

Chapter 24

Forest-Water Interactions Under Global Change



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24.1 Introduction

This chapter aims to characterize the effects of global changes on forest-water interactions and water availability to ecosystems and people. It synthesises current understanding of the implications of present and anticipated changes to forests and tree cover for local and global hydrology and provides an overview of contemporary global change processes and their interactions with forests and water. It focuses specifically on natural and human disturbances and their effects on biotic and abiotic properties of forests and their consequences for hydrological processes.

Forests are dynamic on time scales of years to centuries and beyond, as a result of natural and anthropogenic disturbances. Large infrequent disturbances, both natural and human, may significantly modify forest characteristics and post-disturbance forest succession on a seasonal timescale, to decades, and to millenia (Foster et al. 1998), with corresponding effects on associated hydrologic processes. Natural forest

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disturbances can have significant economic and social costs, including those associated with water ecosystem services from forests. Climate change may alter the frequency, intensity, and timing of natural disturbances and forest characteristics (Overpeck et al. 1990; Dale et al. 2001), consequently affecting hydrological processes.

Human disturbances have further exacerbated forest disturbances by natural processes, and forest changes are increasingly evident in global-scale land cover change detection (Fig. 24.1). In the 1970s, there was considerable uncertainty in estimations of global forest cover (Allen and Barnes 1985), but global high-resolution satellite coverage now permits more accurate estimates of forest cover and the density of trees within forests (Hansen et al. 2013). As of 2000, the largest area of forest within climate domains (e.g., boreal zone, tropics, etc.) contained less than 25% tree cover (Fig. 24.1a; Hansen et al. 2013), attesting to the widespread prevalence of open forest types and the roles of human disturbance in reducing tree cover in forest areas.

There has been a net loss of forest cover in the past half-century (Fig. 24.1b). Over this period, more than 800,000 km² of forest area was gained, but 2.3 million km² of forest was lost (Hansen et al. 2013). The area of forest loss was highest in the tropics, and least in the temperate zone, whereas the area of forest gain was similar across all climate domains (Fig. 24.1b).

The locations and estimated rates of forest cover change differ depending on the type of forest and the time period, as well as improvements in the accuracy of forest cover estimates. In the 1970s, as a result of population growth and agricultural expansion, estimated forest cover loss (both area and rate of loss) was especially high in the native tropical rainforest in Africa (Ghana, Cote d'Ivoire, Cameroon, and present-day Democratic Republic of the Congo), Asia (Thailand, Indonesia, Malaysia), and South America (Brazil) (Allen and Barnes 1985). However, in the recent past (2000–2012), the most rapid rates of forest change have occurred as a result of the expansion of intensive plantation forestry, which produces short-term gains in

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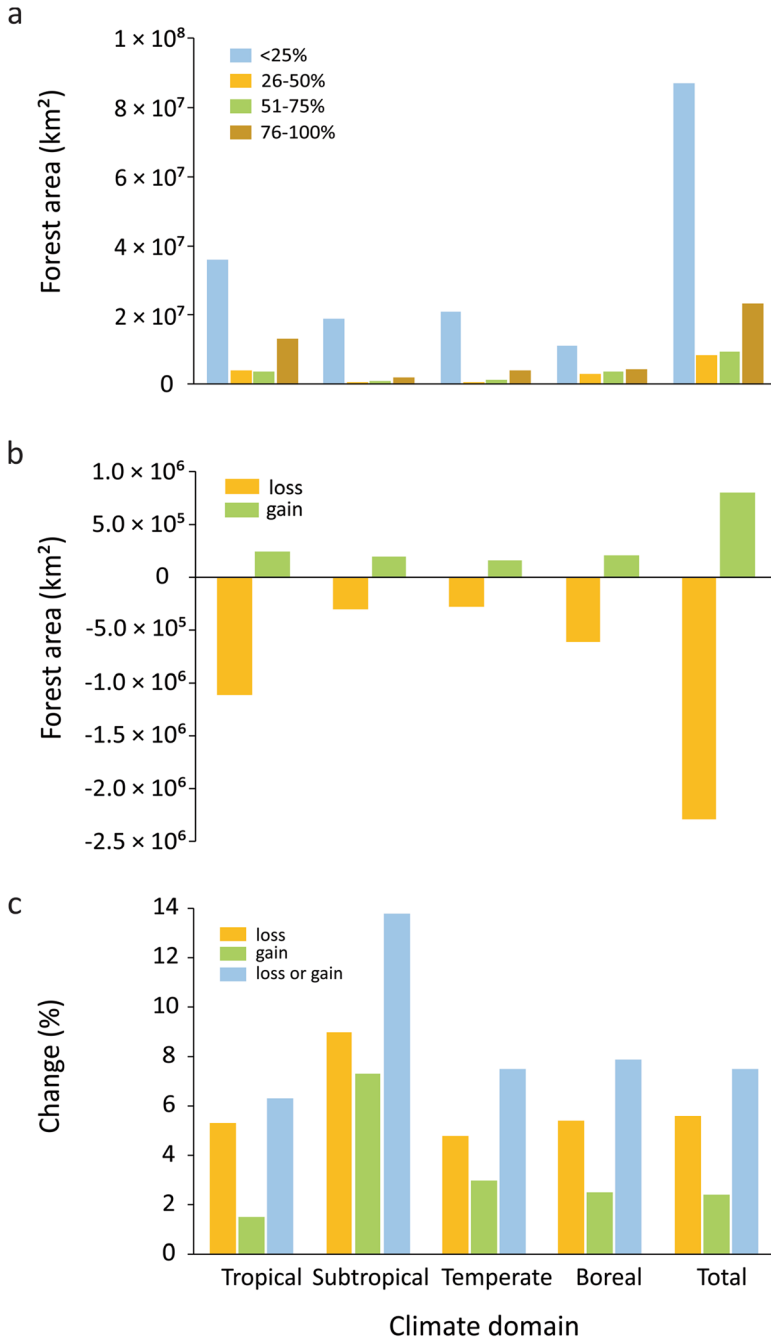


Fig. 24.1 Forest cover area and change from 2000 to 2012. **(a)** Estimated forest area in 2000 by tree density classes in different climate domains. **(b)** Total forest area loss and gain from 2000 to 2012 in different climate domains. **(c)** Change in area of forests with >50% tree cover from 2000 to 2012 (loss defined as loss >50%). (Source: Hansen et al. (2013) High-resolution global maps of twenty-first-century forest cover change. *Science* 342: 850–853. Reprinted with permission from AAAS)

particular types of forest cover, and associated clear-cut. This may in some cases produce forest loss when natural forest is replaced by plantation forests, e.g., in subhumid forests of North America (southeastern United States) and South America (Uruguay, southern Brazil) (Fig. 24.1c; Hansen et al. 2013). At the same time, as indicated earlier, the rates of forest loss in the tropics were two to three times higher than rates of forest gain in the temperate and boreal latitudes (Fig. 24.1c).

These patterns and trends demonstrate that forest change is dynamic and global, with an increasing impact from large-scale forestry plantations that are often monocultures. Forest change affects countries at all levels of economic development and is highly sensitive to global economic, social, and political factors. As a result of forest dynamics and net loss, water ecosystem services from forests are continually changing and being incrementally lost at the global scale.

24.2 Current Changes to Forest-Water Interactions

Drivers and determinants of change operating at a wide range of spatial and temporal scales modify a range of aspects of the climate-vegetation-soil-streamflow system. This section reviews examples of natural and human disturbances and their effects on components of forests (e.g., canopy and leaf area, litter and soil surface, rooting depth, and soil porosity) (Table 24.1) that in turn affect hydrological processes (e.g., precipitation, interception, infiltration, soil moisture storage, percolation, and evapotranspiration) (Table 24.2; Fig. 24.2). These characteristics of forest ecosystems strongly modulate the response of hydrology to forest disturbances (Jones and Post 2004; Peña-Arancibia et al. 2019; Zhang et al. 2019a). The section focuses on five natural disturbance processes (insects and pathogens, wildfire, drought, windthrow, and ice storms) and five human disturbances (silviculture including forest harvest, forest plantations, forestation, agroforestry, and urban and peri-urban forestry) (Fig. 24.2).

24.2.1 *Hydrological Consequences of Natural Forest Disturbance*

Natural disturbances have the potential to significantly alter a range of watershed processes (e.g., water quality, hydrology, channel morphology), as well as ecological functions in forested watersheds. Natural disturbance is intrinsic to forest dynamics (Attiwill 1994; Lertzman et al. 1997), producing spatial heterogeneity and ecosystem complexity and, under certain circumstances, supporting ecosystem

Table 24.1 Short-term effects of selected natural and anthropogenic disturbances discussed in this chapter on forest characteristics that influence water ecosystem services

	Canopy, leaf area	Litter, soil surface	Rooting depth	Soil porosity
Insects and pathogens	X		X	
Wildfire	X	X	X	X
Drought	X	X	X	
Ice storms	X			
Windthrow	X	X	X	
Silviculture, forest harvest	X	X	X	X
Plantations	X	X	X	X
Forestation	X	X	X	X
Agroforestry	X	X	X	X
Urban, peri-urban forestry	X	X	X	X

Table 24.2 Summary of influence of forest characteristics on hydrological processes

	Precipitation	Interception	Infiltration	Soil moisture storage	Percolation to groundwater	Evapo-transpiration
Canopy, leaf area	X	X		X	X	X
Litter, soil surface		X	X	X		X
Rooting depth			X	X		X
Soil porosity				X	X	X

resilience. Catastrophic forest disturbance (e.g., stand-replacing wildfire or large-scale insect pest outbreaks) may, however, cause undesired ecological and economic consequences. Wildfire, insects and pathogens are major natural disturbance agents that may affect global forests at a range of scales (van Lierop et al. 2015).

Disturbance events may be driven by natural or anthropogenic processes. The disturbance regime is a collection of events, with a characteristic frequency, severity, and extent. Natural and human disturbances in forests affect watershed processes. The hydrologic effects of some of the most widely studied natural forest disturbances are described below.

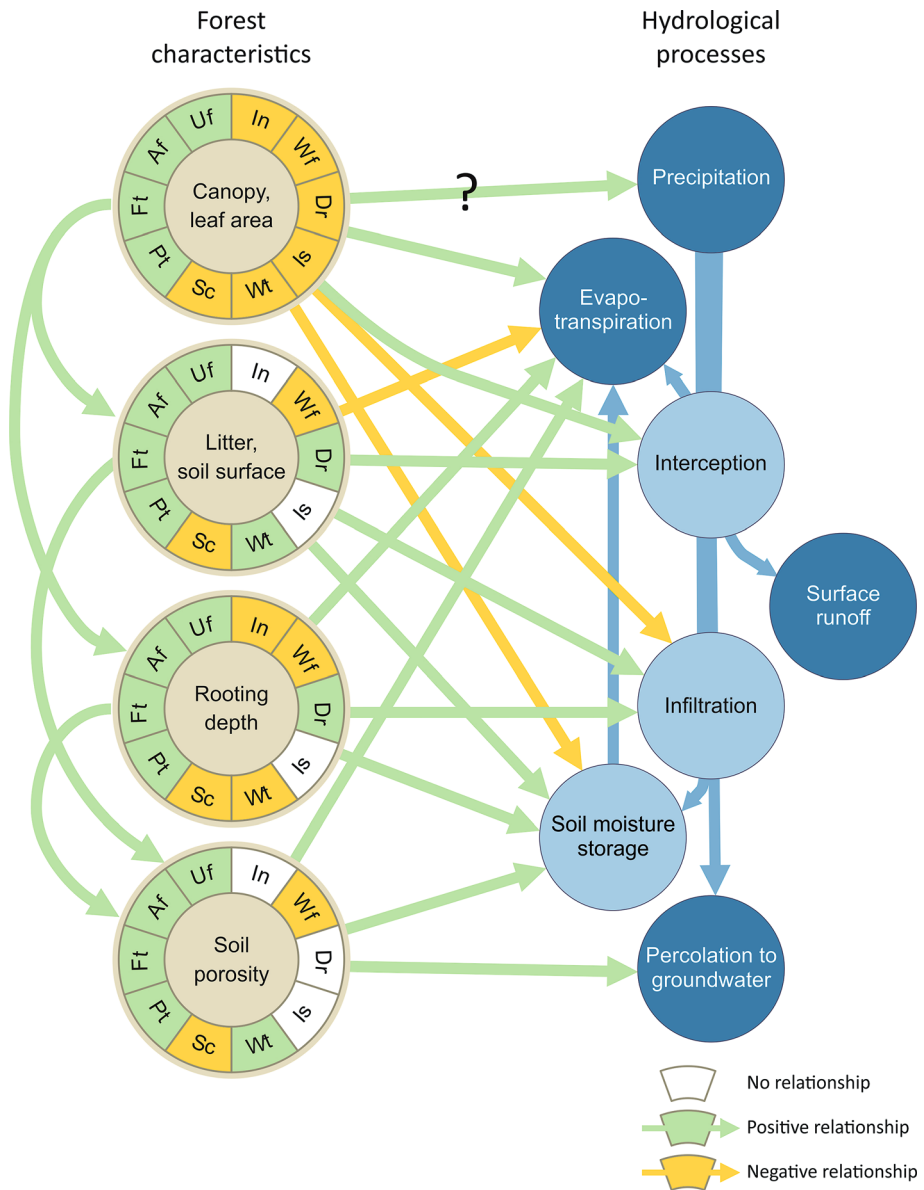


Fig. 24.2 Relationships between selected forest disturbances, forest characteristics, and hydrological processes that influence water ecosystem services. Disturbances are arranged around each forest characteristic; colours of slices indicate no (white), positive (green), or negative (yellow) short-term relationships with forest characteristics. Arrows between forest characteristics indicate relationships between forest characteristics. Arrows between forest characteristics and hydrological processes indicate positive (green) and negative (yellow) relationships with hydrological processes; no arrow indicates no relationship. The question mark (?) indicates that future research is needed to understand the degree and scale of the relationship. Arrows between hydrological processes indicate water transfers. Dark blue circles indicate hydrological processes that are included in water balance; light blue circles indicate hydrological processes that are intermediate. Disturbances: In (Insects), Wf (Wildfire), Dr (Drought), Is (Ice storms), Wt (Windthrow), Sc (Silviculture), Pt (Plantations), Ft (Forestation), Af (Agroforestry), Uf (Urban forestry)

24.2.1.1 Insects and Pathogens

Insects and pathogens are common disturbance agents affecting forests and can significantly influence hydrological processes. A characteristic example is the case of the mountain pine beetle (*Dendroctonus ponderosae* Hopkins) (MPB), a native insect to pine forests of western North America. Unusually hot, dry summers and mild winters, along with forests filled with mature pine, have led to an unprecedented epidemic of MPB in the Rocky Mountains of Colorado and much of British Columbia. Tree mortality from the MPB is caused by larval egg 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. There are three stages to an MPB attack. In the first summer, affected trees stop transpiring, but needle colour does not change, so this is termed a 'green attack'. In the following two to three years, the needles turn red and start to fall; this is the 'red attack' stage. In the third year, trees have died and lost their needles so the stage is termed 'grey attack'. Although changes in canopy colour increase albedo, reducing winter and early spring temperatures (O'Halloran et al. 2012; Vanderhoof et al. 2014), these changes are partially 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 the red attack stage onwards, surviving vegetation makes use of the increase in available resources (i.e., water, light, nutrients), often growing at an enhanced rate (Dhar et al. 2016a). The changes in energy and vegetation caused by an MPB infestation may thus drive hydrological responses dynamically, as the stand moves through the stages of attack and post-attack recovery.

An MPB outbreak serves as a clear example of how pest/pathogen infestation may affect all forest hydrologic processes. Tree mortality following MPB attack reduces foliage cover and density and consequently decreases canopy interception. The more open canopy after an MPB attack speeds snow ablation and advances spring melt by days to weeks compared to unaffected stands (Redding et al. 2008; Winkler et al. 2014). Tree transpiration is reduced following MPB attack, in magnitude proportional to the severity of mortality (Clark et al. 2014). The opening of the forest canopy concurrently increases sun exposure, however, which increases soil evaporation. These competing processes offset each other (Bearup et al. 2014; Biederman et al. 2014), to a degree that 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. This rapid growth of understory also suggests that the warming effects due to albedo change in the grey stage would be short-lived as the albedo change favours cooling. The effects of MPB outbreaks 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; Reed et al. 2014). Salvage logging following MPB infestation significantly increases high flows and advances their timing (Lin and Wei 2008; Zhang and Wei 2014), which can potentially increase flood risk.

Other pest and pathogen attacks on forests may have similar, mixed, and transient effects (Jones et al. 2009; Jones 2011; Adams et al. 2012). For example, water yield and peak flows of large events increased in watersheds where eastern hemlock was lost due to hemlock woolly adelgid infestation (Kim et al. 2017).

24.2.1.2 Wildfire

Wildfire is the most dominant natural disturbance in global forests, particularly in boreal and Mediterranean forests, although fire severity, intensity, and frequency vary according to forest type (Hansen et al. 2013; van Lierop et al. 2015; IUFRO 2018). Wildfires affect millions of square kilometers of land each year, and experience decreases or increases depending on fire suppression, fuel accumulation, and climate change (Westerling 2016; 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 harvesting (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 particular types of fire suppression practices can lead to an accumulation of more fuels, which in turn may increase the chance of more catastrophic fires in the future (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. 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, caused by accumulation in the soil of organics volatilised during the fire, may cause high overland flow rates, especially on steep slopes (Neary et al. 2005), with enough energy to carry freshly fallen litter downhill.

The impact on forest water resources can be highly variable, for several reasons. Intensity, duration, and size all help determine wildfire impacts on tree mortality (Dunn and Bailey 2016; Iverson et al. 2017); it is not always straightforward to distinguish the effects of fire from the tree mortality caused by drought episodes that triggered the fire, but some studies have been able to quantify the relative contribution of both (e.g., van Nieuwstadt and Sheil 2005). The impacts of wildfire on forest water quantity and quality are thus also highly variable (Riggan et al. 1994; Vieira et al. 2015; Hallema et al. 2018). As a rule, streamflow response depends on the extent and severity of wildfire. 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 likely have minimal impact on forest water quality, if the areas are left to regenerate naturally (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 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 (Doerr et al. 1996) 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 may 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 all hydrological processes. Crown fires eliminate above-ground biomass which greatly reduces canopy interception and evapotranspiration (Bond-Lamberty et al. 2009; Montes-Helu et al. 2009) and increases soil evaporation, as soils with altered albedo become exposed to solar radiation. As a result of changes in the relative partitioning of water resources, 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 the kinetic energy of rainfall, limit soil infiltration capacity, and shorten flow paths. 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, spatial and temporal 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, 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).

24.2.1.3 Drought

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).

24.2.1.4 Windthrow

Windthrow, the process by which live trees are toppled by high winds, is a natural process in forests in boreal, temperate, and tropical regions (Bormann and Likens 1979). High winds during severe thunderstorms (Canham and Loucks 1984) and hurricanes (Boose et al. 1994) topple dominant trees in forest stands. In other contexts strong winds (as in typhoons) can lead to defoliation without windthrow, inducing strong hydrologic responses in the short term, but a more rapid recovery (Zhang et al. 2019b). Windthrow releases shade-tolerant tree species when dominant trees are toppled (e.g., Lorimer 1977; Foster 1988). Windthrow increases light to the forest understory and creates spatial heterogeneity of habitats in the form of root

mounds and pits ('pit and mound topography') associated with the root wads of downed trees (Henry and Swan 1974; Carlton and Bazzaz 1998). Forests in certain locations, such as coastal areas, may be especially susceptible to natural windthrow as a disturbance agent (Kramer et al. 2001). Trees in areas exposed to high winds are especially susceptible to windthrow (Foster and Boose 1992), and by creating fresh edges, forest harvest may accelerate windthrow (Sinton et al. 2000).

24.2.1.5 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 supercooled 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 promote bark beetle population increases (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, but as ice storms cause stem and branch breakage and uprooting of trees, it is expected that their effects could be similar to a combination of effects of insects (i.e., dead trees) and windstorms (i.e., uprooted trees). A 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 (Houlton and Driscoll 2011). In 2011, the Hubbard Brook Experimental Forest ice storm experiment produced significant canopy damage and increased canopy openness (Rustad and Campbell 2012).

24.2.2 Hydrological Effects of Human-Driven Forest Changes

24.2.2.1 Silviculture and Forest Harvest

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 significantly 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 (24-hour cyclical) 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). Due to high water use by young, densely spaced trees, forest plantations of native tree species aged 25–45 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. Thinning 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 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). Thinning decreased water uptake for two tree species, especially during dry seasons and dry years, but effects differed between conifer and broadleaf species (Cardil et al. 2018).

Harvesting Timber harvesting removes trees and causes substantial changes in evapotranspiration, which in turn may alter water yield from a watershed (le Maitre et al. 2015). A range of 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; Boggs et al. 2013), although it must be noted that 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) reach similar conclusions.

Timber harvesting can significantly alter other components of streamflow (Zhang and Wei 2014; Li et al. 2018). For example, in northwest North America, forest harvesting increased large flood events, with effects persisting 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, contributing sediment to streams (Wemple et al. 2001). On the other hand, as shown in the southeastern USA, sediment delivery from forest harvest may not significantly affect aquatic biota (Boggs et al. 2015).

24.2.2.2 Plantations

Plantation forests are becoming increasingly common and represent approximately 7% of the world's total forest area (FAO 2015; Payn et al. 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 growth rate and wood quality (Fox et al. 2004). Management practices increase growth by maximising leaf area and growth efficiency (Waring 1982). Increased leaf area can increase water demand by trees (Scott et al. 2004). Fast-growing plantation forests may, under certain conditions, have greater water use efficiency (WUE) than unmanaged vegetation (Gyenge et al. 2008). Site condition also affects WUE in monoculture and mixed-species plantations of native tree species (Douglas-fir (*Pseudotsuga menziesii*) and red alder (*Alnus rubra*)) (Moore et al. 2011a, b).

Numerous studies, many in the form of paired catchment experiments, have shown conclusively that forest plantations of non-native fast-growing tree species, which are often densely planted, generally consume more water than 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). Plantations of eucalypts in South Africa have been convincingly demonstrated to utilise more water than the vegetation they replaced

(Dye and Versfeld 2007; Scott and Prinsloo 2008). Reforestation improved soil infiltration in the Philippines (Zhang et al. 2019a). Evapotranspiration from grasslands was lower than from plantation forests along a rainfall gradient (Zhang et al. 1999, 2001). These findings have helped support the view that forest management involves a trade-off between ‘blue’ water (to streams) and ‘green’ water (to trees) (Calder et al. 2007; Cristiano et al. 2015). This literature, however, has largely remained silent about the downwind impacts of forest cover change (Ellison 2018).

Procedures have been developed to estimate the expected impact of plantation forestry relative to a baseline of natural vegetation 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), as well as expanding rubber plantations in SE China (Guardiola-Claramonte et al. 2010). To date, no procedures have been developed to estimate downwind impacts on rainfall and water availability.

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 streamflows, tree mortality may also decrease water quality (Hibbert 1965; Swank et al. 2001).

24.2.2.3 Forestation

Forestation (used here as a generic term to reflect any increase in tree cover, regardless of methods applied or prior land use) may increase infiltration and soil moisture storage, reduce kinetic energy or precipitation and hence reduce erosion, and contribute to improvement of water quality and quantity. However, this depends on what it replaces, the species used and the approach taken. Forestation projects may involve plantation forestry or regeneration of native forests, which may have different effects on water yield. In a review of forest restoration effects on streamflow by Filoso et al. (2017), most studies reported decreases in water yields following forestation. Studies of forest restoration effects on local water yield are limited, especially for the humid tropics and subtropics.

Although forestation may reduce streamflow locally, larger-scale impacts of forestation, especially downwind from the forestation site and beyond the basin, and interactions with local climate, orography, and other features, may alter precipitation within and beyond the forested areas in ways that increase water availability (e.g., Van der Ent et al. 2010; Ellison et al. 2012, 2017; Keys et al. 2016; Creed and van Noordwijk 2018; Ellison 2018; Staal et al. 2018). Water yield responses to changes in forest cover also may occur over very long time periods (Naranjo et al. 2011).

In the northeastern United States, much of the Allegheny Mountain range was harvested in the early twentieth 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 USA (NY EPA 2015). More recently, China implemented a 'Greening China' initiative. 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). Effects of reforestation vary, depending on the choice of species, methods, and objectives (Mansourian et al. 2017).

24.2.2.4 Agroforestry

Like forestation, agroforestry – the interplanting of trees with crops – can significantly improve water infiltration, soil organic matter, and nutrient status; but, in contrast to forestation, the addition of trees to crop fields may also enhance local water availability to crops (Ong et al. 2014; Zomer et al. 2016). Tree litter enhances soil organic matter content, which in turn increases soil water holding capacity and may offset the water use of trees and crops (Mutegi et al. 2008). Tree roots compete for water with those of annual crops, but can also draw water from deeper layers of soil and, through hydraulic redistribution, replenish dry topsoil layers at night (Bayala et al. 2008). The lower density of trees in agroforestry systems, compared to forestation projects, in addition to improving infiltration, may also limit resulting reductions in groundwater (Ilstedt et al. 2016). Agroforestry systems can redistribute soil water belowground and along slopes (Wu et al. 2017).

Trees on farms can, under certain circumstances, help mitigate the effects of weather extremes on crops such as droughts, heat waves, and heavy rain, as well as provide shelter for livestock in extreme wind, high temperatures and extreme rainfall events. 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, 2019). Further, appropriate agroforestry species can provide additional household food sources, fodder and shade for animals while providing organic fertilisers for the annual crops during the rainy season (Boffa 1999).

24.2.2.5 Urban and Peri-urban Forestry

The majority of the global population now lives in cities (54.5% in 2016 and around 80% in developed nations (United Nations 2016)). The global proportion is expected

to reach 60% by 2030, with the large majority of that population growth occurring in rapidly expanding cities in Asia and Africa (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 concentrate 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) (Grimm et al. 2008; Millennium Ecosystem Assessment 2005). 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 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). Urban trees and forests provide surface cooling (Bounoua et al. 2015), increased infiltration, and attenuation of urban stormwater during precipitation events. Trees in cities mitigate stormwater runoff in at least three ways: by intercepting and storing water in their leaves and branches, by transpiring and thus reducing soil moisture (Chang and Li 2014), and by increasing litter fall and soil carbon, and thus the water holding capacity of soils, as well as increasing soil porosity through root expansion (Bartens et al. 2008). Trees also improve urban water quality by reducing sediments and particulate pollution (Sanders 1986).

Municipalities are increasingly 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 (Ow and Ghosh 2017; Scholz and Grabowiecki 2007). 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).

24.3 Anticipated Changes to Forest-Water Interactions

Multiple future changes are expected to affect forest-water interactions in a range of ways. Climate change will and appears already to directly alter forest hydrological processes. At the same time, social and economic factors will directly alter forest management, through processes such as the expansion of intensive plantations, deforestation, forest degradation, selective logging, loss of riparian forest, and loss of urban trees, all of which will affect the hydrologic cycle. In addition, climate change is likely to alter the disturbance regimes of forests, indirectly influencing forest hydrology. Moreover, forest management will respond to climate change, which in turn will affect forest hydrological processes. These future changes are described below.

24.3.1 Climate Change and Future Forest Hydrology

Climate change will likely lead to an intensification of the hydrologic cycle in areas 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; Li et al. 2017). More extreme precipitation regimes will imply a greater need for the flow-modulating effects of vegetation (Knapp et al. 2008), with flood risk increasing. Climate change is altering forest structure and species composition, and forest cover is expanding to higher latitudes and higher elevation, which may mitigate or exacerbate the direct effects of climate change (Hinzman et al. 2005; Lindner et al. 2010).

Rising atmospheric CO₂ concentrations can increase forest growth, while rising temperatures will promote forest expansion. Both processes may increase evapotranspiration (Wramneby et al. 2010), particularly in regions where there is no significant water or nutrient shortage (Holtum and Winter 2010). Rising atmospheric CO₂ concentrations can, however, also induce a partial closure of vegetation stomata and thus suppress evapotranspiration and increase runoff (Gedney et al. 2006). Rising atmospheric CO₂ concentrations have mixed and interacting effects on evapotranspiration, tree growth, and runoff (Hickler et al. 2008; Norby et al. 2010; Norby and Zak 2011; Silva and Anand 2013).

Forests, especially native and old-growth forests, influence future hydrology indirectly by storing and sequestering carbon and potentially limiting increases in air temperature and evapotranspiration. 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). At a timescale of decades, the use of forest products to substitute for fossil fuels or concrete in construction can lead to managed forests becoming net carbon sinks despite recurring harvests (Lundmark et al. 2018).

On a global scale, evapotranspiration increased between 1982 and 2008 (Jung et al. 2010), although changes in evapotranspiration are variable among regions. Evapotranspiration declined in parts of the world where soil moisture limits vegetation water use – including parts of Australia, East Africa, and South America, but evapotranspiration increased in regions such as China and southern India (Jung et al. 2010). Factors including soil moisture, stomatal closure resulting from rising CO₂ 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).

Forest die-off from drought and heat stress has, as mentioned previously, 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 CO₂ 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 USA, non-catastrophic mortality rates increased rapidly in recent decades, targeting small trees (van Mantgem et al. 2009). Forest mortality to drought may also increase in areas that currently or previously received high rates of atmospheric nitrogen deposition due to changes in tree morphology (McNulty et al. 2017).

Analyses of long-term records at 35 small watersheds (0.01–1 km²) in the USA and Canada indicate that climate change effects on streamflow are not as clear as might be expected, arguably because of ecosystem processes and human influences (Jones et al. 2012). Although air temperature increased at 17 out of 19 sites with 20–60 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 USA 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.

Studies in small experimental catchments indicate that forest dynamics, including legacies of past disturbance and forest management, as well as forest succession and tree diversity, produce a wide range of forest hydrologic response to climate change at individual sites. For example, in the northern hardwood forest of Hubbard Brook (USA), 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 USA (Coweeta, North Carolina), 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, under certain circumstances, make permafrost resilient to large increases in air temperatures. However, as seen in the 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 frequency), 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, streamflows will become less responsive to precipitation and headwater streams may become ephemeral (Jones and Rinehart 2010).

Beyond the scale of individual catchments, it is, as yet, unclear how climate change will affect atmospheric moisture flows across short and long distances, and at seasonal and interannual time scales, and how these changes will interact with future forests to affect the global hydrologic cycle.

24.3.2 Forest Cover, Forest Management, and Future Forest Hydrology

Clearly, as shown thus far, anticipated future changes in forest cover and forest management are diverse. They may include expansion of intensive plantations,

deforestation, selective logging, loss of riparian forest, and loss of urban trees, all of which will affect the hydrologic cycle at multiple temporal and spatial scales. Deforestation has been high, especially in the tropics, since records of global forest cover began (Allen and Barnes 1985). From 2000 to 2012, globally 2.3 million km² of forest were lost, and 0.8 million km² 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; Jones et al. 2017). Boreal forest loss, due largely to fire and forestry, was second in absolute and proportional terms.

As shown earlier, deforestation tends to increase streamflow, and forestation decreases streamflow (Li et al. 2017) even though there are exceptions and cases where forest cover change is not reflected in the hydrological regime, as in a study of four watersheds from the Ethiopian Highlands (Gebrehiwot et al. 2013). But changes in forest cover also extend to other parts of the hydrologic cycle, including evapotranspiration and precipitation recycling. Tropical deforestation results in warmer, drier local conditions (Lawrence and Vandecar 2015; Lovejoy and Nobre 2018). 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 (1–15 km) but was positively correlated with measured precipitation at the regional scale (30–50 km) (Debortoli et al. 2017). Thus, increases in forest cover may also contribute to higher precipitation locally or at broader scales (Ellison et al. 2012, 2017).

24.3.3 Climate Change Effects on Forest Disturbance, Succession, and Future Forest Hydrology

Anticipated future changes in forest disturbance regimes include, as shown earlier, 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).

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 – some of the complexities of processes involved are shown earlier in this chapter. 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). 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). Although limited research has, thus far, been conducted on this topic, there is some evidence that species distribution models can usefully predict trends in average ecological conditions, but not species composition, of future forests (Scherrer et al. 2017).

24.3.4 Climate Change and Future Forest Management

Climate change implies an increased need for hydrological (and other) resilience in our landscapes (e.g., limiting floods and withstanding drought) (e.g., Hatcher and Jones 2013). Forest change may reduce floods and resist droughts, but natural disturbance and human disturbance, including ongoing deforestation, as well as ongoing climate change, continually alter forest hydrology. Interannual climate variation has a much greater impact on flow than forest management at the scale in which it normally practiced (Duan et al. 2016). Under such circumstances, forest management for the future should focus on managing in the face of uncertainty while managing for multiple benefits (Millar et al. 2007). Forest hydrology will continue to respond to multiple system drivers of change. Decisions should thus not be based on expected responses to single factors and be informed by what works under which conditions (Lindner et al. 2014). Forests and forest hydrology have experienced varied responses to change over the past 50–100 years; this underscores the importance of incorporating stochastic variability into projections of future forest conditions and forest hydrology (Daniel et al. 2017). Nevertheless, the demonstrated hydrologic resilience and adaptive capacities of native, unmanaged forests argues for the preservation and improved management of native forests. At the same time, in the face of future uncertainty, the most robust managed forests should have both structural and species diversity, for example, by retaining a large proportion of mature trees while using thinning to create spatial heterogeneity and enhance structural and compositional complexity (D’Amato et al. 2011).

24.4 Data Needs and Knowledge Gaps

Despite ongoing global environmental change, knowledge of forest responses to natural and anthropogenic disturbances and their effects on forest hydrology (as described above) is robust (and growing) and highly useful for predicting and managing the effects of current and future forest change. Ongoing and anticipated forest change will involve many of the same disturbance agents, including insects and pathogens, wildfire, wind, and ice, as well as forest harvest and tree planting, whose effects on forest hydrology have been thoroughly studied.

Nevertheless, the nature of forest changes and their effects on hydrology are poorly documented in many areas of the world, and novel combinations of processes and contexts may produce surprising outcomes. There is thus a clear need for more geographically extensive and long-term place-based studies of forest and water. Finally, new techniques and concepts, such as the tracing of evapotranspired water from forests and its recycling as precipitation, deserve greater attention.

Research topics include (but are not limited to) increased basic data on runoff and precipitation in currently ungauged watersheds, studies of the fate and transport of atmospheric water, and its impact on downwind rainfall and water availability, along with improved models and data products. Action is also needed to make forest hydrology data publicly accessible.

As mentioned above, there is a general lack of hydrological and meteorological monitoring, particularly in developing countries (McNulty et al. 2016). Paired watershed experimental studies became key to the development of forest hydrology as a science, a century ago. But these studies are resource intensive and limited in occurrence and scope. Data from paired watershed experiments have been mostly obtained in temperate moist climate zones (e.g., Hibbert 1967; Bosch and Hewlett 1982; Andreassian 2004; Jackson et al. 2005).

Increasingly, studies are based on remotely sensed data and process-based models. But these studies are weakened by a lack of reliable ground-truthed data (Meijninger and Jarmain 2014). However, intermodel comparisons and validation can be informative when databases are available for such studies (Caldwell et al. 2015). 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).

Improved sharing of data relevant to forests and water would greatly enhance research and related policy discussions (Emanuel et al. 2015). 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.

Uncertainty associated with the transport and fate of water evapotranspired from forests continues to pose a major challenge for science, land management and policy (van Noordwijk et al. 2014). The fraction of local precipitation that is derived from terrestrial sources varies from 0 to nearly 100%, depending on location and distance from upwind oceanic sources. Paired watershed studies, which are limited to usually small catchments, do not account for the water that is evapotranspired and leaves the confines of the basin; thus, these studies are unsuited to testing the hypothesis about the impact of forests on rainfall. Water isotopes are providing insights into the origins of atmospheric water, as well as water residence times in soil, vegetation, rivers, and the atmosphere (Wright et al. 2017; McDonnell et al. 2018). More work is needed to develop and test models of spatial transfers of atmospheric moisture

(e.g., Weng et al. 2018). Although, as shown earlier, studies on precipitation recycling have been conducted in the Amazon (e.g., Bagley et al. 2014; Nobre et al. 2014; Spera et al. 2016; Wright et al. 2017), in Africa (Arnault et al. 2016; Dyer et al. 2017; Gebrehiwot et al. 2019), and in Asia (Hua et al. 2016; Kong and Pang 2016), additional studies are needed to clarify how forests, wetlands, and irrigated agriculture influence the fate and transport of evapotranspired water. Mechanistic studies also are needed to link vegetation, evapotranspiration and emission of biological compounds, and precipitation processes.

24.5 Conclusions

Changes in forest (due to both natural disturbance and human activities) affect how incoming precipitation is partitioned between evapotranspiration and streamflow. A recent global assessment found that forest changes explained, on average, 30% of annual streamflow variations (Wei et al. 2018). At a daily, or event, timescale forests with adequate infiltration maintain a high ‘flow persistence’ that reduces flood risks (van Noordwijk et al. 2017). This study and a range of other findings in forest hydrology reviewed in this chapter show that the relationship of forests and water is critical. It is, however, more difficult to generalise how forests and forest change will affect water in the future. The effects of forests on the hydrologic cycle occur at multiple spatial and temporal scales, and forest succession and disturbance alter the relationships of forests to water over time. Moreover, because water cannot be created or destroyed, local changes of forest-water relationships may be balanced by regional or global changes (Ellison 2018). As a result, the knowledge base to make specific predictions of the consequences of forest change at all relevant spatial and temporal scales is still evolving.

Natural disturbance such as wildfire, insect pests, diseases, and windthrow, among others, can, under certain circumstances, significantly alter a range of watershed processes and ecological functions in forested watersheds. Natural disturbance is part of the dynamics of forest ecosystems, but extreme events (e.g., stand-replacing wildfire or large-scale insect pest outbreaks) may, under certain circumstances, cause undesired ecological and economic consequences.

Human-driven changes to forests include deforestation, forest degradation, forestation, agroforestry, plantation forestry, 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 globally and needs to be well understood to best provide an evidence base for informed management.

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 factor in uncertainty as well as assume that uncertainty is likely to increase. Decisions should be based on expected responses to multiple factors, not single factors, as well as supporting resilience and multiple benefits. Evidence of how forests and forest hydrology have responded to change over the past 50–100 years highlights the need to incorporate stochastic variability into projections of future ecosystem condition and to utilise the precautionary principle in forest management.

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