayan Environmental Degradation

The Theory of Himalayan Environmental Degradation (THED) was propounded at the UN Stockholm Environment Conference in 1972 where a single definition of the problem of flooding in the lower Ganga plains including Bangladesh was provided: it was increasing population pressure leading to growing numbers of ignorant mountain peasants cutting trees in the higher reaches that led to heavy sedimentation of rivers resulting in flooding (essence captured in Eckholm, 1976). Based on this discourse, development agencies such as DfID (then called Overseas Development Agency or ODA) and the World Bank predicted in the late 1970s that no accessible forest would remain in Nepal by 2000 (Thompson and Gyawali, 2007). This catastrophic alarmism had serious policy consequences which led to: governments banning access to forests for the poorest and marginalised in their countries leading to increased poverty; lower riparian communities finger-pointing to bad management by residents in the upper riparian zones and more generally, was used as justification for intervention and resource misallocation in solving the wrong problem (Ives and Messerli, 1989; Thompson and Gyawali, 2007).

It took the 1986 Mohonk conference in upstate New York to debunk THED (Ives, 1987; Ives et al., 1987). Conference scientists showed how wildly varying assumptions behind the deforestation argument by very venerable organisations led to predictions of impending catastrophes that never happened. Essentially what was proven was that, while the Himalayas were facing severe development and environmental challenges, a growing peasant population cutting trees was not the reason for flooding in the downstream plains. Rather, while bad land management practices and deforestation in places (mostly by powerful commercial interests) led to soil erosion and land productivity decline, unstable Himalayan geology and powerful cloudbursts therein led to mass wasting and bedload movement at a scale much greater than anything the peasants could do. Since then a series of new research have highlighted the real (and powerful) drivers behind underdevelopment and deterioration in the Himalayas as well as policies that have had positive outcomes. For instance, thanks to the egalitarian style of managing the commons coming into play (where hierarchism had failed and individualism had led to complete open access and degradation), community forestry has managed to put more land under forest cover than ever before in Nepal (Ojha, 2017; Pandit, 2017).

matter for the production of atmospheric moisture (and thus the recycling of precipitation back to the atmosphere and across terrestrial surfaces), large scale land use practices represent potentially important tools in the basket of options available to water and land use planners and managers. The current ability of water or forest management institutions to influence land cover at a very large scale may however be limited. Furthermore, how much weight is placed on the production of atmospheric moisture depends on the local impacts of producing that moisture and the downwind influences of that moisture. The degree of certainty with which these impacts can be predicted, both locally and in the precipitationshed needs to be considered, even before the issue of frameworks for decision making are addressed. Nevertheless, recognition of the importance of such interactions does suggest that the emphasis for water management today must go beyond catchment boundaries.

Up- and downwind forest-water relationships can thus theoretically be mobilised as a resource for improving the availability of potentially scarce water resources across continental surfaces. As such, the forest management strategies described in this section can be deployed in combination with, or in lieu of, the methodologies described above in Section 6.2, in particular because they focus on ways to increase the supply of available atmospheric moisture across terrestrial surfaces, with the explicit goal of influencing and thereby improving water availability toward continental interiors (Sheil and Murdiyarso, 2009; Millán, 2012; Layton and Ellison, 2016; Syktus and McAlpine, 2016). While forest and water managers may be accustomed to the up- and downstream management of forest and water interactions, the observation that these managers can also manipulate up- and downwind forest-water interactions is comparatively new, and requires both a conceptual framework for thinking about the up- and downwind, supply-side role of forests and water (Ellison et al., 2012, 2017), as well as a relatively simplified modelling framework with which forest and water managers can begin to put such models into effect. The modelling framework currently available, however, is complex (see in particular Keys et al., 2012, 2014; Wang-Erlandsson et al., 2017), and thus not one that forest and water managers are likely to be able to easily put to use on the ground.

The management approaches proposed in Section 6.2 are generally based on given quantities of water entering the catchment system and then adjusting for the changing circumstances. To illustrate, we can consider the possible response of irrigation, drainage and forest operations to climate change impacts where rising temperatures and declining rainfall may lead foresters to increase tree thinning activities for the purpose of reducing evapotranspiration and raising streamflow. While this represents an entirely viable strategy for increasing watershed streamflow (Swank et al., 2001), it is important to recognise that such a strategy may have significant impacts when implemented over large land areas and iterated across up- and downwind catchments (Nobre, 2014; Lawrence and Vandecar, 2015; Spracklen and Garcia-Carreras, 2015; Debortoli et al., 2017). If undertaken as a response to declining amounts of available water, the removal and/or thinning of forest cover in coastal and other upwind forests may lead to increasingly smaller amounts of water being transported across continental land-masses. In such a case, unintended and potentially disruptive consequences may result in continental interiors (Sheil and Murdiyarso, 2009; Lawrence and Vandecar, 2015; Keys et al., 2016; Nobre, 2014).

There is remarkably little literature available to assist interested individuals, groups, organisations and even forest owners in deciding when and where best to plant additional forest (Mansourian et al., 2005; Stanturf et al., 2012; Millán, 2012; Laestadius et al., 2014). While across the world, much effort is being made to reforest large areas and to promote agroforestry, little focus is placed on the important role upwind forests can play in contributing to the catchment water balance. Today, most efforts at increasing tree cover (afforestation, reforestation, restoration, hereafter referred to with the generic term 'forestation') typically focus on carbon sequestration, flood mitigation, improved water quality, or on the provision of other use values to support livelihoods and poverty alleviation, through the production of timber, bioenergy, recreation, fuelwood, etc. (Ciais et al., 2013; Hecht et al., 2016). Rarely is any focus given to forests as water providers, or the potential redistributive effects this might have in downwind locations.

Building upon the broad implications of the supply-side literature (Ellison et al., 2012, 2017; Key et al., 2016; Wang-Erlandsson et al., 2017), the range of possible management approaches to increase tree cover in the context of sustainable water yield includes (but may not be limited to) the following:

1) **Forestation to minimise trade-offs and build upon potential positive synergies**. Adding forest and vegetation cover, for example, to upwind coasts where evapotranspiration is likely to deliver water to potentially drier inland areas represents one possible win-win strategy. Where forests and vegetation cover do not compete significantly with other downstream uses, and in particular where large amounts of water flow unused into oceans, the production of additional atmospheric moisture should generally be considered an advantage for potential downwind terrestrial water users (Makarieva et al., 2006; Ellison et al., 2012, 2017; Millán, 2012; Layton and Ellison, 2016). Forestation of coastal zones may also provide water quality benefits and help protect fragile coastal ecosystems.

2) **Forestation in locations where the water supply is relatively abundant**. Regions that have been deforested in the past and are now prone to flooding (e.g., the Nadi catchment in Fiji), represent locations that are highly suited to the increased planting of forests. The resultant increase in evapotranspiration in these regions actually represents a benefit as opposed to a loss, as atmospheric moisture transfer reduces the risk of soil saturation and surface flooding (Jongman et al., 2015; van Noordwijk et al., 2016). Assuming that

the respective downwind locations which are likely to receive the additional atmospheric moisture and potential rainfall can benefit from this through increased water provision for agriculture, for example, this once again represents a win-win situation (Millán, 2012; van Noordwijk et al., 2016; Ellison et al., 2017).

3) **Trade-offs between runoff and evapotranspiration.** There are many situations in which some trade-off between runoff and increased evapotranspiration is entirely acceptable, though this is clearly not the case in all catchments. For basins where moderate trade-offs are acceptable, forestation can potentially be viewed as an acceptable and possibly advantageous strategy, not only in terms of real economic benefits to local communities (additional harvest, improved water quality and other forest-related benefits), but also for downwind communities who would benefit from the increased water resources that might become available through the additional atmospheric moisture transport (Millán, 2012; Ellison et al., 2017).

4) **Protect and restore water towers in high altitude, montane and cloud forest regions** (Viviroli and Weingartner, 2004; Viviroli et al., 2011). These forests directly extract moisture out of the atmosphere. Since cloud cover is likely to simply move on to other locations in regions where these forests have been significantly depleted through deforestation, there are likely to be significant returns to restoration in such locations. Moreover, many montane and cloud forests contribute disproportionately to downstream runoff (Ghazoul and Sheil, 2010; Bruijnzeel et al., 2011; Ramirez et al., 2017). Thus, restoring high altitude tree and forest cover may significantly improve infiltration and runoff, while helping to reduce outcomes like erosion and sedimentation, as well as downstream flooding. Moreover, since many of these mountain forest ecosystems are migrating upwards in elevation due to climate change, additional forestation efforts could help facilitate this process.

5) **Establish thresholds for forest and tree cover removal from terrestrial surfaces**. As suggested in particular by Ilstedt et al. (2016), there is some as yet not clearly defined level of 'optimal tree cover' that maximises groundwater recharge, while minimising the potential for producing evapotranspiration. The consequences of entirely removing tree and forest cover in order to encourage improved runoff is likely to have the downside effect of degrading soils, increasing the likelihood of flash flooding, otherwise increasing runoff and eliminating or greatly reducing the potential for groundwater recharge. If contextually appropriate thresholds can be adequately determined, coupled with a consideration of the impacts of different tree species on the optimal recharge-evapotranspiration balance this could provide a useful foundation for action to be taken towards achievement of both SDG 6 (on water) and SDG 15 (on terrestrial ecosystems) (see Box 6.2 for an illustration of this from the Himalayas).

Box 6.2

Impact of forest type on spring water quality and quantity

Springs are groundwater discharge points in the mountains where a water bearing layer (aquifer) intersects with the ground surface and water seeps out of rock pores, fissures, fractures or depressions. The traditional view that tree roots, leaf litter and soil act as a sponge and facilitate greater infiltration of water than bare surfaces (Bruijnzeel, 2004) meant that the majority of existing literature on springs attributed drying of springs to deforestation or degradation of forest cover (Valdiya and Bartarya, 1991; Negi and Joshi, 2004; Joshi et al., 2014). That the type, quality and nature of forest cover and discharge of springs and its water quality are co-related is also a belief that is widely held by local communities in the mid-hills of the Himalayas (Joshi and Negi, 2011; Rautela, 2015; Pandey et al., 2018). Many of these arguments, namely, whether or not having a good forest cover leads to better infiltration and recharge and therefore, higher spring discharge and what species of trees are most conducive for recharge have been so far made using perceptions of local communities (Joshi and Negi, 2011; Rautela, 2015) and expert judgement of authors (Sheikh and Kumar, 2010; Joshi et al., 2014; Naudiyal and Schmerbeck, 2017). It is only in recent years that studies based on experimental and modelling data have been used to support these claims, but such scientific studies are still too few in number (some of this evidence can be found in Birch et al. (2014) and Joshi and Kothyari (2003)).

To the best of our knowledge, papers by Ghimire et al. (2012; 2013a and 2013b and 2014), focusing on Nepal, are the only ones that use long term experimental data to look at hydrological impacts of natural broadleaved forests and mature planted pine forest. The main conclusion of their work is that, it is not enough to reforest a degraded forest and expect that hydrological functions be restored. In reality, the species planted, its water interception rates and ongoing forest management practices are just as important a determinant of restoration of hydrological function, as the act of reforestation itself (Amazonas et al., 2018).

6) **Adapt forest management practices to meet the challenges of the 'new normal'.** There are important forest management opportunities in places where climate change is causing increases in rainfall (along with warming temperatures). For example, in the Boreal region, climate change is expected to bring new opportunities for additional forest cover, at little or no impact to downstream communities or existing levels of water consumption (Kellomäki et al., 2008; Lindner et al., 2010). In fact, to the contrary, additional forest cover may provide important positive features, such as the ability to remove additional moisture from the landscape and possibly moderate the otherwise increased likelihood of flooding. It is important to remember though, that even within a region, there are areas that do not follow the regional trends. And indeed, within the boreal zone there are areas where climate ensembles predict less runoff even without changing land cover (see e.g., Arheimer and Lindström, 2015).

7) **Assess site-specific circumstances**. Finally, it is necessary to be attentive to the specific features of individual locations and to assess site-specific circumstances. For example, where the orographic setting is optimal, mountains may keep much of the evapotranspired moisture comparatively close to the basin in which it was produced, resulting in potentially much higher local precipitation recycling ratios than are ordinarily found. Thus, in such locations (see e.g., the discussion of the Los Angeles basin area in Layton and Ellison (2016) or the discussion of a Mediterranean example in Millan et al. (2005), forestation may have higher returns to the local community and ecosystems than in locations where almost all of the evapotranspiration produced will immediately be taken away by prevailing winds.

All of the above proposed forest management strategies essentially suggest that forestation may be used in ways that generally can minimise trade-offs, while having the potential to increase the production of atmospheric moisture, thereby providing additional moisture for rainfall in downwind locations (Ellison et al., 2017).

There is concern expressed from the demand-side literature that additional forest cover can have a negative impact on the water balance, in particular in basins that may, already, be water-stressed. Thus, for example, Bennett and Barton (2018) write: "There is a real potential that, if applied too broadly, the supply-side perspective could be used to justify tree-planting in areas with limited water supply." The supply-side literature, however, recognises such likelihoods as real concerns. Additional forest cover will almost never improve the water balance in the same basin in which it is planted, though it is likely to have a positive impact on the water balance in other, downwind, locations.

There is a further range of concerns that must also be considered when removing or adding additional tree and forest cover. The potential benefits of forests for achieving the additional cooling of terrestrial surfaces has long received inadequate attention. And the forest albedo debate, in particular, helped slow acknowledgement of the cooling potential of forest and tree cover. Awareness, however, that trees can have a net positive impact on surface cooling has been supported by more adequate recognition of the role of the water cycle and evapotranspiration in the cooling process (Pokorný et al., 2010; Hesslerová et al., 2013; Bonan, 2016; Bright et al., 2017).

Thus, the more recent wave of research providing a more holistic view of the impact of tree and forest cover on surface cooling has largely concluded that there is significant potential for additional tree and forest cover in most locations throughout the world. Others have highlighted the relative importance even of lower density tree cover for surface cooling in urban and city landscapes (Bounoua et al., 2015).

6.4 Socio-Institutional Options at Micro-Scales

Managing forest-water interactions necessitates the reciprocal engagement of forest managers, water users and other stakeholders across hydrologically connected landscapes, in mutually dependent relationships (Postel and Thompson, 2005; Sullivan and O'Keeffe, 2011). Biophysical connectivity across the ecological system couples with socioeconomic connectivity between upwind and downwind, and upstream and downstream, communities. Institutional options and interventions are typically designed to find ways to incentivise behaviour and actions that will produce desired landuse outcomes which either enhance the quality and extent of forest, or improve watershed services (Kerr, 2002; Erickson, 2015). Interventions to improve local watershed services are likely to be highly contextually specific. In similar ways, the social institutions which mediate human behaviour across these landscapes also give rise to specific outcomes that are usually affected by locally contextual factors (Andersson and Agrawal, 2011; Kashwan, 2017). This can make prediction difficult, but there are still some useful generalised principles that allow us to understand the implications of different types of social institutions for incentivising particular types of behaviour.

Informal, everyday practices of mutual recognition and reciprocity have been documented from across a wide range of socio-ecological landscapes (Daily, 1997). These are often negotiated and managed through everyday social norms, but can come under pressure as demands increase and established customary systems come under additional strain (Bhusal and Subedi, 2014; Buytaert et al., 2014). In response, local actors might need to develop more structured and formalised systems to share water. In an example from Mustang district in Nepal, for instance, Bhusal and Subedi (2014) document an arrangement where river water is shared between two villages on different days. While this does not remove all conflict, it is an example of a negotiated outcome, mutually agreed between the villages without the need for external intervention and/or formal legal enforcement.

More formal interventions often involve regulatory restrictions on activities within catchments and watersheds, either imposed by government authority, or negotiated and mediated across multi-stakeholder fora (Daily et al., 2009; Zhang and Putzel, 2016). In an example from the Wasatch watershed (US), Blanchard et al. (2015) show how high value recreational use and development activities are managed through a mix of regulations implemented by multiple agencies, coupled with a commitment to public land ownership and conservation strategies oriented towards the delivery of societal benefits (specifically, the supply of water to Salt Lake City). These interactions are managed under umbrella institutions such as the Wasatch Front Regional Council and the Central Wasatch Commission (previously known as the Mountain Accord) which seek to build consensus across multiple stakeholders affected by decisions across the watershed.

In recent years, these reciprocal interactions have used either direct payment mechanisms, or rewards and compensation associated with particular types of behaviour or actions, to specifically alter management practices across hydrologically-connected actors in a landscape (Jourdain

Silver Lake in Uinta-Wasatch-Cache National Forest, Utah (US) and the surrounding Wasatch watershed provide multiple benefits to stakeholders and users

Photo © Andre Purret

et al., 2009). The next section reviews these 'market-based' instruments, and provides some examples that indicate some of the factors that contribute to the effectiveness of these interventions in particular contexts, while also recognising their limitations. The more general point is that there is greater visibility of what have been called "Reciprocal Watershed Agreements" (Asquith, 2011), or "Investments in Watershed Services" (Vogl et al., 2017) as impactful ways to intervene in landscapes to enhance the availability and quality of water. These measures are often triggered by the interests of 'receiving' communities who attempt to reward the behaviour of those who are in a position to influence the supply of watershed services (Muradian et al., 2010).

6.5 Socio-Economic Instruments and Incentives

Over the last decades we have witnessed a growing interest in market-oriented solutions, typically termed market-based instruments (MBIs), in the context of nature conservation and environmental management (also see Chapter 5). The term MBI is still a diffuse and relatively broad concept (Pirard, 2012) comprising a wide variety of tools, for example: taxes, user fees, cap-and-trade schemes, mitigation banking, offsetting schemes, eco-certification and labelling, the so-called payments for ecosystem services (PES), eco-compensation and others (Jack et al., 2008; Muradian et al., 2013). The most widespread, and for long most adopted, definition of PES is that of Wunder (2005) by which PES are defined as

Box 6.3

Connecting gender, water and forests

Feminist and other critical scholars have long pointed out that gender differences affect resource allocation, use, management and decision-making in both the Global North and the Global South (Fortmann et al., 1997; Arora-Jonsson, 2014). Devolution of decision-making in forest and water management has not translated into greater participation or empowerment in either context (Meinzen-Dick and Zwarteveen, 1998). Researchers have described a tendency of government officials and practitioners to rely on unitary models of 'household' and 'community', thereby ignoring structural, cultural and logistical barriers that limit women's nominal and effective participation in decision-making institutions (Colfer, 2013). This observation is true in countries like Canada and Sweden where gender equality is typically assumed (Reed and Varghese, 2007; Arora-Jonsson, 2010) as well as in countries like Nepal and Mexico where high rates of male outmigration have altered customary decision-making practices (Giri and Darnhofer, 2010; Worthen, 2015).

Furthermore, how gender disparities affect the demand for, and use of, ecosystem services remains poorly understood (Villamor et al., 2014). While rural women and men both rely on ecosystem services for food and water security; gender norms, relations and identities affect their access to these services differently. Shackleton and Cobbin (2016) point out that rural women in South Africa are more vulnerable than men to the effects of climate change on ecosystem services due to higher rates of female poverty, infection (HIV/AIDS), and gender-based violence – findings that have been shared in other countries around the world (Colfer, 2013). Climate change exacerbates these inequalities and has yet to be addressed by climate change mitigation programmes in the Global South (Westholm, 2017).

Evidence suggests that strengthening land rights for women can reduce their poverty as well as that of their households. However, the research is sparse and fails to account for the complexity of land right regimes in the Global South, particularly outside of Africa (Meinzen-Dick et al., 2017). At least two challenges remain. First, while there is considerable feminist scholarship within the environment and development literature, much of it has not been exchanged with scientific scholars focused on water-forest connections. Relatedly, there remain large geographic and conceptual gaps in understanding of the social dimensions of water and forest management. Some have noted a strong empirical emphasis on African countries (e.g., Meinzen-Dick et al., 2017), others have remarked on weak conceptualisations of household and community (Colfer, 2013). These gaps mean that there is no consistent or shared terminology among the limited number of researchers working at the forest-water interface. Second, donor agencies have focused considerable attention on market-based instruments such as payments for ecosystem services (e.g., REDD+). Such schemes have tended to make simplistic or unjustified assumptions about resource access and clarity of rights (van Noordwijk, 2017). As existing property regimes have favoured men's access to natural resources (Fortmann et al., 1997), pre-requirements of property rights for PES reinforces existing bias. Payment schemes like REDD+ are more likely to favour men's interests over women's (Westholm, 2017). There remains much to be learned about the nature of land rights and how these may affect and be affected by gender relations (Meinzen-Dick et al., 2017).

"A voluntary transaction where a well-defined service (or land-use likely to secure that service) is being 'bought' by a (minimum) one ES buyer from a (minimum one) provider if and only if the ES provider secures ES provision (conditionality)". Clarity of property rights, cause-effect relations in ES generation and opportunities for monitoring ES provision may not exist in large parts of the world (Swallow et al., 2002). In reality, many of the applications are PES-like rather than PES. New alternative terms and definitions have emerged since, mirroring a conceptual debate about what is needed to become effective in complex and contested landscape realities (Swallow et al., 2009; van Noordwijk and Leimona, 2010; van Noordwijk et al., 2012; Chan et al., 2017). Wunder (2015) reviewed some of the new terms and definitions and the accompanying conceptual debate.

A common feature of MBIs is that they use market mechanisms, such as trading schemes, price signals or auctions (Jack et al., 2009; Ajayi et al., 2012; Wünscher and Wunder, 2017; Leimona and Carrasco, 2017) to induce behavioural changes in pursuit of specific environmental goals. They have frequently been deemed as instruments that help achieving environmental goals in a more efficient way rather than relying only on regulatory (command and control) efforts. MBIs have also been promoted by the assumptions that environmental problems are primarily the result of market failures (Muradian and Gómez-Baggethun, 2013; Reid and Nsoh, 2016), and that MBIs can help to correct failures of current markets by improving price signals in a more flexible setting (Engel et al., 2008). Some MBIs, such as PES are also perceived as an opportunity to produce social and cultural co-benefits including improved livelihoods for ecosystem services providers (Ingram et al., 2014), although this perception has been challenged, for example by researchers working with women where PES for climate change mitigation in the global south has been introduced (Westholm 2017). The growing attention to MBIs has attracted various types of critiques and questions (Brockington and Duffy, 2010; Chiabai et al., 2011; Muradian and Gómez-Baggethun, 2013). Alternative concepts such as compensation and coinvestment, with a stronger focus on balancing fairness and efficiency have emerged, especially in Africa and Asia (Jourdain et al., 2009; Namirembe et al., 2014, 2017; Leimona et al., 2015). There is also a powerful critique from a gendered perspective, suggesting that MBIs reinforce structural inequalities in resource allocation, use, management and decision making (see Box 6.3)

In Latin America, and other developing country contexts such as Southeast Asia (Brouwer et al., 2011; Hoang et al., 2013), implementation of an MBI referred to as payment for water services (PWS) from forests has become increasingly widespread (Martin-Ortega et al., 2013). While less common, these mechanisms have also been applied in China, India, Nepal and some African and Caribbean countries to secure water services supply (Porras et al., 2008). Industrialised countries are also showing an increasing interest in PES (e.g., the debate is particularly notable in Germany, the US (Matzdorf et al., 2014) and the UK (Waylen and Martin-Ortega, 2018)).

Text Box 6.4 provides an overview of the key characteristics of the Latin American experience on payments for water services provided by forest. More recent PWS mechanisms were implemented in Bhutan where the upstream community forest group agreed on six main tasks as part of the PES contract: maintaining a buffer zone of no disturbance to natural vegetation above two main water sources; guarding community forestry from illegal extraction of forest resources; forestation in landslide-prone and barren areas; clearing fallen trees and branches from the streams; restricting cattle grazing to day-time hours; restricting the number of cattle that can be kept per family and protecting spring water sources. For these efforts, the community forestry group receives a yearly payment of Nu 143,000 (~ USD 2,200) from the two downstream users – the Mongar municipality and district hospital. While this amount does not quite compensate the upstream communities for their foregone incomes (from logging and animal husbandry), the community saw protection of forests as a long term investment and was therefore willing to accept a payment that was lower than their immediate lost income (personal communication, Water Management Directorate official).

Payments are expected to be 'conditional' on the delivery of ecosystem services or on the actions that are supposed to deliver those services. Those payments are also expected to provide 'additionality' i.e. go beyond what would be delivered in the absence of the scheme. Environmental additionality is a necessary condition for any positive improvement in the economic efficiency of any PWS or PES scheme. Yet, many if not most of these schemes often lack conclusive evidence on their environmental performance (Brouwer et al., 2011; Asbjornsen et al., 2015), and establishing this link is crucial to those who are paying for these services, and the successful implementation of such schemes (Meijerink, 2008; Porras et al., 2013). Insufficient monitoring and evaluation of PWS or PES performance is commonly cited as a primary limitation in identifying both direct and indirect socio-economic and environmental impacts of these schemes (Asbjornsen et al., 2015). A common problem for practitioners, in the contexts in which many PWS operate, is that the environmental additionality cannot be accurately measured or demonstrated, as it is surrounded by high levels of uncertainty and characterised by incomplete information. Several years of experience gained in monitoring the compliance and effectiveness of PWS schemes in developing countries has provided some lessons that are summarised in Box 6.5.

6.6 Towards Forest and Tree-Based Management in Critical Water Zones

This section presents an overall approach to water-sensitive landscape management, where the flows of watershed services are an explicitly recognised priority for decision-makers and stakeholders. It focuses on the importance of identifying specific parts of the landscape that are of particular importance for securing hydrologic flows of an appropriate quality. These are now often referred to as 'critical water zones', which recognises both the

Box 6.4

Payment schemes for eco-system based water services provided by forests in Latin America

Based on a review of the literature on 40 PES for water ecosystems services provided by forests in Latin America (Martin-Ortega et al., 2013):

- Deforestation is the biggest threat to water resources to which PES try to respond, but there are often various threats acting simultaneously;
- The large majority of transactions include a bundle of services. Half include more than just water-related services (such as carbon sequestration). Often services are poorly defined;
- Improving extractive water supply is the most common service in existing transactions;
- Payments are almost always conditional on inputs (i.e., actions) rather than on outputs;
- Transactions usually include multiple actions carried out by the seller. Forest conservation, reforestation and forest management are the main actions paid for;
- Landowners and farmers are the key service sellers, but the literature does not always make a clear distinction between them. Also, researchers frequently do not differentiate between benefits realised by male and female producers (Westholm, 2017);
- Hydropower companies and domestic water users are the most frequent service buyers;
- Most schemes involve at least one intermediary (commonly an NGO);
- Payment levels are mostly set in top-down decisions rather than through buyer-seller negotiations. The large majority of schemes operate on local-scale rules or arrangements, but some schemes follow a mix of national and local rules;
- Estimates of willingness to pay or opportunity costs are missing and therefore, cannot be compared with actual payments;
- The large majority of transactions involve cash payments but in-kind payments are also important;
- There is great variance in the payments across schemes. Average sellers' receipts are more than 60% higher than the average payments made by buyers, suggesting a subsidising component;

Box 6.5

Key facts regarding compliance and additionality monitoring findings in developing countries

Compliance monitoring (conditionality)

- Land uses/practices are used as proxy indicators of the production of watershed services, and environmental additionality is often based on local perceptions
- Rapid assessment methodologies (e.g., Jeanes et al., 2006) are being promoted to bridge the gap between science and local perception.
- Most common types of compliance monitoring in PES schemes are: self-monitoring and participatory monitoring.
- In many cases, the compliance and enforcement mechanisms are suboptimal, lack appropriate funding and institutional capabilities, and are affected by poor communication between actors involved in the PES scheme.
- High fines often deter noncompliance, but the voluntary nature of PES limits the range of sanctions that can be applied, creating potential incentives to breach contractual responsibilities.

Additionality monitoring

- Most baselines have focused on measuring onsite forest cover, rather than measuring quantity and quality of water.
- Failures on attribution (i.e. causal effect of PES and water services) can lead to confusion and promote projects with little or no impact or even negative impacts.
- Leakage can be one of those negative impacts (i.e. by generating environmental damages elsewhere). A common example is conversion of forest to cropland outside of the targeted area.
- Perverse incentives (i.e. inducing onsite or offsite expansion of environmentally destructive activities) might also be unintended consequences.
- More research is needed to better understand the potential perverse effects and the likelihood of their occurrence.

Source: Own elaboration based on Porras et al. (2013)

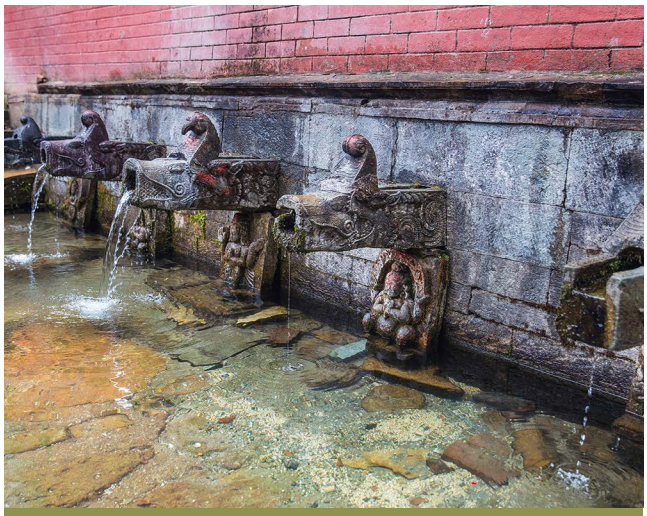
importance of, and pressure on these specific parts of the catchment, with a view to finding ways to mitigate risk. The section also considers the importance of managing stakeholder interactions across forest-water landscapes in the context of environmental and social change.

6.6.1 Identifying Critical Water Zones

A number of critical water zones may be identified across any landscape in which trees and forests exist. The identification of these critical water zones across the landscape is a crucial first step if water-sensitive land use management practices are to be implemented, and watershed services delivered (Groffman et al., 2003; Postel and Thompson, 2005). Exactly what constitutes a critical/sensitive water zone, and how best to identify and delineate these, may differ from country to country. However, the zones that are most commonly considered critical in terms of forest/water relationships include water source areas and riparian/wetland areas, as well as appropriate buffer zones around these (Nava-Lopez et al., 2016; Zheng et al., 2016). The importance of managing and protecting

these critical water zones because of their contributions toward delivery of water of sufficient quantity and quality for downstream users, has been recognised internationally and mapped accordingly (Dudley and Stolton, 2003; Viviroli et al., 2007). This is particularly pertinent in countries characterised by highly variable climate and rainfall, which usually translates into uneven distribution of water resources and often a case of a small fraction of the country producing a disproportionately large amount of usable water. Box 6.6 provides an example of how the recharge zones for springs in the Hindu Kush-Himalaya (HKH) region are a focus of attention, recognising the importance of these critical water zones for the lives of millions of (especially poorer) households in the region.

Previous work mapped South Africa's surface-water source areas and showed that just 8% of the country's land surface area contributed 50% of its runoff (Nel et al., 2011, 2013). The term 'water source area' should ideally include both surface-water and groundwater source areas, and it should include an indication of the strategic significance



Godavari Kunda, a sacred spring located on the outskirts of Kathmandu, Nepal. Thousands of pilgrims come to the spring every 12 years (next time in 2027) to bathe and gain spiritual merit

Photo © Jitendra Bajracharya/ICIMOD

Box 6.6

Critical water zones for spring recharge in the Hindu Kush – Himalaya region

Springs are the main source of water for millions of people in the mid hills of the Hindu Kush-Himalayas (HKH), and springsheds are a critical water zone in this region. A number of studies based on people's perceptions have attributed drying of springs to changes in land use – mostly in the form of conversion of forests to agricultural land (Joshi et al., 2014) and degradation of forests (Rautela, 2015; Pandey et al., 2018), including changes in forest types (Ghimire et al., 2012; Naudiyal and Schmerbeck, 2017).

While it is well recognised that water supply from springs is one of the many provisioning services provided by forests (Paudyal et al., 2017), the regulating role of springs (for example, in maintaining water quality) is not as well known. Some literature has highlighted the heterogeneity in spring habitats. Layers of mosses and debris in conjunction with high diversity in substrate often provide a microhabitat mosaic resulting in colonisation and often elevated levels of biodiversity. Although spatially close in many cases, spring habitats are often isolated and contain unique taxa different from streams, groundwater and other springs (Cantonati et al., 2006).

Our knowledge (or the lack thereof) about spring supported habitats becomes even more important in the current scenario of drying up of springs. Restoration of degraded springs enhances the quality of spring habitat (Lehosmaa et al., 2017). It is possible to restore drying springs by correct identification of recharge zones using knowledge of hydrogeology and then implementing recharge measures in those zones. Various countries in the HKH are increasingly turning their attention to the issue of spring revival. This has been successfully attempted in Sikkim State in India where more than 60 springs have been revived so far (Tambe et al., 2012) and the International Centre for Integrated Mountain Development (ICIMOD) and its various partners have documented the various steps of this spring revival protocol (Shrestha et al., 2018, forthcoming).

The Niti Aayog, the highest planning body in India, recently constituted a working group comprising experts from regional organisations like ICIMOD and civil society bodies in India to design a concrete plan for revival of drying springs in Indian Himalayan states. In Bhutan, the Ministry of Agriculture and Forests has plans to create a national spring inventory and initiate pilot projects to enhance recharge and this has been included as a priority action in the country's 12th Five Year Plan starting from 2018.

Watershed experts of the Nepal Water Conservation Foundation have made some counterintuitive findings in the Bagmati watershed regarding the role of traditional recharge ponds, landslides and village spring flow enhancement (Upadhya, 2009; ICIMOD, 2015; Sharma et al., 2016). Finding landslide control with conventional check-dam building both expensive and ineffective, the Bagmati watershed managers experimented with reviving ponds on the ridge tops, most of which were also buffalo wallowing ponds but had been abandoned and silted up. They found that for a minimal cost of cleaning up the ponds or excavating new ones, landslides were stabilised. The post-hoc explanation is that by putting a break on the flow of floodwaters gushing down during heavy rainfall, the erosive power of water was significantly reduced. Similarly, drying of mid-hill springs were related to either earthquake disturbances or social drivers such as outmigration of youth, decline in livestock and the concomitant abandonment of buffalo wallowing ponds that also served as sources of recharge; unregulated use of PVC pipes and electric pumps; shift from dryland crops to water-intensive vegetable farming etc. Given that rainfall was as stochastic as ever and there was no noticeable decline in precipitation, climate change could not account for the current situation although it is predicted to exacerbate the situation unless the current drivers are first addressed.

of the water source areas from national water resource planning perspectives. Riparian and wetland areas are also critical water zones, and country-specific practical field procedures for identification and delineation of these have been developed. For example, in South Africa, wetlands are considered to be "*land which is transitional between terrestrial and aquatic systems where the water table is at or near the surface, or the land is periodically covered with shallow water, and which in normal circumstances supports or would support vegetation typically adapted to life in saturated soil*" while riparian areas are considered to be "*those areas closely associated with a watercourse which are commonly characterised by alluvial soils, and which are inundated or flooded to an extent and with a frequency sufficient to support vegetation with a species composition and physical structure distinct from those of adjacent land areas*" and the buffer zones around these are considered to be "*the 30m strip from the 1:50 year flood line of a river, spring, natural channel in which water flows regularly, or intermittently, lake, dam or wetland*" (DWS, 2008). Once these critical water zones have been recognised and delineated, historic trends and future projections can help to identify existing and potential threats to these areas, and how these might be either reversed, or mitigated. It is also important to recognise that there are often trade-offs associated with forest management for multiple ecosystem services, in particular in relation to timber production, carbon sequestration, and water quality and quantity (Cademus et al., 2014; Wang et al., 2017). These trade-offs need to be carefully understood, and specific priorities for each management unit need to be negotiated within a context of multi-stakeholder decision making.

6.6.2 Mitigating Risk to Critical Water Zones

To effectively mitigate forest/water related risks to critical water zones it is first necessary for policies to be in place which acknowledge the importance of and pressure on these zones and which formalise appropriate protective and legal measures. Thereafter there is a need for management practices which are SMART (Specific, Measurable, Achievable, Realistic, and Timely) and forward looking (consider the 'new normal').

Following the delineation of water source areas and riparian and wetland areas within catchments, as well as appropriate buffer zones around these, protection of these areas and mitigation of risks to them can be facilitated by a number of practical best management practices (FSA, 2017), including:

- Maintaining native forests in a healthy condition (for flood mitigation and sustained base flow);
- Eradicating alien and invasive species that may reduce water yield from within the critical water zones and buffer zones;
- Actively removing or minimising tree plantations of single and/or exotic species which would reduce water yield from the buffer zones;
- Developing a comprehensive land use map for forested/plantation areas, incorporating proposed forest management units, a soil map, delineation of natural vegetation areas, identification of water courses and wetlands, inclusion of existing roads and any new roads planned, including stream crossings;
- Prohibiting the use of chemicals in forestry operations within critical water zones;
- Designing timber extraction routes, depots, and forest and plantation roads in a manner that limits potential sedimentation of water source areas, rivers and wetlands;
- Disconnecting forest drains from main watercourses as contamination in the former (especially road drains) can lead to water quality deterioration in the latter;
- Managing slash / waste from timber plantations with the objectives of retaining soil nutrients, preventing soil moisture losses and minimising water runoff which may cause erosion;
- Conducting burning regimes which reduce understorey fuel load in commercial tree plantations and maintain the ecological health of fire-driven grasslands and wetlands;
- Initiating rehabilitation measures after timber harvesting operations, to reduce soil erosion and sedimentation; and
- Limiting and responsibly managing applications of chemical herbicides and pesticides to avoid negative water quality impacts.

Programmes which have formalised the removal of invasive alien trees in order to augment water resources/streamflow have been developed in some countries. An example of this is the 'Working for Water' initiative (see also Chapter 7), pioneered as part of the Natural Resource Management Programme of the Department of Environmental Affairs in South Africa (Turpie et al., 2008; van Wilgen and Wannenburgh, 2016). This could also be considered an incentive scheme through job creation, water augmentation and improved environmental health.

6.6.3 Stakeholder Engagement and Decision-Making around Critical Water Zones

Empowering stakeholders to take action in support of water-sensitive forest management requires clarity and established protocols on who can do what, when and how. German (2010) suggests that the principle of subsidiarity (the making of decisions at the lowest possible level of the political-administrative hierarchy) is desirable. Thereafter, the importance of promoting an enabling management framework for local application and empowerment is critical. An example of multi-stakeholder engagement around the management of the ecological (including water) impacts of commercial tree plantations is seen in the South African approach of convening a LAAC (Licence Assessment Advisory Committee) (see Box 6.7). This comprises a meeting of representatives from different stakeholders in the particular basin in which expansion of commercial afforestation is proposed. The anticipated impacts (including water impacts) of the proposed afforestation are discussed and, ideally, consensus is reached as to whether the licence to conduct afforestation may be issued or not. What conditions enable such approaches to succeed, and how knowledge contributes to the ways in which decisions are made, are important considerations.

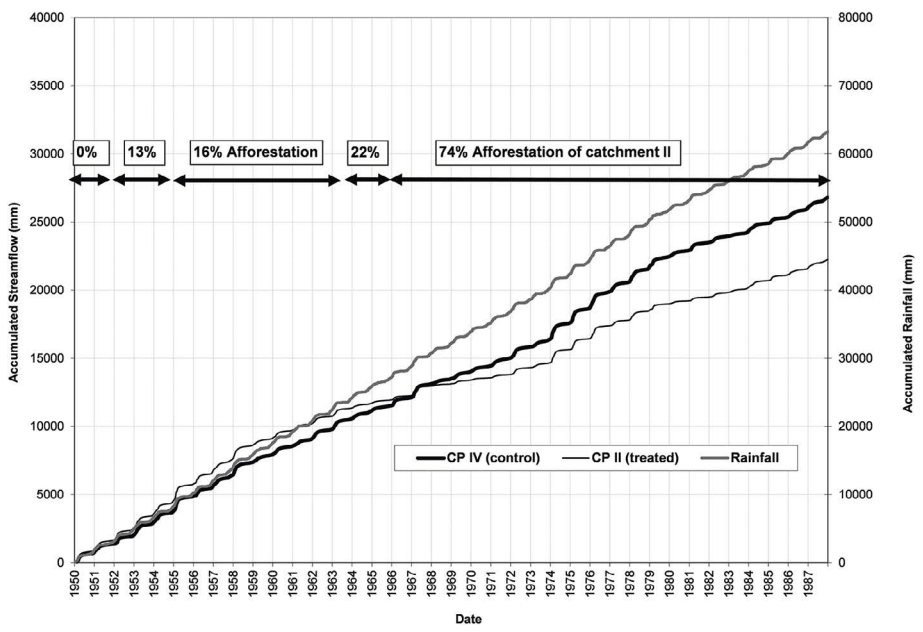
Box 6.7 Figure 6.2

Water sensitive decision making: The case of commercial tree plantations in South Africa

South Africa is a semi-arid country (mean annual precipitation of 500mm), with a strong east-west gradient to rainfall, and minimal native forests. The dominant vegetation types across the country are savannah, grassland and scrub, dominated by shallow-rooted, low leaf area plants, many of which are dormant in the dry season. Areas of native evergreen forest do exist (<1% of the country), however these were officially protected since demand for their timber far exceeded their ability to supply. This forced South Africa to accelerate the expansion of its own commercial forestry industry. Plantations of fast-growing introduced tree species (Eucalyptus, Pines, Acacias) were subsequently established in the high-rainfall regions of the country, which are also important water source areas. Commercial plantations expanded to approximately 1.5 million hectares in 1996/1997 but now cover approximately 1.2 million hectares (FSA, 2017). The deep-rooted, tall, dense, evergreen physiology of these plantations contrasts strongly with the typically short, seasonally dormant vegetation with shallow root systems (e.g., grassland) that they usually replace during establishment. Resultant streamflow reductions led to the initiation of South African forest hydrological research in 1935, and the establishment of long-term paired catchment research stations (e.g., Cathedral Peak). Observed data from these, and other international studies, indicated conclusively that evapotranspiration from forest plantations exceeded that from grasslands or shrublands, and thus reduced annual water yield (streamflow) from afforested catchments (Figure 6.2).

Resultant water policy in South Africa is grounded in the fact that it is a water-scarce country, and commercial tree plantations are consequently highly regulated (Kruger and Bennett, 2013; Scott and Gush, 2017). In order to manage the conflict for a limited water resource, and based on the findings and recommendations emanating from forest hydrology research both in South Africa and internationally, the state introduced afforestation permit legislation in 1972. Subsequently, through the National Water Act (NWA, Act No. 36 of 1998) commercial afforestation was declared a streamflow reduction activity (SFRA) or land use that may reduce the amount of water in rivers and thus what is available to downstream users. This was necessitated by the need for appropriate control over the use of water resources, preventing uncontrolled dwindling of the resource, and allowing sufficient water to meet the Human and Ecological Reserve (water required for basic human consumption and ecological functioning).

The current afforestation licensing and regulation system is based on research which extrapolated results from the paired catchment studies to all potential forestry areas in South Africa through modelling exercises (Gush et al., 2002; Jewitt et al., 2009). The results are used by the relevant authority (Department of Water Affairs) for evaluating licence applications for the establishment of tree plantations, in the context of catchment-scale water resource management decisions. Water use authorisations and forestry licence allocations are currently overseen by regional Licence Assessment Advisory Committees (LAACs). These are co-operative governance committees, which include representatives from the forest industry, the environment, society, and regulators from departments implementing relevant legislation.



Source: Mark Gush (author's own elaboration)

Figure 6.2. Accumulated daily streamflow data (mm) between 1950 and 1987 for Cathedral Peak catchments IV (grassland) and II (afforestation treated). Progressive afforestation treatments (*Pinus patula*) applied to catchment II are annotated on the figure, and accumulated daily rainfall data are also illustrated

6.7 Knowledge Gaps and Data Needs

Successful forest management depends to a large extent on the ability to accurately assess the current forest condition, as well as longer term changes in forest condition over time. Traditionally, such information was gathered through detailed, repeated measurements of forest plots or by more extensive, less intensive sampling (Scott and Gove, 2002). However, the cost of such collections can be an impractical financial burden on developing nations. Additionally, many forest areas may be remote and inaccessible even for those countries that can afford plot level measurements.

The advent of remote sensing since the early 1970s has expanded land managers' ability to observe both the current condition of forests, and disturbance impacts (e.g., wildfire, insect, wind) on these ecosystems. Satellite and laser-based imagery (combined with the use of unmanned aerial vehicles) can be a very cost-effective monitoring and assessment tool. For example, hyperspectral imagery has provided information about forest leaf area (Asner et al., 2003), nitrogen content and productivity (Smith et al., 2002). However, correlations between satellite imagery and forest level structure and function need to be established before many advanced aspects of remote sensing can be applied. Data for algorithm establishment and ground truthing is lacking for many ecosystems in many parts of the globe. Although the technology exists to better manage large areas of forest remotely, the linkages between remote sensing signals and forest structure and function are a major impediment to the deployment and use of these technologies. Furthermore, it is important for the scientific community to make more effort to harmonise the way the satellite and remotely-sensed data is interpreted as lack of consistency between different earth observation systems has led to a lack of clarity about the true extent of forest cover. While recent development of 'drone' technology has enabled a broad expansion of the ways in which forests can be studied as ecosystems, and the ways in which forests can be established in remote areas through drone-based seed dispersal, there is much need for greater understanding of the limitations of these approaches and the best way to utilise their full potential.

Using drones in forest monitoring has become increasingly popular around the world

Photo © Pixabay: Pexels

In addition to the direct use of remote sensing information, the data can also be used to parameterise ecosystem models. These models can be very useful for estimating monthly, seasonal and annual water yield under current and future climates for areas that lack stream gauge systems (McNulty et al., 2016). Such tools can assist land managers to avert future water shortages through thinning and other forms of forest management. It is important to improve model performance; models can often be subject to large errors due to the underlying assumptions, over-simplification of complex processes, the lack of data and poor validation and calibration. These issues need to be addressed before there can be greater confidence in model outputs.

A simple modelling framework is needed to facilitate the application of forest-water interactions to meaningfully improve transport and redistribution of water resources from the local to the cross-continental scale. Opportunities to capture atmospheric moisture could intensify and thus improve our understanding of the hydrological cycle.

An equally important knowledge gap involves the translation of scientific data into practical information and management guidance. Remote sensing data, combined with forecasting models, have the ability to predict forest productivity and composition, but knowledge regarding the relationship between forest productivity and water use is lacking. Better education is needed for forest managers to allow them to find the correct balance between competing natural resource needs given the information that they have been given.

A further knowledge gap concerns evidence on the ecological effectiveness of different types of incentive-based mechanisms for the management of forest-water interactions. Many interventions focus on monitoring inputs into a management system, as these actions are easier to observe and measure. The relationship between these inputs and the ultimate ecological outcomes is mediated by a number of intervening factors, some of which are not directly observable. This means that actors may, in good faith, undertake all the actions that are required under a conditional scheme for improving ecosystem service flows in a landscape, but this might not always result in the desired ecological gains. We need to improve our ability to monitor the actual ecosystem services that are the focus of such interventions, going beyond the use of actions and inputs as proxies for these services.

In addition, this chapter has highlighted that there are a number of ways in which reciprocal relationships across forest-water landscapes are managed in multi-stakeholder decision settings. While there is a growing emphasis within some policy, academic and donor literature on the importance of mediating these relationships

through market, or quasi-market structures, there is a need to recognise that there are alternative ways to organise these social and institutional settings which build on mutual commitment and reciprocity, but do not necessarily rely on the logic of markets and incentives. There is a need for more systematic evidence on these plural institutional forms, and what makes them work in specific settings, to expand the toolkit of interventions beyond the current focus on payments and markets.

6.8 Conclusions

This chapter has examined a range of forest and water management strategies that respond to some of the challenges that have been articulated in the earlier chapters of this report. In particular, it focuses on the types of landscape level and socio-institutional interventions that can respond to the need to prioritise water as a key objective for forest and landscape management. The findings of the chapter can be summarised in seven overarching conclusions:

1. At catchment scale, management responses that increase carbon storage, timber, pulpwood or fuel productivity are likely to reduce catchment annual water yields due to evapotranspiration. Management of forests for particular animal or bird species will impact streamflows differently, depending on the habitat type that is most suitable for the target species – if target species prefer newly cut or open areas, water yields are likely to increase, while management for species that prefer closed canopies and old growth forest would increase forest evapotranspiration and reduce annual water yield.

2. Riparian zone vegetation, cross-slope woodland, soil bunding and brush barriers can be used to slow down the flow of water in a catchment, while also reducing sediment loads and soil erosion. Forest thinning reduces water quality by increasing sediment loads, but an increase in the volume of water in a catchment might dilute nutrient loads and improve water quality. The balance between those two effects, and the appropriate management actions, reflect the nature of the catchment and the surrounding land uses. In an agriculturally dominated landscape, the dilution effect on inorganic fertilisers might be more significant, while sediments and silt loads might matter more in catchments that are susceptible to soil loss and erosion.

3. These localised effects cascade across interconnected catchments and basins, suggesting the importance of looking at wider scales of management. At these scales, it is also important to consider atmospheric transport of moisture, and the role of forests and tree cover to contribute to downwind precipitation. Once these broader effects are taken into consideration, managing forests for water might need to consider both localised impacts at catchment level, as well as impacts on atmospheric moisture and precipitation regimes at larger continental scales.

4. Forest-water interactions necessitate the reciprocal engagement of forest managers, water users and other stakeholders across hydrologically connected landscapes, in mutually dependent relationships. Social institutions which mediate interactions across these landscapes range from informal, everyday practices of mutual recognition and reciprocity, to more formalised regulatory regimes and contractual relationships between interconnected communities.

5. There has been growing interest in the role of market-like and incentive-based mechanisms to mediate stakeholder relationships within forest-water landscapes. These schemes, often called 'payments for ecosystem services', 'reciprocal watershed agreements', or 'eco-compensation mechanisms' have varying levels of expectations in terms of service delivery, conditionality, observability (and monitoring) of actions and outcomes, and the scales at which they are implemented. Such interventions have also been criticised for unequal (gendered) impacts, and the reinforcement of structural inequalities across differentiated landscapes. Despite their growing popularity, many such schemes still lack conclusive evidence of their environmental, economic and social effectiveness.

6. An overall approach to water-sensitive landscape management needs to recognise the importance of critical water zones – water source areas and riparian/wetland areas as well as surrounding buffer zones that have the greatest impact on the socio-hydrologic system. These strategically important areas need to be recognised and delineated, and current and future threats need to be identified, and to the extent possible, mitigated, to maintain their contributions to the forest-water system. Management practices need to be context specific, responding to the structure and function of the biophysical system, as well as the stakeholders who influence landuse and forest management decisions within the landscape, and those who are hydrologically impacted by these decisions at catchment, basin and continental scales.

7. Knowledge and data for a complete understanding of these coupled socio-hydrologic systems remain inadequate, and there is need for better monitoring, as well as an improved used of new techniques, which include modelling, the use of new data sources and techniques, as well as a greater sensitivity to local observation and alternative (including indigenous) knowledge systems. It is also important to understand how different socio-institutional mechanisms (including those that promote markets and incentives) influence stakeholder behaviour, to determine which types of interventions are most suitable for different types of landscapes, different socio-economic conditions, and different management objectives at a variety of scales.

Given the vital role water plays even in facilitating the continuous sequestration of carbon in standing forests, a

lack of understanding of landscape-scale effects amongst the hydrological and forest science communities and policymakers, is of increasing concern as it raises the risk of policy failure in managing forest resources for water quality and quantity.

There is an urgent need to improve the way forest and water managers are trained, to bring them together in a more integrated way so that in the future, forests can be managed explicitly for water as well as other benefits. Indeed, it is important that governments recognise that there is much benefit in facilitating greater cooperation between these two branches of government responsibility.

Without a better understanding of atmospheric hydrology and land use teleconnections, land managers may not be able to generate the maximum benefit from forest management. Forests must be viewed holistically, in full recognition of the multipurpose benefits they generate, not only at the local scale for local users, but for more distant beneficiaries, both downstream and downwind. The important role that forests play in water quality improvement is already well recognised at the local and catchment scales, but the benefits of the other multiple ecosystem services provided by trees and forests may also be dispersed beyond the catchment in which they are growing.

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