Chapter 4

Forest Landscape Hydrology in a 'New Normal' Era of Climate and Land Use Change

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4.1 Introduction

Building on the eco-hydrological insights reviewed in Chapter 2 and the ongoing change in determinants of the forest-water relationship in Chapter 3, this chapter aims to clarify the effects that changing climate and quantity, quality and pattern of tree cover in forests have on the way water becomes available for human use and ecosystem integrity. To do this, the chapter synthesises current understanding of implications for local and global hydrology of current and anticipated changes to forests and tree cover.

4.2 Current Changes to Forests and Implications for Local and Global Hydrology

The drivers and determinants of change operating at a wide range of spatial and temporal scales modify various aspects of the climate-vegetation-soil-streamflow system. We will now review a number of examples of such changes and relate changes to key parameters (leaf area index, soil surface conditions, rooting depth and macroporosity) and to impacts on major hydrological processes.

4.2.1 Hydrological Consequences of Natural Forest Disturbance

Natural disturbance such as wildfire, insect pests, diseases, windthrow, etc. can greatly change various watershed processes (e.g., water quality, hydrology, channel morphology) and ecological functions in forested watersheds. Natural disturbance is part of the dynamics of forest ecosystems (Attiwill, 1994; Lertzman et al., 1997), increasing spatial heterogeneity and ecosystem complexity, and supporting ecosystem resilience. However, catastrophic forest disturbance (e.g., stand-replacing wildfire or large-scale insect pest outbreaks) can cause undesired ecological and economic consequences. Among all natural disturbance types globally, wildfire and insect pests are two major natural disturbance agents (van Lierop et al., 2015).

Natural disturbances can be described according to their type, frequency, severity, intensity, distribution and area affected. In addition, forest disturbances cumulatively affect watershed processes over space and time particularly in a large watershed or landscape. Some of the most-widely studied natural forest disturbances are described below.

4.2.1.1 Invasive Weeds, Insects and Pathogens

Invasive species often have few (if any) predators that can regulate their population. Without a check on growth, invasive species populations can proliferate (IPBES, 2018). Invasive weeds such as kudzu in the southeastern US can grow more than 25 cm per day and completely cover vegetation in one growing season (Boyette et al., 2014). Weed foliage expands and covers tree foliage, and soil water demand by trees decreases as weed water demand increases. For this reason, there may be no net change in forest water use so that stream flow rates would remain unchanged.

Insects and pathogens are common disturbance agents to forests, and they can significantly influence hydrological processes. The following describes the effects of insects and pathogens on hydrology using the Mountain Pine Beetle (*Dendroctonus ponderosae* Hopkins) (MPB) as an example. Tree mortality from the MPB is caused by larval galleries and their symbiotic blue stain fungi in the inner bark of the trunk (Dhar et al., 2016a). In contrast with other major disturbances such as clear-cut harvesting, this disturbance may allow non-affected non-target overstory and understory trees and shrubs to form new structurally diverse stands. In the first summer of MPB attack, affected trees stop transpiring, however the needles are unaffected, so this is termed 'green attack'. In the following 2 – 3 years, the needles turn red and start to fall – the 'red attack' phase. More than 3 years after MPB attack, once the trees are dead with no remaining needles, this stage is termed 'grey attack'. Although changes in canopy colour increase albedo, reducing winter and early spring temperatures (Vanderhoof et al., 2014; O'Halloran et al., 2012), these changes are offset by a reduction in latent heat resulting from reduced evapotranspiration and associated increases in soil moisture, with the net effect that temperature increases in MPB-affected stands (Cooper et al., 2017). From red attack onwards, surviving vegetation makes use of the increase in available resources, often growing at an enhanced rate (Dhar et al., 2016a). Thus, the changes in energy and vegetation caused by MPB infestation drive hydrological responses dynamically as the stand moves through the stages of attack and post attack recovery.

A MPB outbreak affects all forest hydrologic processes. Tree mortality following MPB attack reduces foliage cover and density, and consequently decreases canopy interception. The more open canopy after MPB attack speeds snow ablation and advances spring melt (by days or a few weeks) compared to unaffected stands (Redding

Mountain Pine Beetle (MPB) infestation in Baskerville, British Columbia, Canada,
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et al., 2008; Winkler et al., 2014). Tree transpiration is reduced following MPB attack, in magnitude proportionate to the severity of mortality (Clark et al., 2014). However, concurrently the opening of the forest canopy increases sun exposure, which increases soil evaporation. These competing processes offset each other (Bearup et al., 2014; Biederman et al., 2014), to a degree which is not well quantified. Understory and surviving overstory trees and other vegetation also affect water dynamics after disturbance (Reed et al., 2014). Although evapotranspiration is reduced after insect attack (Dhar et al., 2016b), this may be short-lived as rapid growth of understory vegetation and regeneration increase evapotranspiration to the level prior to disturbance. The effects of the MPB on streamflow are controversial (Biederman et al., 2015; Penn et al., 2016), but appear to depend on the extent and severity of tree mortality and remaining vegetation recovery (Weiler et al., 2009; Wei and Zhang, 2010; Reed et al., 2014). However, salvage logging following MPB infestation significantly increases high flows and advances their timing (Lin and Wei, 2008; Zhang and Wei, 2013), which can potentially increase floods.

Box
4.1

Wildfire and 2018 Southern California mudflows

In January 2018, a series of large-scale mudflows occurred in Southern California. They killed more than 20 people and caused significant economic loss. They followed a month after a series of catastrophic wildfires in 2017 and occurred after heavy rainfall. Wildfires, particularly catastrophic ones, can significantly alter hydrological processes and cause severe soil erosion and lead to mudflows (Vieira et al., 2015; Schärer et al., 2017). Mudflows are triggered by two different factors: soil erosion caused by rainfall runoff and land-sliding caused by rainfall that can no longer be absorbed (Staley et al., 2017). Although mudflows can happen anytime following heavy rain, they are exacerbated by wildfires. In Southern California, just 13 mm of rain in one hour can start a mudflow (Staley et al., 2017).

Wildfires remove vegetation, reduce leaf area index (LAI) and consume forest floor material, and often cause hydrophobic soils with high water repellency (Vieira et al., 2015) reducing soil infiltration capacity. The removal of those 'protection layers' reduces forest rainfall interception, increasing the kinetic energy of heavy rain and its cumulative erosive power for accelerating soil erosion and mudflows.

Wildfires also affect other hydrologic and channel morphological processes in our landscapes. Removal of vegetation due to the fires can, in turn, reduce evapotranspiration and increase runoff. Fire impacts on evapotranspiration and runoff are most clearly seen in the tropical savannahs, African rainforests, and some boreal forests and Mediterranean forests. Wildfires can also have adverse effects on water quality (Schärer et al., 2017). For example, in September 2017, following massive wildfires, 'black rivers' were reported in northern Spain, caused by the ashes and soil transported by runoff from burnt areas.

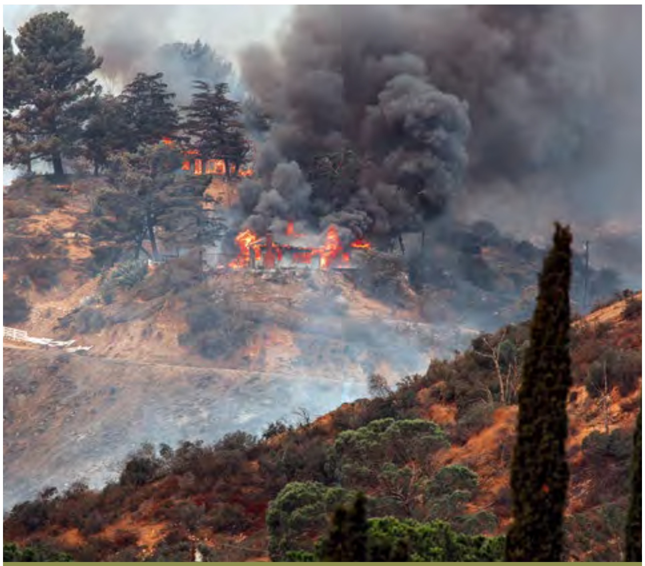
Other pest and pathogen attacks on forest may have similar, mixed and transient effects (Adams et al., 2012). For example, water yield declined and peak flows of large events increased in watersheds where eastern hemlock was lost due to hemlock woolly adelgid infestation (Kim et al., 2017).

4.2.1.2 Wildfire

Wildfire is the most dominant natural disturbance in global forests, particularly in boreal and Mediterranean forests, although its severity, intensity and frequency varies according to forest type (Hansen et al., 2013; van Lierop et al., 2015). Wildfires destroy over 300 hundred million ha of land each year, although this rate has decreased by 25% over the past two decades (Andela et al., 2017).

In Canada alone, wildfire, on average, disturbs 1.6 Mha annually, and accounts for 2.5 times more area disturbed than harvested (White et al., 2017). Wildfire affects both the terrestrial environment and aquatic ecological processes. Severity of forest wildfire depends on meteorological conditions, vegetation type, stand fuel loading and topographic properties (Oliveras et al., 2009). Implementation of fire suppression can lead to accumulation of more fuels which in turn may increase chance of more catastrophic fires (Collins et al., 2013).

In the immediate aftermath of wildfires, the burnt soil is bare and dark, and highly susceptible to erosion, and even mudflows (as experienced in January 2018 in California, see Box 4.1). In the absence of all-consuming crown fires, subsequent tree mortality and litter fall can restore a protective litter layer, but water repellency of soils may cause high overland flow rates with enough energy to carry freshly fallen litter downhill.



La Tuna wildfire in Los Angeles, CA in 2017

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The impact on forest water resources can be highly variable, for several reasons. Intensity, duration and size are all determinants of wildfire impacts on tree mortality (Dunn and Bailey, 2016; Iverson et al., 2017). Therefore, the impacts of wildfire on forest water quantity and quality is also highly variable (Riggan et al., 1994; Vieira et al., 2015; Hallema et al., 2018). As a rule, as wildfires increase, forest leaf area decreases and water flow increases. Variability in climate and the heterogeneous nature of wildfire-induced forest loss can mask this relationship (Hallema et al., 2017). Wildfire impacts on water quality are even more complicated. On forests with little or no slope, wildfires will have minimal impact on forest water quality if the areas are left to naturally regenerate (Hallema et al., 2017). Determinants of both wildfire and post-wildfire impacts (e.g., vegetation loss, soil infiltration change) need to be considered to better assess changes in forest water quantity and quality. Surface fires or ground fires, can change the composition and porosity of soil. Forest fires tend to volatilise waxes and oils from litter, which may condense on soil particles, producing hydrophobic (water repellent) conditions in soils that in turn, reduce infiltration and increase overland flow (Neary et al., 2005). Soil texture, state of aggregation, pH, mineral composition of the clay fraction and microbial activity also affect soil water repellency (Cesarano et al., 2016).

Crown fires or stand-replacing fires are more severe, not only affecting soils, but also destroying canopy structures, potentially impacting on all hydrological processes. Crown fires eliminate above-ground biomass which greatly reduces canopy interception and evapotranspiration (Montes-Helu et al., 2009; Bond-Lamberty et al., 2009) and increases soil evaporation as soils with altered albedo become exposed to solar radiation. As a result of increased net precipitation, soil hydrophobicity and decreased evapotranspiration, crown fires increase annual runoff at the hillslope and catchment scales (Hallema et al., 2017; Kopp et al., 2017). In rain dominated watersheds burned by crown fires, canopy removal and hydrophobic soils increase kinetic energy of rainfall, limit soil infiltration capacity and shorten flowpaths. Consequently, the magnitude of peak flows is increased, and their timing is advanced (Liu et al., 2015). Unlike peak flow and annual runoff, the effect on base flow is uncertain with great climatic and spatial variability. For example, base flow increased following fires during the dry season in many Mediterranean regions (Kinoshita and Hogue, 2011; Bart and Tague, 2017), while in northern Mongolia for example, baseflow declined in the dry season after wildfire, partly due to the diminished water retention capacity of the organic surface layer (Kopp et al., 2017).

4.2.1.3 Ice Storms

Ice storms are winter events characterised by freezing rain, and are common in East Asia (Ding et al., 2008) and North America (Irland, 2000). An ice storm forms along a narrow band on the cold side of a warm front, where surface temperatures are at, or just below, freezing; under these conditions, rain becomes super-cooled and freezes upon impact with cold surfaces (Irland, 2000). Ice storms often have a large spatial extent and may catalyse other types of forest disturbance. For example, ice storm mortality and weakening of trees promoted bark beetle populations (de Groot et al., 2018). The relationship between damage severity, topography and forest type was found to be significant at the watershed scale (Isaacs et al., 2014). Trees with narrow crown, coarse branching, strong branch attachments or low surface area have greater resistance to damage from ice storms (Hauer et al., 1994). Research on the effects of ice storms on forest hydrology is very limited. A sole case study conducted in the Hubbard Brook Experimental Forest to monitor the impacts of the 1998 ice storm on hydrology and biogeochemistry found that stream discharge was not significantly altered, while the NO_3 loss to drainage waters was most apparent (Houlton and Driscoll, 2011).

4.2.2 Hydrological Effects of Human-Driven Forest Changes

4.2.2.1 Silviculture

Specific growing conditions and silvicultural practices have an important bearing on the hydrology of forested watersheds. Managed and unmanaged forests vary with respect to stand density (stems per hectare), tree age distribution (rotation lengths), tree species, stand management practices (weeding, pruning, thinning, etc.) and tree health (du Toit et al., 2014).

Diversity of species

Opportunities to actively manage water use by forests derive from the fact that tree species vary in their use of water from different soil depths or at different times of year (Moore et al., 2011a; Kerhoulas et al., 2013). Forest stands of mixed species display complementary water resource utilisation and may have higher water use efficiency in both temperate and tropical climates (Forrester, 2015; Schwendenmann et al., 2015). Such complementary water resource utilisation suggests that mixed-species forests may be more resilient to drought. In dry regions, management actions that maintain or create low-density stands of large, deeply-rooted trees increase tree access to water from winter precipitation stored in deep soil layers (Kerhoulas et al., 2013). The effects of species and leaf area on stand level water use may be countered by differences in soil moisture and nutrient status among sites (Moore et al., 2011a).

Age

Water use by individual trees in forest stands increases from the seedling stage to the closed canopy stage (Scott and Smith, 1997; Dye and Bosch, 2000), but stand-level transpiration appears to decline in old-growth native forests or mature plantations (Scott and Prinsloo, 2008). Tree age has the greatest effect on differences in water use with young forest stands using much more water than old-growth forest stands, followed by differences in basal area and finally species composition (Moore et al., 2004). Transpiration is more strongly coupled to streamflow when soils are wet, but transpiration may produce lagged, diel variations in streamflow during dry seasons (Moore et al., 2011b).

Very few studies have attempted to scale tree and forest-stand water use to the watershed. At the watershed scale, native forest stands of old-growth trees use more water in the wet season, thus mitigating floods, while simultaneously using less water during dry periods, compared to closed-canopy managed forests of native tree species (Jones, 2000; Jones and Post, 2004). Because of high water use by young, densely-spaced trees, forest plantations of native tree species aged 25 to 45 years produce persistent dry-season streamflow deficits exceeding 50% relative to native old-growth forests (Perry and Jones, 2017). In addition, vegetation cover transition can greatly affect evapotranspiration and consequently long-term water balance responses at the watershed scale (Naranjo et al., 2011). Overall, the landscape scale effects of forest cover and management on hydrology depend upon the spatial arrangement of forest stands, which vary in age, density and species composition.

Thinning

Studies of how forest stand conditions (density, species, age) affect water use are typically conducted at the scale of individual trees or small forest stands. Stand-level transpiration is higher in stands with greater stem density (e.g., Whitehead et al., 1984). Thinning reduces interception and transpiration, and consequently increases soil moisture and leaf water potential. It also increases water availability benefitting growth of dominant trees (Nnyamah and Black, 1977; Bréda et al., 1995; Lechuga et al., 2017). However, thinning can increase soil evaporation due to more exposure of soil surface after thinning, which may partially offset the water saving from thinning. In general, canopy conductance (stand level transpiration) increases with leaf area when soil moisture is not limiting, but vapour pressure deficit and soil moisture deficits can limit transpiration (Granier et al., 2000).

Harvesting

Timber harvesting removes trees and causes substantial changes in evapotranspiration which in turn alter water yield from a watershed (le Maitre et al., 2015). Various literature reviews based on experimental studies of small paired-watersheds have shown that harvesting operations reduce evapotranspiration and consequently increase annual streamflow (e.g., Bosch and Hewlett, 1982; Andréassian, 2004; Brown et al., 2005), even though there are large variations in changing magnitudes of streamflow. Several recent reviews based on large watersheds (Li et al., 2017) or both small and large watersheds (Zhang et al., 2017) also reach similar conclusions.

Timber harvesting can significantly alter other components of streamflow (Li et al., 2018; Zhang and Wei, 2013). For example, in northwest North America, forest harvesting increased large flood events, and the effects persisted for multiple decades (Jones and Grant, 1996; Jones, 2000; Moore and Wondzell, 2005). Forest roads shorten flow path lengths and advance peak flow timing in steep forest lands, permanently modifying streamflow response (La Marche and Lettenmaier, 2001; Wemple and Jones, 2003). Forest harvest affects snow accumulation and melt, which in turn increases the magnitude of extreme rain-on-snow floods (Harr, 1986; Jones and Perkins, 2010), and associated landslides, which reduce water quality (Wemple et al., 2001).

4.2.2.2 Plantations

Plantation forests are becoming increasingly common and represent approximately 7% of the world's total forest area (Payn et al., 2015; FAO, 2015). Highly managed conditions, which include stand fertilisation, thinning, regular tree spacing, genetically improved growing stock, controlled burning and other practices, are designed to increase the growth rate and wood quality (Fox et al., 2004). Management practices increase growth by maximizing leaf area and growth efficiency (Waring, 1982). Increased leaf area can increase water demand by trees (Scott et al., 2004).

Pine plantation in South Africa
Photo © Mark Gush

Numerous studies, many in the form of paired catchment experiments, have shown conclusively that plantations of introduced fast-growing tree species generally consume more water than natural vegetation types such as native forests, grasslands or shrublands, and thus reduce water yield (streamflow) from reforested/afforested catchments (Bosch and Hewlett, 1982; Farley et al., 2005; Jackson et al., 2005; Amazonas et al., 2017). This has led to a focus on the 'blue' versus 'green' water trade-off (Calder et al., 2007; Cristiano et al., 2015). Zhang et al. (1999, 2001) illustrated how the range of evapotranspiration from grasslands differed relative to that of a plantation forest, along a rainfall gradient. The case of Eucalypts in South Africa has been particularly well studied (Dye and Versfeld, 2007; Scott and Prinsloo, 2008), and induced specific policy responses (see Chapter 7). In spite of higher water consumption, plantation forests with higher productivity may have greater water use efficiency (WUE) (Gyenge et al., 2008). However, Moore et al. (2011) showed that site condition is the most important factor for WUE in monoculture and mixed-species Douglas-fir (*Pseudotsuga menziesii*) and red alder (*Alnus rubra*) stands. Estimation procedures to compare the expected impact of plantation forestry against a baseline of natural vegetation have been developed in South Africa (Gush et al., 2002; Gush, 2010), and similar calculations have been used to estimate effects of removal of invasive exotic species from riparian zones (Dzikiti et al., 2016) and expanding rubber plantations in SE China (Guardiola-Claramonte et al., 2010).

Monoculture plantations also have less biodiversity compared to natural stands (Brockerhoff et al., 2008) which can 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 stream flows, tree mortality may also decrease water quality (Hibbert, 1965; Swank et al., 2001).

4.2.2.3. Forestation

Forestation (used here as a generic term to reflect any increase in tree cover, regardless of methods applied on prior land use), depending on what it replaces, the species used and the approach taken, can contribute to improving water quality and quantity. For example, in the northeastern United States, much of the Allegheny Mountain range was harvested in the early 20th century (Cleland, 1910). This loss of forest area drove both an increase in streamflow and a severe deterioration of water quality. Having recognised the forest area problem, much of the region was placed under strict protection to encourage restoration (natural regeneration) and prohibit cutting. A century later, the region is now again covered in mature forest and supplies New York City with some of the highest quality drinking water in the US (NY EPA, 2015). More recently, China implemented a 'Greening China' initiative (Box 4.2). Over a decade, tens of millions of ha of forest were planted to stabilise soil and improve drinking water standards (Cao et al., 2011). An adverse side effect of this practice has been reductions in groundwater tables in areas of planted forest and competition between farmers and foresters for limited water resources (Cao et al., 2011). Policymakers thus need to choose between the benefit of reduced dust storms with revegetation, and reduced water availability. Reforestation serves as a robust control of both water quantity and quality, but the choice of species and methods, and clarity of objectives are essential to overall success (Mansourian et al., 2017).

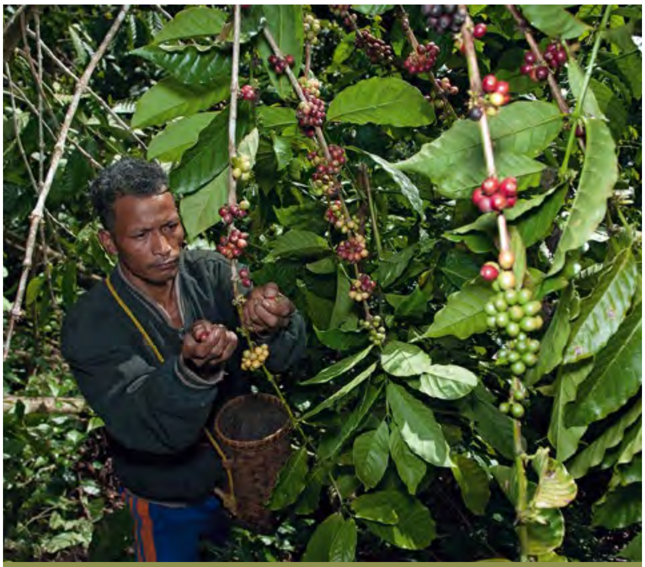
Box
4.2

Reforestation/afforestation programmes in China

Large-scale deforestation before the 1980s in China had caused serious environmental problems including significant soil erosion, decline of land productivity, severe loss of wildlife habitat and biodiversity and various environmental hazards (e.g., floods, droughts, sandstorms) (Ran et al., 2013). In order to improve environmental conditions and relieve poverty, China has launched a series of major ecological stewardship programmes since the 1980s (Ran et al., 2013; Zheng et al., 2016). The large-scale flood occurring in the Yangtze River basin in 1998 (Wei et al., 2008) acted as a further wake-up call to China to recognise the importance of forest protection and reforestation. These stewardship programmes include the Natural Forest Protection Programme, the Grain for Green Programme, the Beijing and Tianjin Sandstorm Source Control Programme, the Three North and the Yangtze River Basin Shelter Forestation Programme and the Grassland Restoration Programme (Ran et al., 2013; Zheng et al., 2016). Recently, to combat climate change impacts, China has pledged to raise forest cover to 20% through the afforestation of 40 million hectares by 2020, while increasing timber volume by 1.3 billion m^3 concerning 2005 levels (Ahrends et al., 2017). The implementation of all those programmes has increased forest cover significantly while having positive environmental consequences (e.g., reduced soil erosion and sediments, increased carbon stocks, improved flow patterns) in China (Ran et al., 2013). The country has also derived some relevant lessons regarding the application of inappropriate tree species in some locations facing water shortages, as those resulted in a reduction of streamflow and groundwater levels, and eventually caused tree dieback (Xu, 2011).

4.2.2.4 Agroforestry

Agroforestry can significantly improve water infiltration, water productivity and nutrient status (Ong et al., 2014; Zomer et al., 2016). Tree litter enhances soil organic matter content, which in turn increases soil water holding capacity, offsetting the higher water use of the trees and the crops (Mutegi et al., 2008). Since trees and annual crops draw most of their water from different layers of soil, there is rarely direct competition (Bayala et al., 2008). Agroforestry systems can redistribute soil water belowground and along slopes (Wu et al., 2017). Trees on farms can mitigate the effects of weather extremes on crops such as droughts, heat waves and heavy rain. The tree roots in agroforestry systems are also able to take up nitrogen, phosphorus and pesticide residues, as well as heavy metals, and therefore improve groundwater and downstream water quality (Pavlidis and Tsihrintzis, 2018). The tree components of agroforestry systems stabilise soils against landslides, raise infiltration rates to limit surface flow during the rainy seasons and increase groundwater release during the dry seasons, which can help crops to cope with drought and flood risks under future climate change (Ma et al., 2009; van Noordwijk et al., 2015). Appropriate agroforestry species can provide fodder and shade for animals while providing organic fertilisers for annual crops during the rainy season (Boffa, 1999).



Coffee is a commodity of the Wae Rebo people in East Nusa Tenggara, Indonesia. Their coffee plants are directly adjacent to natural forests

Photo © Aulia Erlangga/CIFOR

4.2.2.5 Urban and Peri-Urban Forestry

The majority of the global population now lives in cities, at 54.5% in 2016, and typically reaching around 80% in developed nations (United Nations, 2016). The global number is expected to reach 60% by 2030, with the large majority of that population growth occurring in Asia and Africa in rapidly expanding cities (United Nations, 2016). This represents a dramatic demographic change from a population that was just 10% urban at the start of the twentieth century. While cities only represent approximately 3% of the terrestrial surface of the planet, they have global environmental effects (e.g., carbon emissions) and place high demands for ecosystem services within cities (e.g., recreation), adjacent to them (e.g., water supply), and across the world (e.g., food, consumer goods) (Millennium Ecosystem Assessment, 2005; Grimm et al., 2008). Perhaps the most substantive adverse effects of urbanisation on water quality and quantity are due to the increased amount of impervious surface cover in urban watersheds (Shuster et al., 2005).

The ecosystem services provided by urban forests and peri-urban forests have been the subject of a growing body of research (e.g., Vailshery et al., 2013; Duinker et al., 2015; Sanusi et al., 2017). One of the key ecosystem functions and services that urban trees and forests perform is the attenuation and infiltration of urban stormwater during precipitation events. Trees in cities mitigate stormwater runoff in three ways: physically, by intercepting and holding rainwater in their leaves and branches; chemically, through transpiring and reducing soil moisture (Chang and Li, 2014); and by increasing soil porosity mechanically though root expansion and movement (Bartens et al., 2008). For example, in the US, trees save municipalities approximately USD 400 billion a year by reducing total volumes of water destined for treatment and the need for grey infrastructure (e.g., pipes) and stormwater retention (Lerner and Poole, 1999). In addition to cost savings, trees improve urban water quality by reducing sediments and particulate pollution (Sanders, 1986).

Municipalities are looking to green infrastructure solutions that combine built environments with vegetation (Seitz and Escobedo, 2011). For instance, bioretention installations, permeable pavements, and structural soil cells are increasingly using technologies for stormwater management that also provide sufficient soil volumes and irrigation for trees (Scholz and Grabowiecki, 2007; Ow and Ghosh, 2017). Such urban greening initiatives help to simultaneously provide necessary conditions to establish and grow trees in difficult urban settings while also mitigating the adverse effects of urbanisation on hydrological processes. Sustainably-managed urban and peri-urban forests also represent green infrastructure that can play a central role in helping cities to adapt to the changing climate (Brandt et al., 2016).

4.3 Anticipated Changes to Forests, Hydrology and Partitioning for the Local and Global Scales

4.3.1. Climate Change and Future Forest Hydrology

Climate change will likely lead to an intensification of the hydrologic cycle in places where vegetation water use is currently energy-limited, but can elsewhere lead to a net drying effect (Huntington, 2006; Cook et al., 2014; Burt et al., 2015). More extreme precipitation regimes will imply a greater need for the flow-modulating effects of vegetation (Knapp et al., 2008), with flood probabilities increasing. Climate change may also alter forest structure and species composition, and forest cover may extend to higher elevation (see Box 4.3), which may mitigate or exacerbate direct effects of climate change.

Rising atmospheric CO_2 concentrations can increase forest growth and may increase evapotranspiration (Wramneby et al., 2010) particularly in regions where there is no significant water or nutrient shortage (Holtum and Winter, 2010). However, rising atmospheric CO_2 concentrations can also induce a partial closure of vegetation stomata and thus suppress evapotranspiration and increase runoff (Gedney et al., 2006**).** The effects of rising atmospheric CO_2 concentrations on evapotranspiration, tree growth and runoff are debated (Hickler et al., 2008; Norby et al., 2010; Norby and Zak, 2011; Silva and Anand, 2013).

At the same time, forests are important for sequestering atmospheric $CO_{2.}$ Many studies have shown that old-growth forests and old trees can continue to accumulate carbon in vegetation and in soils, while harvesting old-growth forests results in net carbon release (Harmon et al., 1990; Zhou et al., 2006; Luyssaert et al., 2008; Stephenson et al., 2014).

Box
4.3

Potential impact of rising treelines on water resources

One observable result of global warming has been the northward (in the northern hemisphere) and upward shift of tree lines in many mountain areas, and in ecotones ranging from tropical rainforest to alpine regions (Gehrig-Fasel et al., 2007; Pennisi, 2013; Carboni et al., 2018). However, there have been comparatively few studies that have specifically addressed the potential impact of climate change on water resources, coupled with rising treelines (Koeplin et al., 2013). Shifting treelines may have unexpected impacts on the watershed. Koeplin et al. (2013) found that in the Swiss Alps, increased forest cover due to rising treelines had a minor, but seasonally variable effect on evaporation and soil moisture, and a negligible effect on annual run-off. Their study suggested that all water balance components were primarily determined by glacial melt and climate change. These findings are opposed to the ones of Duan et al. (2017), in the slopes of the Da Hinggan mountains of Northeast China, where changes in forest cover and shifts in tree species were more important than permafrost thaw or climate change in regulating water supply. In fact, the interactions with retreating glaciers may even lead to counterintuitive consequences such as increased forest fires driving a downwards shift of the treeline on Mount Kilimanjaro (Hemp, 2005).

Such interactions are likely to be location and species specific. For example, if winter tundra is invaded by a deciduous forest, there will be minimal changes in the albedo. However, its conversion to coniferous forest could potentially exacerbate global warming, due to both their intrinsically low albedo and interaction with snow cover. Whereas snow remains on the ground in both treeless areas and deciduous forests, it melts under conifers. Expansion of a coniferous treeline could impact water resources, raise global temperatures – and thus lead to a further upward march of global treelines (Grace et al.. 2002).

Extrapolations from dendrochronological data indicate that treelines could rise in elevation by 140 to 700 m within a century (Grace et al., 2002). There is thus likely to be a major expected impact of expanding forests on water resources, but we currently lack sufficient evidence in most regions to predict the direction and magnitude of these changes.

On a global scale, evapotranspiration rose from 1982 to 2008 (Jung et al., 2010), although the changes in evapotranspiration are variable among regions. The southern hemisphere – especially in most parts of Australia, East Africa and South America – saw a reduction in evapotranspiration while regions such as China and southern India are characterised by increasing evapotranspiration (Jung et al., 2010). Factors including soil moisture, stomatal closure resulting from rising CO_2 concentrations, land-use change, or declining wind speed all may cause evapotranspiration changes (Piao et al., 2007; McVicar et al., 2012; Rowland et al., 2015). It is important to distinguish the effects of climate change versus land use change on hydrology (Box 4.4).

Box
4.4

Overview of tools available for separating climate and land use change as drivers of hydrological change

Forest change (or land cover change) and climatic variability are two major drivers that influence change in watershed hydrology in forest–dominated watersheds. Quantifying their relative contributions is important to fully understand their individual effects, particularly in large watersheds or landscapes and various tools have been developed in the past few decades. After reviewing several studies, Wei et al. (2013) suggested eight methods for separating relative contributions of climate and land cover change to annual streamflow. These techniques can be broadly classified into statistical and modelling categories. They include hydrological modelling, trend analysis, double mass curves, quasi-paired watershed method, sensitivity-based approach, simple water balance, time trend method and Tomer-Schilling framework. Because each method or technique has its own strengths and weaknesses, combining two or more methods is a more robust approach than using any single method alone (Wei et al., 2013). Some studies include 'engineering measures' as additional explanatory factors beyond land cover change (degradation or restoration) and climate variability (Ma et al., 2014).

The majority of the above-mentioned methods is applied on an individual watershed (particularly large-sized watersheds: >1000 km2) where long-term climatic and hydrological data are available. Interestingly, separating relative contributions is rarely done in paired watershed experimental (PWE) studies where the exclusive focus is on assessing the effects of forest changes on hydrology.

Extreme drought is associated with water stress and tree mortality (Bréda et al., 2006). Trees respond to drought by shifting the allocation of carbon from foliage to roots (Doughty et al., 2014). Drought also influences the hydrologic function of the soil (Gimbel, 2016). Forest die-off from drought and heat stress has occurred around the world and is expected to increase with climate change (Anderegg et al., 2013). In northern and western Europe, where soil moisture may not be limiting, increased atmospheric CO_2 concentrations and warmer temperatures are expected to increase forest growth, whereas in southern and eastern Europe increasing drought and fire risks are expected to reduce forest productivity (Lindner et al., 2010). In southern European forests, progressive crown defoliation occurred from 1987 to 2007 apparently in response to increased water deficit (Carnicer et al., 2011). It has been argued that tall trees of old-growth forests are at the greatest risk of mortality due to moisture stress (McDowell and Allen, 2015). However, in unmanaged old forests in the western US, non-catastrophic mortality rates increased rapidly in recent decades, targeting small trees (van Mantgem et al., 2009).

Analyses of long-term records at 35 small watersheds (0.01 to 1 km^2) in the US and Canada indicate that climate change effects on streamflow are not as clear as might be expected, apparently because of ecosystem processes and human influences (Jones et al., 2012). Although air temperature increased at 17 out of 19 sites with 20 to 60-yr records, climate trends were directly related to streamflow trends at only seven sites, and all of these involved changes in ice and snow. At other forest sites undergoing warming, other factors such as past forest disturbance and forest succession mimicked, exacerbated, counteracted, or masked the effects of climate change (Jones et al., 2012).

Interannual variability of climate significantly influences interannual variability of streamflow at forested headwater sites. For example, in the above-mentioned North American dataset (Jones et al., 2012) streamflow was significantly correlated with the El-Niño Southern Oscillation (ENSO), the Pacific Decadal Oscillation (PDO) and/or the Northern Atlantic Oscillation (NAO) at 26 of 30 forested headwater reference watersheds.

Forested sites differ in their sensitivity to interannual climate variability. An experimental analysis of long-term experimental watersheds in Canada and the US was conducted over 5-year cool and warm periods to test whether changes in dryness were associated with consistent responses of water yield (Creed et al., 2014). Alpine sites, whose hydrology was dominated by water stored in snow and ice, showed the greatest sensitivity to warming, and any warming led to increased water yields.

These studies indicate that forest dynamics, including legacies of past disturbance and forest management, as well as forest succession, produce a wide range of forest hydrologic responses to climate change at individual sites. For example, in the northern hardwood forest of Hubbard Brook (US), climate change effects on ecosystem structure and function, and hydrology appear to be modified by interactions with a spatially variable history of land use and a wide range of current human activities and concurrent environmental changes (Groffman et al., 2012). At Hubbard Brook, both air temperature and precipitation have increased, but winter precipitation has increased less. As a result of reduced snowpack accumulation, snowmelt-induced peak flows in spring have declined (Campbell et al., 2011) and have occurred earlier (Hamburg et al., 2013). In contrast, both winter and summer streamflows have increased. In winter, the increase is due to reduced storage of precipitation in the snowpack, whereas in summer (typically a low-flow season), streamflow has increased due to increasing precipitation and declining evapotranspiration (which has shown slight but significant declines since 1959). The cause of the decline in forest evapotranspiration is not known but may result from changes in vegetation composition, structure, or productivity, or forest response to increasing atmospheric carbon dioxide concentrations, or other factors (Groffman et al., 2012).

In mixed oak-hickory hardwood forests of the southeastern US (Coweeta), forest succession has responded in unexpected ways to long-term changes in climate, perhaps reflecting long-term forest responses to burning, grazing, and logging more than one hundred years ago. At Coweeta, air temperature, drought severity, and precipitation extremes have increased since the late 1970s (Laseter et al., 2012). Annual water yield increased by as much as 55% from 1938 to the mid-1970s in some watersheds, which were undergoing forest succession after logging in the early 1900s (Caldwell et al., 2016). However, from the 1970s to 2013, water yield declined by 22%, associated with a shift in dominance from xerophytic oak and hickory tree species to mesophytic tree species including red maple (*Acer rubrum)* and tulip poplar (*Liriodendron tulipifera*) (Caldwell et al., 2016).

Forest vegetation succession provides strong negative feedbacks that make permafrost resilient to even large increases in air temperatures. However, as seen in boreal forests of Alaska, climate warming is associated with reduced growth of dominant tree species, plant disease and insect outbreaks, warming and thawing of permafrost, drying of lakes, increased wildfire extent, and increased post-fire recruitment of deciduous trees. These changes have reduced the effects of upland permafrost on regional hydrology (Chapin et al., 2010). Surface water, in contrast, provides positive feedbacks that make permafrost vulnerable to thawing even under cold temperatures (Jorgenson et al., 2010). In watersheds with low permafrost, base flow is higher, and annual water yield varies with summer temperature, whereas in watersheds with high permafrost, annual water yield varies with precipitation (Jones and Rinehart, 2010). With climate warming and loss of permafrost, stream flows will become less responsive to precipitation and headwater streams may become ephemeral (Jones and Rinehart, 2010).

Mirror Lake near Hubbard Brook Experimental Forest (West Thornton, New Hampshire, US)

Photo © Richard Guldin, GuldinForestry.com

4.3.2. Forest Management, Forest Cover Changes and Future Forest Hydrology

Anticipated future changes in forest cover and forest management are diverse. They may include expansion of intensive plantations, expansion of agriculture, selective logging, loss of riparian forest, and loss of urban trees (see Chapter 6). Deforestation has been high, especially in the tropics, since records of global forest cover began. From 2000 to 2012, globally 2.3 million km^2 of forest were lost, and 0.8 million km^2 of new forest were gained (Hansen et al., 2013). Intensive forestry practised within subtropical forests resulted in the highest rates of forest change globally (Hansen et al., 2013). Boreal forest loss due largely to fire and forestry was second in absolute and proportional terms.

4.3.3. Changes in Natural Disturbance Regimes and Future Forest Hydrology

Anticipated future changes in forest disturbance regimes include more wildfire, more frequent and intense storms, and spatial changes in insect/pathogen outbreaks. Disturbances from wind, bark beetles and wildfires have increased in Europe’s forests throughout the twentieth century (Schelhaas et al., 2003). For example, the mountain pine beetle outbreak in British Columbia (Canada) produced changes in carbon cycling equivalent to approximately 75% of the average annual direct forest fire emissions from all of Canada during 1959–1999 (Kurz et al., 2008). Climate change is expected to interact with forest disturbance regimes (Dale et al., 2001). Models predict a lengthened fire season, and significant increases in the area experiencing high to extreme fire danger in both Canada and Russia (Stocks et al., 1998).

4.3.4. Forest Succession and Future Forest Hydrology

Anticipated future forest succession processes will include changes in forest age, structure and species composition that may increase or reduce water yield and water storage. Hydrologic responses to drought can be either mitigated or exacerbated by forest vegetation depending upon vegetation water use and how forest population dynamics respond to drought (Vose et al., 2016). Current species distribution models of forests cannot accurately predict changes in species composition (Scherrer et al., 2017). Tree species differ in canopy- and leaf-level stomatal conductance response to vapour pressure deficit, so ecophysiological differences, as well as structural differences among species influence evapotranspiration (Ford et al., 2011).

4.3.5. Anticipated Changes in Water Partitioning from the Local to the Continental Scale

Deforestation tends to increase runoff, and re- or afforestation decreases runoff (Li et al., 2017). Tropical deforestation results in warmer, drier local conditions (Lawrence and Vandecar, 2015). Climate model simulations of Amazonia indicate that deforestation was associated with reduced rainfall (Spracklen and Garcia-Carreras, 2015). Forest cover in Amazonia was not correlated with precipitation at the local scale (1 to 15 km) but it was positively correlated with measured precipitation at the regional scale (30 to 50 km) (Debortoli et al., 2017). Thus, in regions where precipitation is declining, forest thinning or reductions in forest cover may help counteract declining runoff. Such approaches may also reduce evaporation and, perhaps, regional precipitation, but little is known of these potential effects.

Climate change implies an increased need for hydrological resilience in our landscapes (e.g., limiting floods and withstanding drought) (e.g., Hatcher and Jones, 2013). Forest succession may reduce floods and resist droughts, but human and natural disturbance and ongoing climate change continually alter forest hydrology. Under such circumstances, forest management for the future should focus on managing in the face of uncertainty (Millar et al., 2007). Forest hydrology will continue to respond to multiple system drivers of change. Therefore, decisions should not be based on expected responses to single factors (Lindner et al., 2014). The varied responses of forests and forest hydrology to change over the past 50-100 years underscore the importance of incorporating stochastic variability into projections of future ecosystem condition (Daniel et al., 2017). Benefits of multi-aged management systems that maintain a large proportion of retained mature trees while using thinning to create spatial heterogeneity and enhance structural and compositional complexity are to be appreciated (D’Amato et al., 2011).

4.4 Data Needs and Knowledge Gaps

Research is urgently needed on a variety of topics. These include basic data on runoff and precipitation in ungauged watersheds, making data publicly accessible, and improving models and data products.

There is a general lack of hydrological and meteorological monitoring, particularly in developing countries. Increasingly, studies are based on remotely-sensed data and process-based models, with insufficient calibration or validation as reliable groundtruthed data are scarce. The intricate linkages between terrain, climate, forest conditions, disturbances (either natural or human-made) and hydrological processes often prevent adequate transferability of findings from well-studied watersheds to those with limited data. In many cases, empirical models of hydro-ecological interactions cannot be extended outside the site where they were created (Kimmins et al., 2010). There is also a widespread lack of measurements of isotopes of water, which are central to estimates of water residence times in soil, vegetation, rivers and the atmosphere. The distinction between ‘data’ and ‘models’ is increasingly blurred, as the interpretation of any measurement is itself dependent on ‘models’, with model assumptions often less explicit than would be desirable. The toolbox for exploring trees as history books (Box 4.5) and using isotopes to reconstruct water sources for plant production offers hope that past changes in data scarce areas can still be unravelled (Box 4.6).

Improved sharing of data relevant to forests and water would greatly enhance research and related policy discussions. Repositories that archive and make freely available data on precipitation and runoff are rare. One valuable example is the Climate and Hydrology Database Projects (CLIMDB/HYDRODB), https://climhy.lternet.edu, supported jointly by the US Forest Service, US Geological Service, and US National Science Foundation's Long-Term Ecological Research programme.

The fraction of local precipitation that is derived from terrestrial sources (and as such dependent on upwind land use change), as well as the fraction of local evapotranspiration that will return as rainfall over land rather than over oceans varies from 0 to nearly 100%, depending on location (Box 4.6). This location-dependence is a major challenge for those who seek generic (place-independent) truths, as well as for those who want to adopt concepts and water-based legislation from other locations (see van Noordwijk et al., 2014 for comments on the transfer of South African legislation to East Africa in this respect). Given the abundance of current studies on the Amazon and indications of the geography of atmospheric recycling, studies in Africa (e.g., on the role of the Congo Basin; links between White and Blue Nile precipitationsheds) and Asia (including precipitation patterns in China) are urgently needed. The relative importance of forests vis-à-vis wetlands and irrigation agriculture in the time-space patterns of evapotranspiration need to be further clarified. At the mechanistic level, the 'proof of principle' evidence on rainfall triggering by agents of biological origin (including phyllosphere bacteria, fungal spores and volatile organic compounds) needs to be coupled with inventories of which vegetation types and land uses are the major sources of atmospheric moisture, across the main 'prevailing winds' patterns of the world.

Box
4.5

Trees as history books reflecting past climate variability

Tree rings provide a valuable archive of past climate, and have also been widely used to reconstruct hydrology. Tree rings form the basis for multi-century reconstructions of precipitation, temperature and river discharge. They have also been used to understand historical dynamics of the water table, and to reconstruct flood history using the dating of flood scars on trees (Gottesfeld and Gottesfeld, 1990). Such reconstructions provide valuable baselines for understanding recent hydrological events and trends, and inform water management strategies that rely on understanding the natural variability in precipitation, streamflow and flood frequency.

The recent (2017) extreme California drought was widely reported in the media, and resulted in significant socioeconomic impacts, notably for irrigation-based agriculture. However, short instrumental records made this event difficult to place in a broader context. The development of long tree-ring-based reconstructions of drought, precipitation and snowpack have been used to demonstrate that this drought event was exceptional in nature (Griffin and Anchukaitis, 2014; Belmecheri et al., 2016). For example, long tree ring chronologies from blue oak trees revealed that the 2015 record low was not only exceptional in terms of the observational record, but was unprecedented over the last 500 years (Belmecheri et al., 2016). However, this extreme drought was not just the result of low precipitation linked to ENSO, as was popularly reported. Combining independent tree-ring-based reconstructions of precipitation and temperature, Griffin and Anchukaitis (2014) showed that the precipitation anomalies associated with the drought were unusual, but not exceptional in the context of the last 1,200 years. Instead, tree-ring-based analysis revealed that the recent extreme drought resulted from the combination of low precipitation and record high temperatures. Higher temperatures increase evapotranspiration, exacerbating precipitation deficits and are also associated with record low snowpack depths in the Sierra Nevada mountains (Belmecheri et al., 2016).

In Africa, a recent dendrochronology study in the Blue Nile established a 360 year climate record for an area where other data are scarce (Mokria et al., 2017). The ring-width chronology captures climate signals across the whole northern Ethiopian highlands and parts of the Sahel belt. This dataset confirms the existence of large-scale atmospheric teleconnections to dry/wet changes with substantial geopolitical consequences: land cover change in country A affects rainfall in countries B and C through changes in the recycling of atmospheric moisture from evapotranspiration. The chronology highlights inter-annual, decadal and multi-decadal variations in large scale atmospheric circulation patterns and teleconnections.

In addition to studies that have used paleoclimatic reconstruction to understand hydrologically relevant climate variables, other tree-ring based reconstructions have focused directly on streamflow. Understanding natural streamflow variability is important where this resource is used as part of socioeconomic activity. For example, in British Columbia (Canada) small mountain catchments are used for hydropower generation, agriculture and underpin an important local salmon fishery industry. However, historical river gauge data extends to only a few decades. This potentially limits the effectiveness of water management strategies, which rely on accurate data on natural variability in streamflow. Coulthard et al. (2016) used a multi-century tree-ring-based reconstruction of summer streamflow to demonstrate that while recent streamflow droughts had important socioeconomic impacts in British Columbia, they were not unusual in a historical perspective, and 16 other identified events since 1658 were more severe than any event recorded in the instrumental period. Analysis of the reconstruction indicated that extreme droughts occur, on average, more regularly in these catchments than would be indicated by an analysis of the shorter instrumental records. This example illustrates how the management of water resources at the catchment level can be aided by a more complete understanding of natural streamflow variability, reconstructed using tree-ring chronologies.

Box **4.6**

Stable water isotopes and sources of precipitation

Stable water isotopes of oxygen, hydrogen and carbon are valuable tracers of the origin and history of air masses. Rozanski et al. (1993) summarise the main processes: the origin of air masses, continental and altitude effects and rainfall amounts. The combination of improved measurement and modelling of water vapour isotopic composition opens the door to new advances in our understanding of the atmospheric water cycle (Galewsky et al., 2016). Molecular differences in common isotopes cause fractionation during most phase transitions: heavier isotopes (HDO) preferentially condense, whereas lighter isotopes ($H_2{}^{16}O$) preferentially evaporate. Evaporation from the ocean surface and condensation during transport deplete both the deuterium and oxygen content of water vapour relative to its source (Farquhar et al., 2007).

Analysis of the stable isotope composition of precipitation can directly infer the fraction of recycled moisture and can even differentiate the precise shares of the transpiration and evaporation components. Isotopic evaluation is based on the assumption that the stable isotope composition of the vapour that produces precipitation is an isotopic mixture of vapours sourced from advection (convergence of the moisture advected into the region or basin by winds), evaporation, and transpiration. Each of these sources has its own unique isotopic composition (Clark and Fritz, 2013). Based on a three-component isotopic mixing model, Wang et al. (2016) were able to determine that the proportional contribution of recycled moisture relative to local precipitation, at the large oases of Urumqi, China, is approximately 16.2% (surface evaporation and transpiration are about 5.9% and 10.3%, respectively). Isotopic fingerprints in atmospheric moisture have unambiguously identified rainforest transpiration as the primary moisture source for shallow convection during the dry-to-wet season transition in the Amazon (Wright et al., 2017).

Tree rings and their stable isotope analysis can provide supporting evidence for the positive effects of forests on the water cycle at micro and larger scales. Williams et al. (2011) measured annual ^{18}O ratios of *Juniperus procera* from northern Ethiopia and found a decline in the proportion of precipitation originating from the Congo Basin during the past half century and increasing precipitation variability and drought frequency over the Greater Horn of Africa. Brienen et al. (2012) demonstrated that oxygen isotope ratios in tree rings in Tropical Cedrela accurately record the isotopic composition of meteoric precipitation during the wet season ($\delta^{18}O_{prec}$) and a very strong proxy for annual to decadal scale variation in the amount of precipitation over the entire Amazon basin and for basin-wide river discharge. The record shows significant correlations with $\delta^{18}O$ in precipitation in the central and western Amazon and ice cores in the Andes, indicating that the interannual variation in $\delta^{18}O$ of precipitation contains a spatially coherent signal over large parts of the basin, indicating water vapour recycling of rainwater by vegetation.

4.5 Conclusions

Changes in forest (due to both natural disturbance and human activities) affect how incoming precipitation is partitioned between evapotranspiration and streamflow. In a recent global assessment (Wei et al., 2017) forest changes explained, on average, 30% of annual streamflow variations. Any future global water resource assessment requires consideration of climate, vegetation and their interactions.

Natural disturbance such as wildfire, insect pests, diseases, windthrow, etc. can greatly change various watershed processes and ecological functions in forested watersheds. Although natural disturbance is part of the dynamics of forest ecosystems, catastrophic forest disturbance (e.g., stand-replacing wildfire or large-scale insect pest outbreaks) can cause undesired ecological and economic consequences.

Human-driven changes include silviculture, agroforestry, plantation forestry, restoration and urban and peri-urban forestry. Native forests provide more sustained water yield compared to managed forest plantations. Re-establishment of forests may enhance sustained water yield, but effects vary depending on site conditions, and may require years to decades to be detectable. The role of forests in hydrological regimes and associated watershed functions varies among the regions of Earth.

Climate change is altering hydrological processes directly, and is affecting forests, thereby altering hydrology indirectly. Climate change may also alter forest structure and species composition, which may mitigate or exacerbate direct effects of climate change. Forest dynamics, including legacies of past disturbance and forest management, as well as forest succession, produce a wide range of forest hydrologic response to climate change at individual sites.

Forest management for the future should consider uncertainty. Decisions should not be based on expected responses to single factors but rather multiple ones. Knowledge of responses of forests and forest hydrology to change over the past 50-100 years, highlights the need to incorporate stochastic variability into projections of future ecosystem condition.

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