3 Managing forests for water

Key points

- A growing human population and a changing climate have put pressure on many ecosystem services, increasing the need to manage forests for water. The demand for water is expected to continue increasing through the twenty-first century.
- Sustainable forest management for other ecosystem goods and services, including timber, is compatible with water-quality objectives, although trade-offs may be required. There may also be synergies; for example, water quality is closely linked to soil conservation, a priority of sustainable forest management for timber production.
- Increasing the resilience of forests to environmental stress will help reduce the risk of a catastrophic decline in forest ecosystem services, including those related to water.
- Many ecosystem management tools are available to assist in managing forests to benefit water quantity, quality and timing. Conversely, poor forest management can have long-term negative impacts on forest health and water resources.

Forests are often managed for a wide range of purposes, such as wood production, recreation and biodiversity conservation. Healthy, well-managed forests also store and filter water as well as reduce surface runoff and flood risk. Regrowing forests, on the other hand, can reduce downstream water supplies. Forests that are unmanaged may become overstocked (i.e. have a very high density of trees per unit area). This, in turn, can increase susceptibility to insect outbreaks and the risk of wildfire from the accumulation of fuels (Shang *et al.*, 2004), both of which can have significant impacts on the forest hydrologic cycle (Goeking and Tarboton, 2020). Additionally, some unmanaged and potentially overstocked forests use more water and therefore may produce less streamflow than managed forests (i.e. with less growing stock). Forest managers need to achieve a balance between optimizing water yield (Evaristo and McDonnell, 2019) and keeping sufficient canopy to minimize soil erosion, maintain albedo (i.e. the proportion of incident light or radiation reflected from a surface) and promote water quality. Competing trade-offs between water and non-water natural resource demands from forests is a major forest management challenge (Sun and Vose, 2016). The need for clean, abundant, consistent water supplies is likely to increase as the climate changes and the human population continues to increase (Sun and Vose, 2016). Currently, about 4 billion people are affected by water scarcity at least once in any given year (Mekonnen and Hoekstra, 2016); this number is projected to grow to 6 billion by 2050 (Boretti and Rosa, 2019). Therefore, forest management that is explicitly designed to increase high-quality water supply is needed urgently.

Forests already provide much of the water used by humans, but this contribution must increase to ensure adequate water security. Even if a forest is not managed primarily for water, a better understanding of the principles associated with water management will help enable a forest's efficient contributions to co-benefits, including water. Compared with other land uses (e.g. agriculture and livestock grazing), forests generally produce less surface and subsurface water runoff due to their relatively high rates of transpiration. This chapter addresses forest management approaches that optimize the quantity, quality and timing of water resources.

Principles of forest–water relationships

Forests regulate the flow of water through evapotranspiration, soil water storage and storm runoff (Andréassian, 2004; Smith *et al*., 2011). The removal of plants (trees, shrubs, forbs and other vegetation), and changes in land use to low or seasonal vegetation cover, can have major impacts on the regulation of water quantity, quality and timing. Impacts on forest soils can affect forest water (Smith *et al*., 2011); therefore, undisturbed forests often have the highest water quality (Fredriksen, 1971).

An understanding of the principles of forest–water management is crucial for ensuring best-practice uses of a water resource (McNulty *et al*., 2010). Forest water comprises two general components: 1) water stored in the soil used by forest flora and fauna; and 2) water that either recharges the groundwater or is exported from the forest as streamflow. Changes in groundwater supply and streamflow determine water quality and quantity (Ellison, Futter and Bishop, 2012).

The forest overstorey is the primary source of leaf litter, which, when it falls and decomposes, contributes to healthy forest soils and helps ensure good water infiltration and filtration. This, in turn, is important in the water cycle and the supply of drinking water (Hongve, Van Hees and Lundström, 2000; Boggs, Sun and McNulty, 2015). Streams and springs in forests continue to flow with relatively high-quality water long – perhaps months – after the last precipitation event due to the slow rate of water infiltration through the profiles of healthy forest soils (Che, Li and Zhang, 2013). Forest-overstorey tree roots also help in mitigating mass wasting (i.e. landslides) and soil erosion by holding soils on hill slopes (Marden and Rowan, 2015). Water emerging from some forested watersheds (e.g. those serving Vienna in Austria and New York and Seattle in the United States of America) is of sufficient quality that only minimal secondary treatment is required before human consumption. Below, we consider the effects of forests on water in relation to three water-resource properties (referred to generally as water services or values): quantity, quality and timing.

Water quantity. Growing forests can have a direct impact on water availability. Planted forests use more water than natural forests due to a "plantation effect" (Kuczera, 1987) in which trees planted at the same time and growing at the same high rate result in high water demand; consequently, they have relatively greater potential than natural forests to reduce water availability in periods of high growth.

Water yield (i.e. quantity) from a forest is determined by the amount of precipitation minus evapotranspiration and water stored in the soil. Forest managers cannot control precipitation but they can influence evapotranspiration through management practices. All trees use water for photosynthesis, and they also lose water during leaf respiration. Therefore, most forests lose soil water through their canopies, although, in some circumstances, forest canopies can increase soil water by intercepting water directly through fog drip from the leaves to the soil (see the discussion on tropical montane cloud forests in Chapter 5). Thus, the density of leaves (known as leaf area) of a canopy has an important impact on the amount of water lost through tree transpiration.

A second variable affecting forest water use is the efficiency with which trees and vegetation use water to grow and sustain themselves – known as water-use efficiency (WUE). A tree species with a low WUE uses more water to produce the same volume of growth compared with a tree species with a higher WUE.

The third consideration for understanding forest water use is how quickly trees grow. Faster growth involves a higher absolute use of water per unit of time (Forrester, 2015; White *et al.*, 2014). Anthropogenic climate changes in air temperature and precipitation are also likely to have significant impacts on the quantity of freshwater supplies (Box 3.1).

BOX 3.1

Global changes in freshwater river discharge as output to marine systems

Various anthropogenic pressures, including climate change, affect river discharge patterns, physical properties and biogeochemical cycling at local scales (Grill *et al.*, 2019). Most approaches for understanding and assessing climate risk to river discharges rely on the statistical analysis of historical discharge time series or on large, physically based runoff models coupled with general circulation models.

Analyses of historical data across large, ocean-reaching rivers indicate both increases and decreases in streamflows, with a larger number of decreases (Gerten *et al.*, 2008; Dai *et al.*, 2009; Su *et al.*, 2018). For all oceans except the Arctic Ocean, the quantity of river discharge is trending downward. An increasing discharge trend is evident in high-latitude areas and a decreasing streamflow trend is prevalent in low-latitude areas. This pattern can be attributed to uneven precipitation and the effects of global warming (Su *et al.*, 2018). Large-scale ocean circulation patterns such as the El-Niño Southern Oscillation, the Arctic Oscillation and the Pacific Decadal Oscillation may also cause shifts in river discharge through their influence on precipitation (Su *et al.*, 2018).

Coupled climate–hydrology models have the benefit of simulating hydrologic processes under multiple climate scenarios and explicitly forecasting future hydrographs. Modelling suggests that, by the end of the twenty-first century, annual mean precipitation, evaporation and runoff will have increased in high latitudes in the Northern Hemisphere, in southern to eastern Asia, and in Central Africa, and they will have decreased in the Mediterranean region, southern Africa, southern North America, Central America and Australia (Nohara *et al.*, 2006; van Vliet *et al.*, 2013). The seasonality of river discharge is expected to increase, and high-latitude rivers are expected to experience shifts in flow timing because of earlier snowmelt (Nohara *et al.*, 2006; van Vliet *et al.*, 2013).

Water quality. The quality of water coming from forests is almost always higher than from other land uses (e.g. agriculture) that expose the soil, but water quality varies over time and space. Water generally has more oxygen and lower levels of suspended sediment in headwater forests compared with downstream forests. Similarly, water quality can be lower after a large precipitation event compared with the same stream during base flow (i.e. non-precipitation) periods due to increased turbidity and chemical contamination from overland flows. Forest management can strongly influence water quality: for example, operations such as harvesting, soil preparation and fertilizing can increase the quantity of suspended sediments and nutrients in water bodies, and certain activities (such as fertilizing and the use of pesticides) can contribute chemicals to water bodies (Neary, Ice and Jackson, 2009).

Water timing. Forest growth and management affect the division of rainwater into runoff and infiltration. Rapid forest growth can reduce water availability; conversely, the clearfelling of trees can cause dramatic increases. Changes in tree cover can affect the amount of precipitation stored as snow and – by influencing soil health – the amount of water stored in soils (Box 3.2). These types of impact can alter the seasonal timing of flows. Monitoring is essential for ensuring that management practices are not causing negative impacts on water timing (Harris *et al.*, 2007).

Trees in forests are a source of organic material for building new soils. Forest litter (e.g. leaves, branches and boles) decomposes in contact with the soil (Krishna and Mohan, 2017). If the rate of litter input is faster than the rate of decomposition, an organic horizon is formed on top of the mineral soil (Van Cleve and Powers, 2006). In addition to surface organic matter, the decomposition of roots and other biotic components can incorporate organic matter into the soil profile. This process is

BOX 3.2
Soil: a key to forest–water relationships

Healthy soils are essential for the timing of downstream water supply; the protection of water quality due to soil water filtration; and minimizing the loss of water quality due to erosion. Forests contribute to healthy soils by protecting against both episodic and chronic soil degradation. Tree roots anchor the soil mass and increase its macroporosity, increasing infiltration. The risk of mass wasting and debris flows decreases as water infiltration increases. Tree roots in riparian areas and along river channels slow in-channel and flood flows, thereby protecting against bank erosion and debris flows while allowing coarse sediments to settle and filtration processes to occur. The rate of movement and the energy associated with water moving across the land during heavy precipitation increase with increasing steepness of slope (Miyata *et al.*, 2009). In intense rain events, the rapid movement of water can cause rill erosion, reducing the amount of water that can be stored in the soil and ultimately leading to gully erosion. Forest canopies also provide a barrier against the physical process of water striking and dislodging soil. A closed (also called full) canopy protects the soil surface from the direct impacts of rain droplets and thereby reduces soil erosion. As soil erosion is reduced, soil organic matter is conserved and soil water infiltration improves. Conversely, exposed soil is at greater risk of erosion, with a consequent loss of water quality (Jiang *et al.*, 2019).

essential for water infiltration and therefore for reducing rapid overland waterflows during precipitation events (Krishna and Mohan, 2017) and maintaining natural seasonal timing and moderated flows, even after intense rainstorms. Conversely, streamflows from a forest with very eroded soils are highly variable, with episodes of large runoff followed by periods of limited streamflow (Yoho, 1980). Some ecosystems, such as peatland forests (see Chapter 5), have organic horizons many metres deep. Because this organic matter is highly porous, peatland forests can hold large volumes of water (Miller, 1983).

MANAGING FORESTS PRIMARILY FOR WATER

In Europe and elsewhere, early forms of forest management were likely byproducts of preferential species selection for growing fuel, building materials and fodder (Dufraisse, 2008). Preferred tree species and timber size classes were favoured to meet community needs, shifting the forest tree composition and dominant species. Forest management expanded in the medieval period to include wildlife (Jørgensen, 2004), such as in English Royal Forests, where rulers could grant access for the hunting of game species (e.g. rabbit, fox and deer). There are similar examples of early forest management in many other cultures. Although, in some cultures, water was not traditionally considered a primary forest management objective, others – such as the Chinese, Mayan and Native American – recognized the close relationship between forests and clean, sustainable supplies of water and managed their forests accordingly (Neary, Ice and Jackson, 2009).

Forest managers engage in a range of practices to manipulate forests to achieve desired results, such as increasing forest growth, conserving biodiversity, sequestering carbon and reducing the risk of pest outbreaks or wildfire. Many of these silvicultural practices also affect water services (Figure 3.1) and, if well designed and implemented, can contribute to water management goals. (Forest management in which optimizing water quality and quantity is the primary management objective can provide co-benefits in much the same way.)

The impacts of wood harvesting operations on water quality can be reduced by adhering to the following three principles:

1. **Minimize soil compaction, which reduces water infiltration.** This may require reducing or eliminating the use of heavy equipment; limiting forest operations to periods when the soil is less prone to compaction (e.g. when dry or frozen); avoiding harvesting when unacceptable levels of compaction are likely; and developing road networks to balance the deleterious effects of roads on soils with those of the off-road activities of heavy machines.
2. **Minimize soil erosion due to surface waterflows.** This may require avoiding tree felling and extraction on steep slopes; reducing the size of the contiguous harvest area, especially on steeper slopes; and, perhaps most importantly, constructing and maintaining roads deploying best management practices (e.g. the use of broad-based dips, construction away from stream crossings, and the use of bridge mats, culverts and switchbacks).
3. **Maintain appropriate undisturbed buffers between harvesting areas and surface water.** It is essential to maintain relatively undisturbed buffer zones of trees, shrubs and other natural groundcover along streams and around lakes, ponds and other water bodies, in part to ensure continuous tree litter cover on soil surfaces next to water (which can reduce soil sedimentation and minimize unusual water temperature fluctuations) and to reduce erosion. Selective harvesting may be permitted in buffer zones under certain conditions (e.g. if it can be done without heavy machinery entering the buffer zone). For small streams, an adequate buffer-zone width might be in the range of 10 m to 30 m – a rule of thumb is that the width of the buffer zone on each side should be at least equal to the width of the stream. The special needs of riparian forests are addressed later in this chapter.

Many silvicultural practices can help maintain or improve water values, although their application may vary due to factors such as forest type, other forest management objectives, forest condition, resources available for management, time of year, and desired future condition (Sun *et al*., 2008; Filoso *et al*., 2017). The impacts of some commonly used management practices – the construction and maintenance of road infrastructure; harvesting; and regeneration – on forest water resources are examined below, along with key means for minimizing such impacts (FAO, 2008; Boggs, Sun and McNulty, 2015; Boggs *et al*., 2015).

Log extraction with a track skidder in a logging concession in Indonesia

Road infrastructure. Roads are essential for effective forest–water management because they enable access for the application of management practices, but they can also have negative impacts on water services. For example, in the Pacific Northwest of the United States of America, road and ditch networks reduce soil erosion by rerouting precipitation away from streams during heavy rainfall events (Harr *et al.*, 1975; Jones and Grant, 1996). Water that flows along roads and ditches can pick up contaminants that then enter streams, rivers and reservoirs without the benefit of soil filtration. Water flowing along roads and ditches often moves at high velocity, enabling the transport of large particles of sediment and increasing the risk of erosion, debris flows and mass wasting where high-velocity water flows over soils or along stream channels. Because road networks alter peak flows and have the potential to reduce water quality, it is important to keep roads hydrologically disconnected from stream networks via culverts and other forms of engineering (Harr *et al*., 1975).

A logging road with an improperly constructed stream crossing

Roads can have large spatial and long temporal impacts on forest quality and quantity (Figure 3.1). The following practices can help minimize negative impacts:

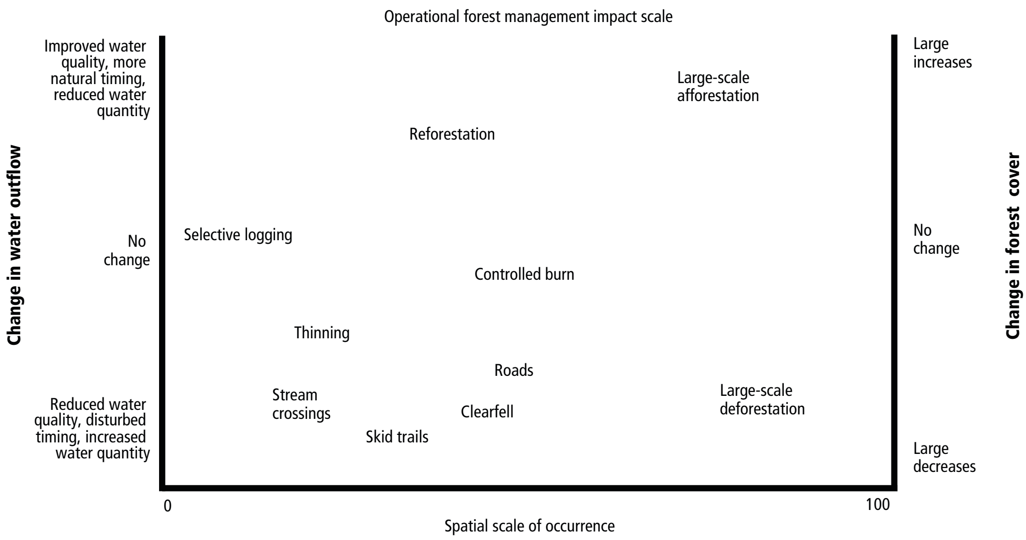
- identifying, describing and mapping all streams, wetlands and other water features, as well as slopes and soil types, and taking these fully into account in road planning;
- pre-logging planning of roads, skid trails and landings to provide access to the forest and the trees to be harvested while minimizing soil disturbance and protecting streams;
- setting a maximum skid-trail density (e.g. 20 km per 100-ha block) to guide planning;
- ensuring the proper construction and maintenance of roads in accordance with sound environmental and engineering standards;
- locating roads on stable soils at an adequate distance from streams and avoiding landslide-prone areas;

- building properly designed stream crossings such as bridges and culverts;
- in highly vulnerable soils, evening out soil pressure from heavy forest vehicles by preparing skid trails with branches to reduce soil damage;
- equipping roads that enter stream buffer zones with roadside ditches and properly spaced cross-drains, with drain outflows diverted to surrounding vegetation at least 50 m before stream crossings, and with sedimentation traps placed in drains and ditches;
- constructing roads in the dry season; and
- properly maintaining all roads in the network.

Some climate-vulnerability assessments and adaptation plans have suggested closing and even removing roads because of concerns over their impacts on downstream ecosystem services (Halofsky *et al*., 2011). Restoration by road decommissioning can be valuable in watersheds where limiting human impact is a goal, but, for many forests, maintaining access is important for supporting the provision of ecosystem goods and services. For example, in the United States of America, Native American tribes have emphasized the need for road access into forests to enable traditional practices (Long and Lake, 2018). In Southeast Asia, some climate-vulnerability assessment and adaptation plans recognize the importance of roads for ensuring adaptive capacity (Yusuf and Francisco, 2009) because they enable local communities to get their goods to market and receive services during emergencies. Roads may also be essential for forest management in fire-dependent ecosystems by facilitating access for fuel-reduction treatments, managed wildland fires and the suppression of wildfires (Spies *et al*., 2018).

Harvesting. Forest thinning and harvesting – logging – are part of productive forest management to obtain timber and woodfuel from forests. Tree harvesting temporarily reduces the leaf area of a forest, reducing forest evapotranspiration (Yan *et al*., 2012) and potentially increasing forest water yield (Goeking and Tarboton, 2020). Other factors associated with logging, such as an increase in albedo and a reduction in cloud-water capture, can also affect stream yield (Goeking and Tarboton, 2020). Bare soils tend to have higher albedo (i.e. reflect more incident light) than forests with intact canopy cover. As albedo decreases, a forest will (by definition) absorb more energy and thus use more water, leading to a decrease in forest water yield if all other factors remain constant. On the other hand, a reduction in forest canopy may lead to increased erosion and decreased soil organic matter (and thus loss of water quality),

FIGURE 3.1
Natural and human-originated disturbances can affect water quality and quantity at different spatial scales due to changes in forest cover



partly because more rain will directly strike the soil surface; a balance must be found between wood harvest volume, increased water yield, water quality, and the timing of waterflows.

Soil erosion is generally the most serious risk to water quality associated with forest harvesting. This is often caused by harvesting on steep slopes and by poorly planned or constructed logging roads and skid trails. Practices to minimize soil disturbance due to ground-based logging include winching logs to reduce soil disturbance associated with skidding; making use of yarding systems that protect soils by suspending logs above the ground (e.g. the use of "logfisher" and cable systems, and helicopter logging); avoiding ground-based harvesting on steep slopes above a certain threshold (e.g. 15°–40°) and avoiding all harvesting on the steepest slopes; designing skid-trail networks and landings to maximize uphill skidding; and minimizing wet-weather skidding.

Other operational considerations to minimize soil disturbances during harvesting operations are discussed below.

Clearfelling. Clearfell timber harvesting (also called clearcutting)[8] increases the risk of mass wasting and soil erosion and can harm forest soil functionality through compaction by heavy machinery (Poff, 1996). Although clearfelling can increase short-term water yields, its impacts on forest soils can cause long-term declines in water quality (Borrelli *et al.*, 2017). Rebuilding a stable and functional soil layer after clearfelling can take decades.

Selective logging. Well implemented, selective harvesting[9] involves less vegetation removal and soil disturbance than clearfelling, resulting in less surface runoff and lower peak water discharges and erosion. Undesirable impacts of selective logging on soils and water can be reduced substantially through the adoption of low-impact measures by appropriately trained, supervised and compensated logging crews (Putz *et al.*, 2008). Importantly, however, there is often a lack of incentives for forest managers and operators to implement measures to protect or restore forest–water values beyond what is legally required (unless, for example, the forest is certified as well-managed and this is an important part of marketing efforts). In many developing countries, ensuring compliance with the provisions of harvesting permits is difficult due to, for example, the remoteness of the forests, inadequate resources and capacity for monitoring, and weak governance. Protecting and maintaining forest ecosystem services, including water services, will likely have lower priority in the absence of incentives to cover the costs of implementing additional measures. Recognizing the value of forest ecosystem services, and incentivizing forest–water management, are further discussed in Chapter 4.

Forest thinning. Variable-density thinning (VDT) is a silvicultural tool for managing uneven-aged native forests, the goal of which is to increase environmental variability (e.g. forest structure and function) across a landscape while maintaining the resilience of native tree species and reducing negative impacts on forest water yield (Sun, Caldwell and McNulty, 2015). Typically, VDT targets relatively young homogeneous forest stands and removes smaller individuals of the most abundant tree species ("thinning from below"), thereby maintaining larger trees and improving the relative species diversity while decreasing competition for light and water resources and increasing growing space for the residual trees.

Controlled burning. Controlled burning can be used as a silvicultural tool to reduce ground vegetation and influence the distribution of tree species (Ditomaso *et*

[8] Clearfelling is a harvesting system in which all merchantable trees within a specified physical area of land are felled and no significant tree cover remains (Dykstra and Heinrich, 1996).

[9] A selection harvesting system ("selective harvesting") is a logging system in which crop trees are removed on a cycle of felling entries that occur more frequently than the rotation. In such systems, not all crop trees are removed during a particular felling entry; selection of those to be harvested and those to be retained may be based on diameter at breast height (dbh) (e.g. only those crop trees larger than 60 cm dbh are to be removed) or other criteria (Dykstra and Heinrich, 1996).

al., 2006). In the short term, killing certain plants on the forest floor through low-intensity fire reduces leaf area and evapotranspiration and increases forest-floor albedo, with resultant changes in hydrology (Hallema *et al*., 2018). Caution must be used in applying controlled burns to ensure that fires remain low-intensity and manageable. The escalation of a controlled burn into a wildfire can cause major reductions in water quality; in extreme situations, the recovery of water quality may take many years (Hallema *et al*., 2018).

Species selection. Not all tree species use the same volume of water per unit leaf area. Some, such as many *Eucalyptus* species, have high water demands to support rapid growth, and other species are more water-conserving (Aranda *et al*., 2012). Moreover, some species are better adapted to drought conditions than others (Eilmann and Rigling, 2012). Such factors should be considered when planting or harvesting trees for fuel or timber. A fast-growing species such as loblolly pine (*Pinus taeda*) can produce a large volume of timber quickly but at the cost of high water use and consequent reduced stream water yield (Sun and Vose, 2016). If water management is the primary objective of a forest, a balance may be needed between slower-growing, minimal-water-using trees and faster-growing, higher-water-using trees. Locally adapted native tree species are often best suited to reforestation for improved water management because of their high WUE in local conditions and resilience to local environmental pressures, although this might be at the expense of lower biomass production than might be achieved using fast-growing non-native species.

Mixed-species forest regeneration using several tree species with differing rooting morphological characteristics is an option with several advantages. Some species (e.g. *Pinus* spp.) have single taproots that can penetrate deep into soil profiles to find water that might be unreachable by species with shallower, more widespread roots (e.g. *Quercus* spp.; Vose *et al*., 2016). Additionally, mixed stands may be more efficient at maximizing the capture of solar radiation, are likely to support a greater diversity of plant and animal species, and are at lower risk of severe pest outbreaks. Thus, mixed-species forests generally provide a wider range of ecosystem services than monoculture plantations.

Tree-planting can be an important silvicultural tool for increasing the diversity of native species of trees and desired understorey shrubs (such as those that produce berries or provide habitat for key animal species) in uneven-aged native forests (Richards *et al*., 2012). Planting can be targeted at stands recently subject to VDT and areas recovering from disturbances such as fires or storms. Prescriptions should vary according to local circumstances (and variability can improve resilience) (Reynolds *et al*., 2013).

Managing for drinking-water supply

An estimated 80 percent of the freshwater resources in the United States of America originates in forests, with much of the nation's drinking water flowing from the 78 million ha National Forest system (Levin *et al*., 2002). Nationwide, 3 400 towns and cities depend on National Forest catchments for their public water supplies, and an additional 3 000 administrative sites such as campgrounds, picnic areas and historical sites rely on the same or similar sources (Ryan and Glasser, 2000). Approximately 70 percent of the forest area in the United States of America is outside the National Forest system, and more than 50 percent of the population relies on forest lands to produce adequate supplies of good-quality water (USDA Forest Service, 2014).

The percentage of cities using water from forested catchments is even higher in Canada – which has a vast forest area – than in the United States of America (Bakker, 2007). The City of Toronto draws water from Lake Ontario, one of the Great Lakes, the watersheds of which are mostly forested. Montreal's water supply comes from two lakes and two rivers; the land use around these lakes and the lower reaches of the rivers is a mix of agriculture and urban settlement, but the headwaters are forested. Vancouver's high-quality water supply comes from three forested catchments north of the city.

Germany has established water conservation districts (*Wasserschutzgebieten*) for the protection of municipal water supplies (Napier, 2000), most of which are forested. Land use is tightly regulated, and there are three levels of water protection, from wellhead (level 1) to entire catchments (level 3).

The large land mass of Australia holds less than 1 percent of the world's freshwater resources (Pigram, 2006). The major cities of Brisbane, Canberra, Hobart, Melbourne, Perth and Sydney all rely on water flowing from mostly forested catchments. A sustainable supply of good-quality water is a prime constraint on the country's economic and population growth.

Municipal water supply is often obtained from forested watersheds because trees tend to grow in relatively wet landscapes and help ensure reliable and clean water, either as runoff on the surface of the land via streams and rivers or as belowground percolation through karst geology to underground storage (Richards et *al.*, 2012). In most places, precipitation – the primary source of water in a watershed – is highly seasonal (Robinson *et al.*, 2013) and requires some form of water-storage facility to ensure reliable year-round water supplies. Rivers can be dammed to create water-storage reservoirs, enable hydroelectricity-generating resources and reduce downstream flooding, but this can have negative impacts on native fish populations, downstream flows and the overall ecology of river systems. Precipitation that falls as snow can serve an important water-storage function (Forman, Reichle and Rodell, 2012) because it can take months – often well into summer – for winter-accumulated snow to fully melt and flow downhill into streams and springs.

The quality of the water produced by a watershed is generally a function of the land use in that watershed (Fiquepron, Garcia and Stenger, 2013). A forested watershed can produce a clean and plentiful water supply; a cleared or otherwise "developed" watershed, on the other hand, might produce water that requires treatment to make it safe to drink. Industrial-scale water-treatment facilities are usually expensive, motivating water managers to reduce forest removals and improve forest and land management within drinking-water-source watersheds (Calder, 2007), which might require the consolidation of municipal land ownership and limiting public access.

Forest management in watersheds that are sources of municipal water supplies should focus on maintaining a continuous cover of natural forest as part of a healthy water cycle (Richards *et al.*, 2012). Natural forests are adapted to local environmental conditions and provide the primary structure and function of the terrestrial phase of the water cycle. Water plays an important role in the net primary productivity of forests (e.g. the accumulation of biomass), and leaf fall (litter) from trees helps build and maintain healthy forest soils that hold, filter and percolate precipitation through gravity-generated drainage and subsurface flows. Natural forests also contain ecosystem-adapted biodiversity that is most likely to be resilient to natural disturbances (Thompson *et al.*, 2009; Welch, 2008).

Continuous-cover forestry should ensure the minimization of negative management impacts on water quality; it may include the conservation of primary (i.e. old-growth) forest, the adoption of no-harvest buffers along streams and rivers, and the restoration of degraded areas. Typically, the largest threat to water quality in fully forested watersheds is erosion and mass wasting from forest roads in steep terrain (Neary, Ice and Jackson, 2009). Minimizing the development of roading, and decommissioning high-risk or unnecessary roads, can reduce this threat (see page 36).

Even-aged forest management and clearfell timber harvests have negative impacts on the quantity, quality and timing of stream flows (Segura *et al.*, 2020). Clearfelling is also incompatible with maintaining continuous forest cover and increases the risk of mass wasting events by removing the primary physical structure (i.e. tree roots) holding soils on hillsides (Barik *et al*., 2017). Uneven-aged management, VDT and selective harvesting can be deployed to restore degraded forest areas (Puettmann *et al*., 2016) while generating revenue through timber sales. VDT in dense stands that exhibit stagnating growth and density-dependent mortality typical of the stem-exclusion stage of forest succession[10] is accomplished by thinning from below (i.e. cutting smaller trees), harvesting only the most abundant species to improve the relative abundance of rarer tree species, and maintaining the largest individual trees. VDT will have limited ecological benefits in older stands that are naturally emerging from the stem-exclusion stage (i.e. exhibiting canopy differentiation among dominant and co-dominant trees). Younger stands subject to VDT when still in the stand-initiation stage may require later thinning when the canopy closes and would otherwise potentially cause growth to stagnate. Wood generated from thinning at this early stage has limited commercial value because of its small diameter.

Typically, a VDT should remove no more than 30–35 percent of a stand's basal area, although small cleared patches (0.25 ha) might be created to stimulate the regeneration of shade-intolerant species (Knapp *et al*., 2012). Slope will determine log-yarding methods (e.g. ground, cable or animal-assisted) during harvesting, but attention should be paid to the potential impacts of log extraction on the soil to avoid excessive compaction and the potential for erosion. Treatments should be developed to increase structural variability in a landscape in terms of tree density, tree size (e.g. height, diameter and canopy structure) and species diversity (Wederspahn, 2012) – increased variability translates into increased resilience (Koontz *et al*., 2020). Thinning can be followed by additional planting to increase numbers of less-abundant tree and shrub species. Restoration planting can also be conducted in degraded watersheds lacking adequate tree cover or tree species diversity.

Thinning operations should be situated in landscapes so as to minimize impacts on the water supply (e.g. away from streams). They can also be deployed to benefit habitat connectivity for rare species, buffer sensitive areas (e.g. old-growth forests, rock outcrops and wetlands), mitigate the spread of disease and reduce wildfire severity.

Forest management for municipal water supplies can help increase ecosystem resilience. Threats to forests from climate change include alterations to historical patterns of precipitation (e.g. increased periods of drought), an increased risk of wildfire, the increased spread of non-native species, and increased infestations of insects and disease. The biodiversity of sustainably managed native forests confers buffering capacity in the face of perturbations and shifts in climatic parameters. Thinning practices can help maintain the vigour of residual trees by giving them more space to capture sunlight, moisture and nutrients; this may be especially important under changed moisture regimes (Willis, Roberts and Harrington, 2018). The removal or mulching of slash created by thinning and selective harvesting will reduce the availability of flammable materials, thereby mitigating wildfire.

Ensuring multiple and expandable sources of drinking water (e.g. more than one watershed; aquifers; and desalination) provides redundancy and increases water security in the face of increasing climate variability and the demands of growing human populations (Simpson, Shearing and Dupont, 2020). Water conservation programmes aimed at consumers can help reduce wastage in water use, which may be vital, especially in times of high demand and low supply, although such programmes may reduce revenues based on water use (Spang *et al*., 2015). Box 3.3 describes the establishment and management of a forested watershed dedicated to the water supply of Seattle in the United States of America. Box 3.4 reports a study of the relationship between forest cover and the cost of water treatment for Mumbai, India.

[10] The stem-exclusion stage in forest succession is the stage at which sunlight and soil resources become limiting and additional plants are excluded.

BOX 3.3

The City of Seattle's municipal watershed

The Cedar River Municipal Watershed (CRMW) supplies roughly two-thirds of the drinking water for the City of Seattle in the Pacific northwest of the United States of America and its 1.5 million residents. The watershed encompasses 36 680 ha on the western slopes of the Cascade Mountains, ranging in elevation from 165 m to 1 650 m above sea level. The maritime climate receives 1 450–3 550 mm of precipitation annually, with winter snows having an important water-storage function in the annual water-supply cycle.

Forests cover 95 percent of the CRMW, and they occur across three distinct zones based on elevation and potential natural vegetation (Franklin and Dyrness, 1973). All three zones are conifer-dominated, and the few deciduous species present are in low abundance. Native plant diversity is relatively low, but net primary production is relatively high.

The Cedar River Municipal Watershed viewed from Rattlesnake Ledge, Washington, United States of America

The Cedar River basin was identified as a potential source of water following a fire in 1889 that destroyed the 26-ha Seattle business district. In 1901, water was diverted for the first time from the Cedar River into a pipe at Landsburg Dam for the 46-km journey to the city. Concerned about water quality, the City of Seattle started a long process of acquiring all land in the watershed above the diversion; the process was finally completed in 1996.

Management of the CRMW is driven primarily by two federal laws. One is the Safe Drinking Water Act (1974), which is administered federally by the Environmental Protection Agency and locally by the Washington State Department of Health. This law sets water-quality standards and motivates the City to keep the CRMW closed to unsupervised public access to maintain high water quality (and thereby avoid the need to build an expensive water-treatment facility). Despite a long history of settlement in the CRMW, no one lives there now, and recreation is prohibited.

The other law that drives the management of the CRMW is the Endangered Species Act (1973), which is administered federally by the Fish and Wildlife Service and the National Oceanic and Atmospheric Administration (NOAA). NOAA listed the local population of Chinook salmon (*Oncorhynchus tshawytscha*) as threatened with extinction in 1999, and the extraction of drinking water from the Cedar River was considered to conflict with the protection of this anadromous fish species. A Habitat Conservation

Plan (HCP) was developed (as required under the Endangered Species Act) in 2000 (City of Seattle, 2000) to provide certainty in water management and ensure the long-term survival of the species. The HCP also pertained to 82 other fish and wildlife species that may inhabit the CRMW and came with City-guaranteed habitat restoration funding for 50 years.

The HCP officially ended commercial timber extraction and declared the CRMW an ecological reserve, including 5 660 ha of old-growth forest. The majority of the restoration programmes funded by the HCP target the damage caused by more than a century of clearfelling for timber in the other 23 590 ha of forest in the CRMW, including by decommissioning surplus forest roads and restoring stream and forest habitats. Stream restoration includes eliminating artificial diversions and improving fish-spawning habitat by placements of large wood. Forest restoration includes thinning young forests to promote tree vigour and planting seedlings in degraded areas to improve native species diversity. These programmes also directly benefit water quality and quantity.

Recently, the management language of habitat restoration has morphed into the language of adaptation and resilience to a changing climate. Wildfire, pests, diseases and invasive species are everyday management concerns. Fortunately, the tools for habitat restoration are similar to those used to increase landscape resilience.

BOX 3.4

Deforestation-induced costs on Mumbai's drinking-water supplies

Mumbai, one of the world's most populated cities, depends for its water supply entirely on reservoirs fed by sources located far from the city and which are, in turn, dependent on forested watersheds that face the threat of deforestation and degradation due mainly to grazing, treefelling and development.

Using turbidity as a measure of raw water quality, Singh and Mishra (2014) investigated the relationship between forest cover and the cost of water treatment. They found that every 1 percent decrease in forest cover increased water turbidity by 8.4 percent and the cost of treating drinking water by almost 1.6 percent. Moreover, water losses due to backwash and desludging increased by 0.6 percent for every 1 percent of forest-cover loss. The total impact of annual deforestation on drinking-water supply, calculated as the sum of increased treatment costs and water losses, was estimated at around USD 1.3 per m^3 of treated water per ha per year.

Water-related hazard control

Forests may be considered nature-based solutions to a range of environmental problems due to their capacity to reduce the erosion of soils, riverbanks and coastlines and to mitigate natural hazards such as flooding, mass wasting, landslides, rockfalls, avalanches and storm surges. When managed to mitigate the risk of these hazards, such forests are sometimes called protective or protection forests. Protective forests are situated mostly in upland areas and along coastlines on soils prone to erosion, but riparian buffer zones and many urban and periurban forests and trees also serve protective functions (Box 3.5). Protective forests are often in highly dynamic environments and should be managed to ensure they can continue to serve their protective functions as conditions change.

The sound planning, design and management of forests and trees in urban and periurban areas can be instrumental, not only in increasing the availability and quality of water supplies to cities but also in preventing and mitigating water-related disasters. In urban areas, the optimization of tree cover can substantially increase the urban

BOX 3.5
Urban and periurban forestry

It is projected that 1.7-2.4 billion of the global urban population will face water scarcity by 2050 (He *et al.*, 2021), and the safety of many urban communities is at risk due to increasingly frequent floods and drought. By protecting watersheds, filtering water and increasing soil permeability, urban and periurban forests can make substantial contributions to sustainable urban water supplies (Nagabhatla, Dudley and Springgay, 2018). Well-managed and healthy periurban forests and other tree systems can protect watersheds, mitigate climatic extremes, support natural ecosystem processes, intercept air pollutants, reduce sediments and filter rainwater and thereby ensure the delivery of high-quality water to cities for residential, industrial and agricultural uses.

permeable surface, improve the general water cycle and facilitate water infiltration into the soil, thus reducing runoff and the severity of flooding events. Forest management should aim to ameliorate growing conditions for urban trees to minimize the stress arising from environmental pressures imposed by urban environments. The role of urban trees and forests in reducing stormwater flows may also lessen the risk of hazardous sewer overflows. Research shows that green interventions can contribute substantially to urban water management at a cost that is lower than, or competitive with, grey infrastructure projects (e.g. Copeland, 2014; McGarity *et al.*, 2015). Forested bioswales, permeable pavements, green roofs, green streets (a stormwater management approach that incorporates vegetation (perennials, shrubs, trees), soil, and engineered systems (e.g., permeable pavements) to slow, filter, and cleanse stormwater runoff from impervious surfaces (e.g., streets, sidewalks), wooded wetlands, rain gardens, bioretention, bioinfiltration, forested filter strips and linear stormwater tree pits are examples of forest-based solutions that can mitigate the impacts of stormwater runoff in cities. Some coastal tropical cities use mangroves as protective shields against the effects of coastal hazards that affect people and infrastructure (FAO, 2007), as well as to treat wastewater and remove chemical contaminants and thereby mitigate coastal pollution. An emerging concept is that of “sponge cities”, which involves planning and designing cities to maximize their capacity to absorb rainwater, which is then filtered by the soil and allowed to reach urban aquifers for subsequent extraction, treatment and reuse as part of city water supplies.

Because protective forests often grow on poor soils, they tend to be relatively susceptible to large-scale impacts from disturbances such as forest fires, windstorms, floods and insect infestations. Climate change is also an increasing threat to the protective functions of forests, given the potential impacts of temperature rise, variations in

July 2016, students working at a small garden designed to absorb rainwater in the Soundview neighborhood of the Bronx, N.Y.

Green street

Linear stormwater tree pit

precipitation and more intense storms and drought. Many protective forests are in vulnerable environments: in upland areas, for example, ecological zones are shifting with changes in temperature and precipitation regimes; in coastal areas, sea-level rise and changes in salinity loom as major threats to protective forests.

Protective forests in mountain areas are often even-aged – because generally they are established after disturbance events – and have limited tree species diversity. As these stands age, they become more susceptible to pests, diseases, forest fire and other risks, which can decrease their protective function. Land-use conversion is often less of an issue because of the impracticality of developing land on steep slopes but, on the other hand, such areas are susceptible to grazing pressure from both domesticated and wild animals, which can lead to poor regeneration outcomes.

Ecological regimes are shifting because of climate change, with treelines advancing upslope a widespread phenomenon (Greenwood and Jump, 2018), including in areas prone to avalanches, rockfalls, floods and landslides. Forests reduce the risk of avalanches and rockfalls because tree canopies, trunks and root structures buffer the kinetic energy of falling snow and rocks, thus reducing their downslope speed. Moreover, soils covered by trees or shrubs generally have higher water-retention capacity than other vegetation, with the effect of reducing surface runoff and erosion and increasing soil infiltration and permeation. Mountain soils with forests are often deeper than other vegetation types, with high organic content and water-storage capacity. Depending on the type, intensity and frequency of precipitation events, forests generally reduce local flooding and torrents in upland areas. Thus, peak discharges in forested catchment areas are generally lower –

BOX 3.6

Risk-based forest management

In France, forest management planning is based on the Hazard Control Index (Indice de Maîtrise d'Aléa – IMA), which quantifies the protective role of forests from 0 (the vegetation has no effect on reducing the hazard) to 6 (maximum effectiveness). The index was developed as part of a national programme to renew protective stands launched in 2005.

The National Forests Office estimates risk based on a rating grid in which the indicators (percentage of plant cover in summer/winter, as well as density and diameter) reflect the hazards under consideration (e.g. rockfall, avalanche, surface erosion, torrential flood and landslide). Although not applied exclusively in mountain areas, the index is particularly useful in such areas, where forests have important protective roles.

After applying the IMA and mapping hazard zones in 555 600 ha of state-owned forests, it was found that most protective forests were more than 100 years old and required significant effort to maintain their protective functions. The 7 percent of forests defined as having high protection potential were prioritized for restoration in the first phase, which was completed in 2011. A second phase for the renewal of protective forests according to the IMA grading has commenced, at a cost of EUR 3 million–4 million per year (Dubois, Marco and Evans, 2017).

and with a time lag – compared with non-forested catchments, meaning a reduction in the potential for downstream flooding.

The capacity of a forest to reduce landslides depends on factors such as slope, soil depth and type, and the type, frequency and intensity of precipitation events (Segura, Ray and Maroto, 2014). Forests have been shown to mitigate the effects of shallow landslides (2 m depth). On the other hand, driftwood from lateral erosion, avalanches, landslides, windfalls and flooding can increase the risk of debris flows by forming log jams.

Targeted, site-specific forest management designed to maintain permanent forest cover, including efforts to encourage regeneration, species diversity and uneven-aged stands, can increase forest protective functions. Such management may include risk-based planning (Box 3.6); the exclusion or minimization of grazing; thinning; the management of coarse woody debris; and shelterwood regeneration and restoration. Management should aim to optimize the species diversity of forest stands, bearing in mind likely future climatic conditions and associated shifts in ecological zones. Ultimately, uneven-aged, mixed-species stands are likely to be more resilient to natural and human disturbances and thereby better able to serve their protective functions.

Deltas and other coastal areas are in constant flux between erosion and deposition. A major function of rivers and streams is to transport sediments that help shape aquatic habitats downstream, including floodplains, deltas, salt marshes, mangroves and other coastal ecosystems. The quantity and rate of flow of sediment downstream can be regulated at least partially by upstream forests, which can slow water movement and trap sediments. When there is either too much or too little downstream sediment supply, however, coastland accretes or erodes.

Native mangrove species are adapted to particular levels of salinity, as regulated by accretion, erosion and incoming freshwater flows. Shifts in freshwater quantity and timing can therefore have negative impacts on mangrove forests. Changes in mangrove forests can lead to dramatic shifts in aquatic biology. Many juvenile fish, for example, use mangroves for feeding and rearing; coastal shrimp fisheries tend to be highly associated with the extent and quality of mangroves, as well as freshwater flows. Deltas and river mouths are crucial parts of the life cycles of anadromous species such as hilsa (and other shads).

Agroforestry

Agroforestry is a valuable option for achieving the sustainable use of water in agricultural lands. By increasing ground cover and soil organic matter compared with monoculture food crops, well-designed agroforestry systems can reduce water runoff and soil evaporation and increase water infiltration rates and soil-retention capacity (Bayala and Wallace, 2015; Anderson *et al*., 2009). This, in turn, increases the biomass of trees and crops produced per unit of water used, improving overall water productivity, particularly in areas where water is scarce (Ong, Black and Muthuri, 2006).

By shading crops with their canopies and protecting them from winds, trees on farms can reduce soil evapotranspiration and help maintain soil moisture, with consequent benefits for crop productivity. Trees planted along contours can help reduce water runoff and stabilize soils. Alley-cropping systems, homegardens and plantation–crop combinations all have higher rates of water infiltration and retention capacity than monocultures due to their production of tree litter and the use of branch prunings as mulch to increase soil organic matter and consequently water retention. Quesungual and Kuxur Rum (in Honduras and Guatemala, respectively) are agroforestry systems developed as alternatives to slash-and-burn agriculture to increase productivity in hilly areas in the Dry Corridor of Central America. In both systems, plots are cleared of vegetation manually and the cuttings are shredded and distributed on the soil surface as mulch. By retaining tree root systems, permanently covering soils and increasing soil organic matter, these systems enable the infiltration, retention and conservation of large volumes of water over long periods while also reducing surface runoff and soil erosion.

Agroforestry is a proven strategy for mitigating water-quality losses arising from intensive agriculture. Riparian buffers on the borders of agricultural fields intercept and remove contaminants from surface runoff and shallow groundwater that might otherwise reach water bodies (Bayala and Prieto, 2020). Agroforestry systems in upland buffers can also help reduce soil erosion and nutrient losses in pastured watersheds, thereby protecting water quality. A study in an area subject to a watershed regreening project since the 1970s in South Sulawesi, Indonesia, found that the project's agroforestry interventions enabled the watershed to remain ecologically healthy for at least the next two decades (McNie *et al*., 2008).

Possible competition for water should be taken into account when designing and establishing agroforestry systems. Two strategies to minimize declines in crop yields are selecting tree species that are complementary in their water use with crops grown on the same land and deploying appropriate tree management interventions to minimize competition between trees and agricultural crops (Cannell, Van Noordwijk and Ong, 1996; Ong, Black and Muthuri, 2006). Complementarity may be either spatial (e.g. when trees exploit nutrients and water that are inaccessible or not required by the crop) or temporal (with the main demand for water occurring at different times for trees versus crops). Generally, faster-growing trees use more water and deep-rooted species reduce dry-season flows. Tree-pruning and reducing tree density can be valuable management options for minimizing transpiration and thus tree water demand. Tree species with low water demand should be used in environments where water is scarce – such as in arid and semiarid climates, where water availability is a main constraint to production. Sites should be selected carefully for the establishment of agroforestry because the extent to which such systems intercept and treat waterflows is partly determined by local soils, topography, surficial geology and hydrology (Tomer *et al*., 2009).

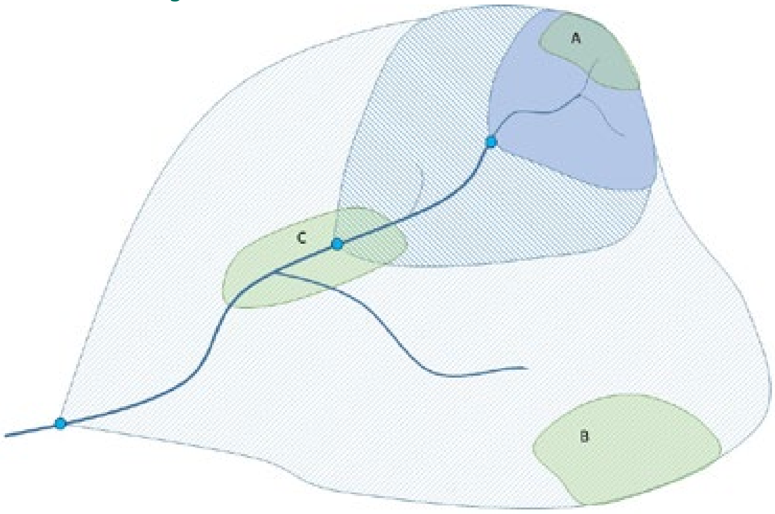
WATERSHED-BASED FOREST MANAGEMENT

When precipitation – in any form – occurs, the water might begin moving downstream immediately, or it might be stored temporarily as snow and ice or in soils. As runoff, water moves in surface waters such as rivers and streams and may be stored in lakes and wetlands, and it may also enter longer-term storage as groundwater in aquifers. In any case, water falling as precipitation in a watershed ultimately flows downstream through the river network. Much of this is visible as surface water, but some water also flows as subsurface (i.e. water that stays below the soil surface) or hyporheic flows through gravel and rocks below the surface and alongside streams. Forest management influences surface water as well as subsurface flows and long-term groundwater supply.

The quantity, quality and timing of water supply are intricately linked to the condition of the watersheds in which water is stored and through which it flows. Watersheds are subject to many biological, socio-economic and physical processes active in landscapes (Beechie *et al*., 1996; Dobrowolski and Thurow, 1995). Watersheds are a convenient unit for restoration and management planning because they can be identified on maps and from remotely sensed data and because they do not change much over time (Reid, Ziemer and Furniss, 1996; Bohn and Kershner, 2002).

The role of forests in ensuring the maintenance of water values differs according to their location in a watershed, therefore requiring differing management approaches (Figure 3.2). Forest management decisions need to consider factors such as the regulation of water temperature and flow, water quality, and downstream fisheries at the watershed scale. For example, a decision-support system in place for the Lewis River basin in Washington, United States of America, enables managers to estimate the influence of restoration actions in different parts of a watershed covering 270 900 ha on multiple downstream ecosystem services (Steel *et al*., 2008).

FIGURE 3.2
Schematic diagram of three nested watersheds in a river network



Notes: Blue lines indicate the river system and the three blue polygons represent the nested watersheds draining to each of the bright blue points on the river system. Green polygons A, B and C indicate potential forested patches in a watershed context. The management of headwater forest A has a strong influence on water supply at the most upstream point; a medium influence on water supply at the middle point; and a lesser influence on water supply at the most downstream point. Forested area A also contains a headwater stream, which is an additional management consideration. The management of forested area B lies in the headwaters of the most downstream point. Of the three points identified in blue, the management of forested area B only has a direct influence on the most downstream point, but many other points could be identified on the river network that would be influenced by the management of forested area B. Forested area C contains large sections of riparian forest and, most likely, floodplain forest. Management considerations in this area might therefore differ and would directly concern conditions at the middle point as well as water supply at the most downstream point and river stability at the confluence of the two largest forks.

Watershed delineation. The identification of watershed boundaries at a national scale is an important first step in effective water management because it enables the consideration of forest management in a watershed context. The United States Geological Survey uses a nested watershed scheme to classify the entire United States of America into hydrologic units (HUs) identified by HU codes (Figure 3.3). The shortest codes represent the largest basins – usually well-known and named river basins. It is possible to telescope down within each HU to smaller nested HUs representing smaller river systems within each larger watershed (Figure 3.4).

FIGURE 3.3
Four-digit hydrologic unit codes identifying major river basins, United States of America

Note: Increasingly small watersheds are identified by increasingly long numeric codes that represent the inherently nested nature of watersheds (see Figure 3.4).
Source: USGS (2018a).

FIGURE 3.4
Nested structure of watershed boundaries, United States of America

Source: USGS (2018b).

Riparian forests

Riparian forests play a clear role in regulating water services (Boggs, Sun and McNulty, 2015) and are crucial for the long-term maintenance of downstream water quality. They are transitional between terrestrial and aquatic ecosystems and distinguished by gradients in biophysical condition, ecological processes and biota. Surface and subsurface hydrology connect waterbodies to their adjacent uplands in riparian zones, with significant exchanges of energy and matter between terrestrial (upland) and aquatic (lowland) ecosystems (Quinn, Wilhere and Krueger, 2020).

Riparian forests dissipate energy and attenuate overland flows during flooding (Bentrup, 2008). Forested riparian buffers confer resistance on bank erosion and supply woody debris to streams, which serves to create pools and backwater habitats that reduce water velocity in high-water events. Riparian buffers along streams, and protected floodplain forests, help maintain resilient stream systems that are more resistant to unexpected increases in discharge (Boggs, Sun and McNulty, 2015).

When managed to increase and maintain water values, riparian forests also provide many co-benefits, including for recreation and tourism (addressed in detail below). Intact riparian forests increase stream channel and riverbank stability (Hupp and Osterkamp, 1996; Hubble, Docker and Rutherford, 2010) and help regulate watertables (Burt *et al*., 2002; Schilling, 2007). Riparian forests provide habitat for both terrestrial (Williams, O'Farrell and Riddle, 2006; Gillies and St Clair, 2008) and aquatic species (Fausch *et al*., 2002; Stanford *et al*., 2019; Quinn, Wilhere and Krueger, 2020) and often feature plant species that do not grow beyond riparian areas. They provide seasonal habitat for many species that cannot live year-round in drier upland areas (Stromberg *et al*., 2013). Insects, seeds and detritus that fall into the water from trees provide food and nutrients to sustain aquatic life. The root systems of riparian vegetation provide shelter and habitat for fishes, and tree shade helps keep water cool in hot conditions.

The removal of riparian canopy increases the amount of sunlight hitting the water, increasing photosynthesis among water weeds and algae and raising water temperatures; increases in organic nutrients can also act to alter the trophic balance of phytoplankton and zooplankton and shift aquatic communities towards fast-reproducing generalist species. High levels of nutrients in the water may favour fast-growing, opportunistic species of water weed, which can shade out and smother habitats, slow down waterflows (exacerbating sedimentation) and, in extreme cases, lead to the deoxygenation and stagnation of water. Many aquatic species have relatively small tolerance ranges for pH and alkalinity and have adapted to the particular systems in which they live. Changes in land cover (such as deforestation) and associated leaching can have major impacts on pH and consequently cause the loss of sensitive species and, in more extreme cases, of entire ecosystems (e.g. the loss of almost all biodiversity in highly acid lakes and rivers).

Floodplain forests

Floodplain forests grow in river valleys that receive alluvial deposits from frequent flooding; they usually differ from upland forests in structure and species composition (Yin, 1999). Floodplain forests typically comprise herbaceous plants, small tree species, saplings, shrubs and canopies of mature trees that dominate the community (Yin, 1999). Because they are adjacent to and part of river systems, floodplain forests generally consist of a wide range of vegetation types associated with the amount and duration of water inundation (Hamilton *et al*., 2007). Among the many ecosystem services produced by floodplains and floodplain forests are sediment and nutrient retention, carbon sequestration and groundwater recharge (Opperman *et al*., 2017). These services are performed through the interaction of discharge events (whether low- or high-flow) and a given forested floodplain (Opperman *et al*., 2017). This terrestrial–aquatic interaction, based on the discharge amount, duration, frequency, magnitude and residence time of water, helps determine the quantity, quality and timing of downstream water supply. The influence of forested floodplains varies not only with flood frequency and inundation but also with the timing and predictability of flows (Opperman *et al*., 2017). Thus, climate plays a role in regulating the impacts of floodplain ecosystems on water quality and quantity. Floodplain ecosystems occur in association with (among other things) tropical seasonal, temperate seasonal, temperate aseasonal, boreal, ephemeral desert and alpine rivers and streams (Winemiller, 2004; Opperman *et al*., 2017).

Tropical seasonal floodplain systems include some of the world's largest floodplains, such as the Amazon, Mekong and Congo rivers (Opperman *et al*., 2017). Flooding in these systems is often predictable and seasonally long-term (i.e. months) and may encompass large swathes of forested floodplains. Sediment and nutrient retention and the ability to recharge groundwater, particularly in the forested areas of tropical seasonal floodplains, can be higher than in open-water areas (Smith *et al*., 2000).

Temperate seasonal floodplain ecosystems are more variable than tropical and subtropical systems in their flooding extent and timing, which are linked to latitudinal and elevation differences with respect to the timing of peak discharges (e.g. between snowmelt-dominated and rainfall-dominated systems) (Winemiller, 2004). Large storms and flooding may occur at any time of the year in seasonal temperate floodplain systems such as the Brazos River (Texas, United States of America) and Australia's Murray–Darling river system (Opperman *et al*., 2017). In both these temperate systems, forest floodplains play key roles in the retention, assimilation and integration of sediment, nutrients and water chemistry, as well as provide benefits for fish and wildlife (Johnston, 1991; Opperman *et al*., 2017).

Boreal rivers and their associated floodplain systems are subject to an additional type of flooding caused by ice-jam breakups that can elevate flood levels, increase the amount of suspended sediment and alter water-quality parameters, including pH and metals content (Peters *et al*., 2016). Floodplain forests in boreal systems play a crucial role in the distant transport of dissolved organic carbon, bank stability, food-web dynamics and the maintenance of upland tree species (Peters *et al*., 2016).

Short-term desert streams have unique floodplain systems due to their flash-flood hydrology (Grimm and Fisher, 1989), and exchanges between surface and subsurface flows influence the riparian vegetation (Grim and Fisher, 1989). Desert floodplain systems, and their water and biogeochemical exchanges, can differ considerably – for example, some systems have little or no vegetation and an abundance of coarse channel sediments and others have extensive emergent vegetation that includes wide sections with slow-moving waters (Heffernan, 2008).

Regardless of the system, the connection between riparian zones and rivers is crucial for maintaining hydrologic functioning, which translates into geomorphic and ecological functioning and ultimately into the quality and timing of downstream water supply.

Many floodplain systems have been altered, simplified and compromised to the point where numerous functions are no longer provided (Winemiller, 2004; Opperman *et al*., 2017). Efforts to restore such systems have been ongoing for decades (Opperman *et al*., 2017), but recently the focus has changed from character-specific or technique-based restoration actions to the restoration of riverine and watershed processes (Beechie *et al*., 2010; Wohl, Lane and Wilcox, 2015; Powers, Helstab and Niezgoda, 2019). Such process-based efforts that restore physical connectivity between stream channels and floodplains and the natural diversity and variability of flow and sediment regimes are more effective in restoring ecological functions (Cluer and Thorne, 2014; Wohl, Lane and Wilcox, 2015; Powers, Helstab and Niezgoda, 2019). The connection between river and floodplain is crucial because water is conveyed and stored both at the surface and in subsurface areas, the latter of which are difficult to see and assess. The majority of water retained and conveyed in an alluvial-dominated watershed is typically through subsurface portions of the stream network, including hyporheic areas associated with floodplain forests (Stanford and Ward, 1993).

A wide range of human activities can cause stream-channel incisions and a subsequent reduction in water storage and conveyance capacity due to the disconnection of floodplain from their rivers, such as upstream dams that cut off sediment supply and flows; urbanization that causes larger, more frequent flow events; and forest clearing, which can lead to larger and more frequent flow events, direct disconnection from stream channelization, and the loss of in-stream wood (Abbe *et al*., 2019).

To reverse the disconnection of floodplain forests, stream-restoration specialists propose the use of GIS and field-based analyses to develop maps of predisturbance valley surfaces. These maps can guide the filling and removal of valley bottoms in ways that enable river, floodplain and valley to re-establish surface and subsurface connections that allow natural ecosystem processes to re-emerge (Powers, Helstab and Niezgoda, 2019).

Forest plantations

Forest plantations are an intensive form of planted forest, usually established with the primary objective of wood production. Trees grown in fast-growing plantations typically consume large quantities of water, although they are usually efficient in the production of wood per unit volume of water.

One of the most important aspects to consider in forest plantation management is regional water availability. Calder (2007) proposed an initial framework for zoning water availability, and Ferraz *et al*. (2019) developed a modified decision framework for assessing water availability in fast-growing plantations. There are three broad scenarios of water availability to guide decisions on forest plantation establishment and management:

1. **Low water availability –** in regions with water insecurity for most or all of the year, forest plantations are not recommended due to the high risk of conflicts over water (Box 3.7).

BOX 3.7

Management techniques for forest plantations in areas at risk of conflicts over water

Any technique for minimizing the risk of water-related conflicts over forest plantations will involve economic trade-offs. Not every technique will be applicable at a local level; forest managers should develop their own site-specific prescriptions to reduce water consumption and ensure the maintenance of water values, with independent monitoring.

Create land-use mosaics. For reasons mainly related to transport logistics, it is common to establish large forest plantations around mills and log yards, thus concentrating related hydrologic impacts in catchments (Garcia *et al.*, 2018). These impacts can be reduced by creating occupation land-use mosaics in which forest plantations are intermixed with areas of conserved natural vegetation and agriculture. Land-use mosaics help dilute the impacts of forest plantations in space and time (Ferraz *et al.*, 2014), although the best outcomes will be achieved with a cohesive approach among land uses because water moves through landscapes and all land uses have impacts on water.

Extend rotations. Wood can be grown in very short rotations in forest plantations in tropical and subtropical areas, but caution is required because the very short intervals between harvests, and the intensive inputs of resources such as fertilizers, can increase the impacts of management on water services. Longer harvesting cycles are better able to meet most water management objectives because they increase the interval between disturbances, and more mature plantations are less water-demanding.

Reduce management intensity. The managers of forest plantations use a range of approaches to encourage high wood productivity, including the application of fertilizers, the use of pesticides and high-density tree-planting; these and other tools can affect water values. Techniques that can reduce the water-related impacts of forest plantations include the adoption of best practices in soil and water conservation; the on-site spreading of harvesting residues; reducing the use of agrochemicals; and the construction of new plantations (Gonçalves *et al.*, 2017). The water-use efficiency of the species used (Stape, Binkley and Ryan, 2004) and tree density (Hakamada *et al.*, 2020) are important considerations in reducing the impacts of forest plantations on water quantity.

Increase genetic and species diversity. Industrial-scale forest plantations are usually monocultures, and the most used species are *Eucalyptus*, *Pinus* and *Acacia* and their hybrids. Planting stock often comprises clones of improved tree hybrids, further reducing genetic variability. In some areas, mixed-species plantations have had beneficial interactions in relation to the use of water and nutrients (Forrester *et al.*, 2010). The diversification of species and age classes in stands can reduce total stand water use because trees of different species and age use water differently.

Reduce the size of clearfelling coupes. Forest plantations are usually subject to clearfelling over relatively large areas to optimize the logistics of mechanized harvesting and log transportation. Using smaller coupes to create mosaics of cut and uncut areas and diverse stand ages in a catchment or watershed can increase the consistency of water yields over time and may be particularly important in areas with fragile soils or steep slopes (Stednick, 1996).

2. **Intermediate water availability** – in regions where water is relatively plentiful but periodic water insecurity can occur, there remains a risk of water-related conflicts, depending on the severity of water limitations, seasonality, and competition for water, such as from urban centres and agriculture uses. In such situations, water use should be monitored and management techniques used to reduce water use when necessary.
3. **High water availability** – forest plantations are likely to face only a low risk of water conflicts in areas where water is abundant. Good practice in forest–water management will include providing appropriate buffers in riparian zones; properly designing, building and maintaining roads; ensuring the rapid re-establishment of plantation areas after harvesting; minimizing the use of pesticides, fertilizers and fungicides; and minimizing soil erosion.

Restoration

Forest restoration efforts should take into consideration their potential impacts on water quantity, quality and timing. Rapidly growing trees may diminish the quantity of water available for other purposes, and poorly managed planted forests with high seedling mortality may facilitate the intrusion of invasive species or suffer disturbances due to fire, grazing, pests and disease, with deleterious effects on water values (Filoso *et al.*, 2017).

Forest restoration generally focuses on building forest structure, but the development of soils that enable water infiltration is equally important. Improving forest soils may take longer than the forest structure to develop (Lozano-Baez *et al.*, 2019), and the benefits of forest restoration for water-yield regulation and water quality may take longer to realize in planted forests than in naturally regenerating forests.

Planted forests are most water-demanding in periods of rapid growth – typically in young forests before the canopy closes. When a new forest is first planted, its trees all grow rapidly simultaneously, with a consequently high volume of water use; mature planted forests, on the other hand, use less water because of their reduced growth. Thus, the establishment of a planted forest is likely to cause an initial reduction in the total available volume of water. Most transpired forest water eventually re-enters soils through precipitation – but it is not possible to control where this will be. If water quantity is a management priority, options for reducing forest water consumption include reducing the density of trees planted on a site and planting in mosaics across a catchment (Bonet *et al*., 2012). Compared with other land uses, forests – including planted forests – generally increase the quality and predictability of water yields; moreover, although absolute yields of water quantity may be lower, the quantity of useful water is generally higher.

Forests and water supply at watershed scales

The impacts of forest management on the timing of water supply varies considerably across ecosystems. In areas where snowmelt is not a factor, timber harvesting is less likely to have a measurable effect on the severity of flooding because, even in natural conditions, forest canopies and soils are effectively saturated in large storms; therefore, a reduction in canopy cover and the interception of precipitation has less influence on flood size. In areas with snow, road networks, snow-generated flows and snow followed by rain can substantially increase the risk of high flows, particularly in the first ten years after timber harvesting (McCabe, Clark and Hay, 2007). Less is known about tropical ecosystems because these have very different patterns in precipitation and drivers of forest evapotranspiration and are less studied.

In general, the most pronounced effects of forest management on peak flows have been observed in small streams (e.g. Bosch and Hewlett, 1982), where even brief storms over small areas can saturate the relatively short flow paths. Storm intensity often varies across large drainage basins, and floodwaters from individual tributaries may be out of phase in reaching river mainstems. Moreover, harvest area decreases as a percentage of the total area of a catchment with increasing catchment size and it is increasingly difficult, therefore, to detect increases in peak flows due to timber harvesting.

In theory, forest–water management should aim to produce yields of the highest useful, economically feasible volume of water while maintaining the yields of other forest goods and services. Putting in place simple rules and regulations that are agreeable to all and that can be applied consistently is usually the best way to achieve this. At an experimental scale, it may be possible to limit overall leaf area and increase water yield while also maintaining high wood yields (e.g. through extensive, well-planned intensive harvesting). In practice, however, social priorities and the diversity of landowners, as well as the desire to maximize water quality, mean that it is rare for forests to be managed to obtain a specific leaf area with the aim of maximizing water yields (Evaristo and McDonnell, 2019). More commonly, forest regulations and requirements allocate certain land areas as forest or non-forest for this purpose, or certain tree species in planted forests perceived or known to use more water may be restricted or banned.

Many of the principles of watershed management are universal. The interaction of the environment and communities may vary by endemic species, climate and culture, but the need to find a balance among competing watershed resource demands remains constant, as illustrated in Box 3.8.

Baseline information is crucial to the successful development and implementation of forest–water management operations. Its absence can lead to unwanted outcomes, as illustrated in Box 3.9.

BOX 3.8

Comparing the Phetchaburi watershed, Thailand, and watershed-scale planning in the United States of America

In Thailand, the Watershed-based Adaptation to Climate Change project was a collaborative watershed-scale planning initiative to evaluate vulnerability to climate change and create an adaptation plan for the Sirindhorn International Environment Park in Cha-am District, Phetchaburi Province (Long and Steel, 2020). The park is located in one of the driest parts of Thailand in a region important for agriculture and tourism. Watersheds have been modified extensively, including by numerous reservoirs and diversions to support development and agriculture. The project identified many concerns related to forest–water management, including a belief that a loss of forests in the last century had contributed to a decline in the water supply.

The assessment and planning done under the project were based on multiple sources of information, ranging from highly quantitative climate downscaling work across the entire Phetchaburi River basin to predict changes in water availability, to qualitative interviews and meetings with community members and leaders to understand impacts on local economies. The Sustainable Development Fund (SDF) collected field data in communities selected to represent the main economic sectors in the watershed. In the upper watershed, the SDF focused on a Karen community, which was in conflict with the government over land rights and agricultural activities (as were other indigenous peoples in mountainous forested areas). In the central watershed, the SDF identified four villages that relied on particular cash crops and two communities struggling with urban expansion and water-supply management. In the lower watershed, the SDF focused on three communities suffering from floods and droughts and a community in which many livelihoods depended on coastal salt-farming. The methodology explicitly considered both climate and non-climate factors contributing to vulnerability. A collaborative approach made it possible to identify climate indicators that were meaningful to the communities related to the duration of flooding and dry spells that influence crop success.

The project showed the need to consider interactions between forests and trees in various areas and sectors in the basin as well as the human element in managing across large spatial scales. Challenges for sustainable forest management included setting management targets for agroforestry conducted by marginalized groups in forest headwaters; securing and balancing water supply for direct human use; and assessing downstream instream flow requirements for ecological purposes.

Tensions between ecological and social goals and between headwater and downstream communities have long complicated efforts to manage forests for water in the western United States of America. In the 1960s, for example, the State of Arizona and private water users sponsored projects to increase water yields from forests, including on tribal lands, through a process involving clearcuts in high-elevation forests; dragging heavy chains and spreading herbicides to clear juniper from rangelands; reseeding with non-native grasses; and cutting, girdling and poisoning both native and non-native riparian vegetation. These efforts were sold as a win–win situation for tribal communities and downstream water users but led to bitter fights, court battles and a legacy of distrust.

Changes in societal values in the United States of America moved watershed management away from large-scale vegetation manipulation to increase water yields and towards restoration to sustain ecological functions and biodiversity. Watershed rehabilitation efforts have continued to include soil-erosion control, including the treatment of gully erosion. In recent years, there have been calls to thin forests that have become dense as a result of fire suppression. The canopy openings created by thinning will help retain snow as a means of mitigating expected declines in snowpack and increase the resilience of terrestrial and aquatic communities to drought, insect outbreaks and wildfire (Harpold *et al.*, 2020).

These two examples illustrate the challenges in developing forest management regimes that promote ecological sustainability and ecosystem services for the benefit of society without disadvantaging particular communities or eroding public trust. Well-integrated science, and public engagement, are important for informing such regimes.

Asian elephants in headwater forest in the Phetchaburi watershed

BOX 3.9

The Sumberjaya watershed, Sumatra, Indonesia

In Indonesia, large tracts of land were classified in colonial days as protection forest, based largely on hillslope. An attempt was made in the 1980s to improve the delineation and classification of protection forest based on slope, elevation, rainfall and soil. This effort was formalized in the forest-land-use-by-consensus project, *Tata Guna Hutan Kesepakatan* (also known as the TGHK map). In the absence of high-quality soil maps and a dense rainfall measurement network, however, the new delineations were based primarily on slope and elevation.

Research uncovers the importance of geology at the watershed scale. Land-use allocations in Indonesia are particularly important because of investments in a large hydroelectricity expansion programme. A multiyear research and development project in Sumberjaya compared erosion at the plot and subcatchments scales to assess the source and quantity of sediment ending up in a small storage lake in front of a recently constructed hydropower dam. Catchment-wide, the most frequent land-use types were shade coffee, monoculture coffee, forest and paddy rice.

The project produced surprising results: the largest net contributors of sediment were the Way Besai tributaries originating on the northern flanks of the central Bukit Rigis mountain, which also had the largest amount of forest cover. The sediment yield at the catchment scale exceeded soil loss at the plot scale (on a per unit area basis) by a factor of 3 to 10. Landslides, riverbank erosion and the concentrated flow erosion of small footpaths were the dominant erosive processes explaining soil loss at the catchment scale.

Implications for managing forests for water supply. Efforts to understand why the forested headwaters contributed sizeable amounts of sedimentation pointed to the importance of underlying geology. Topography and lithology control sediment production in Sumberjaya and are more influential than land use. Verbist *et al.* (2010) illustrated a clear mismatch between the geologically sensitive areas of the watershed and the protected forest area. Given the large extent of volcanic areas in Indonesia, it is a safe assumption that the above conclusions hold for many other catchments as well.

Although one of the two most important factors, lithology is not included sufficiently in planning and research on water supply. Often, measured plots and catchments are small, or there is little variability in soils between studied areas.

Although policies based on clear and simple criteria (e.g. the TGHK map) are appealing for their ease of implementation and transparency, the lack of high-quality baseline data can jeopardize their value. In the case of Sumberjaya, the application of simple criteria in the 1990s led to violence against and the expulsion of indigenous peoples (Kerr *et al.*, 2017). It serves as an illustration of the risk of top-down regulations and the importance of incorporating watershed processes in forest management and restoration.

THE CO-BENEFITS OF MANAGING FORESTS FOR WATER

Carbon co-benefits

The emerging need to maximize terrestrial carbon sequestration creates a challenge for forest–water management. Information on water yield and carbon storage is essential for meeting management objectives. Forest-based carbon sequestration is generally most rapid in the humid-tropical and temperate regions with favourable growing conditions for trees, including high water availability; at the other extreme, arid climates have limited capacity for rapid forest-based carbon sequestration. In most environments, the need to optimize both water values and carbon will increasingly require the evaluation of trade-offs.

Tree biomass can store carbon for long periods (potentially centuries; Box 3.10) while maintaining beneficial functions in the provision of water services. In most environments, an increase in evapotranspiration (e.g. in a regenerating forest) will produce lower water yields across a landscape but result in an increase in carbon sequestration, and mature forests can store large quantities of carbon. Timber harvesting can temporarily increase the water yield from a forest area, although this will vary depending on factors such as forest type, harvest intensity and climate, and shift carbon storage from the trees to harvested wood products.

BOX 3.10

Managing forests for carbon in Alaska, United States of America

Alaskan coastal forests store the largest amount of carbon per unit area in the world when soils are included in the total (Heath *et al.*, 2011; McNicol *et al.*, 2019), with the carbon stored in large-stature conifers exceeded by belowground carbon storage in soils (Leighty, Hamburg and Caouette, 2006). Alaska's coastal forests are regarded as a carbon reservoir, but active management occurs in specific management zones on both public and private lands. Managed forests in the region are routinely treated to reduce tree density due to the vigorous regeneration of young-growth trees. The associated reduction in aboveground biomass and the decomposition of thinned trees reduces the total carbon at treated sites (D'Amore *et al.*, 2015). The net negative carbon accretion is brief, however, and is followed by the rapid accumulation of carbon in aboveground biomass at a rate of about 5 tonnes of carbon per ha per year (D'Amore *et al.*, 2015). The combination of low air temperature, abundant precipitation and low population density mean there is little conflict between maximizing forest carbon sequestration potential and water quantity. Additionally, the accumulation of above- and belowground carbon protects the ecosystem from soil erosion and potential reductions in stream water quality. This case study shows that optimizing for both carbon and water is achievable in some ecosystems.

Biodiversity and food system co-benefits

The availability of clean water, especially during periods of limited rainfall, is essential for supporting and maintaining diverse terrestrial wildlife communities. Water from forests also plays a central role in maintaining aquatic biodiversity in both river networks and nearshore systems (Box 3.11).

The living biodiversity associated with aquatic ecosystems – whether boreal, temperate or tropical and from montane headwaters to floodplains, swamps, wetlands and deltas – is adapted to local conditions of water quality (e.g. related to temperature, mineral content, pH, oxygenation, turbidity and nutrients), quantity and flow timing. In any aquatic system, conditions are determined by elevation and latitude, vegetation cover, soil conditions and climatic factors, especially the form and timing of precipitation.

Healthy aquatic biodiversity and fisheries require a predictable supply of clean water. High levels of sediment, for example, interfere with the gills and respiration of aquatic animals, particularly at younger life stages and in sensitive species. Sediments can also physically smother eggs and juveniles as well as plants, either killing them or reducing their ability to grow and reproduce. High turbidity due to suspended solids reduces light penetration and therefore the productivity of plants and phytoplankton, altering food webs and reducing overall productivity; accreting sediments alter substrates and their associated benthic life. Most commonly, major reductions in water quality can cause the displacement of sensitive species to the extent that affected aquatic communities may ultimately be composed of only a few hardy generalist species. Reductions in water quality may result in the complete loss of fish and amphibians in some clearwater systems. Intact forests, and sustainable forest–water management, can help prevent these types of changes.

BOX 3.11

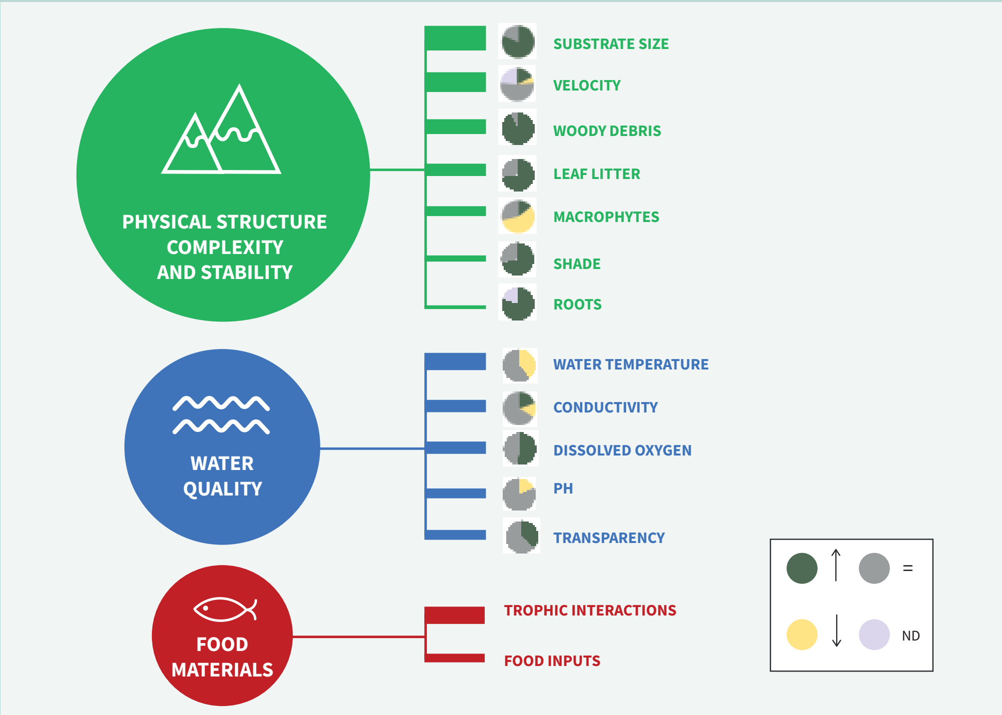
Links between forests and freshwater fish in the tropics

A recent literature review by Lo *et al.* (2020) revealed the myriad ways in which forests interact with freshwater ecosystems and how this influences freshwater fish communities in the tropics.

- *Forests and physical habitat:* forests are important for maintaining fish diversity by increasing the heterogeneity of freshwater habitats. Assessing the functional traits of fish species can help to better understand ecological responses along forest gradients and predict which species are most at risk from land-use change.
- *Forests and water quality:* there are divergent findings in the literature on the influence of forests on water quality in the tropics, which could be due to differences in methodological design. Nevertheless, studies have found that shading by riparian forests causes changes in water temperature, which, in turn, affect aquatic biodiversity (Figure 3.5).
- *Forests and food materials:* terrestrial inputs into aquatic environments are more abundant in forested environments than in non-forested areas, leading to a higher abundance of aquatic plants and insects. The feeding traits of individual fish species are likely to influence fish community dependence on forests and responses to land-use change.
- *Scale:* in the tropics, riparian forests play a role in the local physical structure of freshwater habitats, and other functional roles of forests may be observed at the landscape or watershed scale. Studies suggest that the effects of forest cover at the catchment and landscape scales are ultimately mediated by the ecological condition of local riparian buffers that determine overall species composition. The impact of deforestation events on fish populations may not be immediately detectable due to time-lag effects.

FIGURE 3.5

The strength and relationship of correlations between tropical forests and freshwater environments, broadly categorized into physical structure, water quality and food



Notes: The thickness of the lines represents the number of responses in which the linkages between forests and the characteristics of freshwater habitats were measured. The pie charts show the proportion of responses to forest cover/presence that were positive (dark green), negative (yellow), null (dark grey), and not determined (ND; light grey).

Source: Modified from Lo *et al.* (2020).

Changes in flow have both dramatic and subtle effects on aquatic biodiversity. The drying of upland rivers and streams due to reduced dry-season flows will result in the loss of aquatic wildlife, the populations of which may not recover if the breeding stock in a watershed is lost. Spate flows caused by a loss of water-retention capacity can cause erosion, with consequent impacts on aquatic wildlife.

The maintenance of downstream water supplies requires connected river networks. The partitioning and disconnection of aquatic systems has a rapid and extreme effect on species that require waterflows and linkages across the system for the transport of foods, oxygenation, breeding and movement. Many species require seasonal low or high flows to move upstream for breeding or to transport their young downstream to habitats suitable for feeding and growing. Interruptions in connectivity can effectively eliminate migratory fish and invertebrate species from a river system. An extreme example of such interruptions is the damming of rivers, but other less drastic changes can also have strong effects; for example, stream crossings by roads using culverts of insufficient size can effectively partition and disconnect streams and rapidly change the balance of aquatic life. On the other hand, there are opportunities to create synergies between biodiversity and freshwater services (Box 3.12).

BOX 3.12

Biodiversity and freshwater: synergistic ecosystem services

Studies have identified synergies between biodiversity and freshwater services at multiple scales. A multicriteria analysis by Larsen, Londoño-Murcia and Turner (2011) highlighted the potential and scope for aligning objectives on biodiversity conservation and the provision of freshwater at a global scale. There was little overlap – about 3 percent globally – between priority areas identified based on a single objective (i.e. either biodiversity conservation or water supply), suggesting that efforts to conserve biodiversity and provide other ecosystem services would be inefficient unless multiple management objectives are taken into account.

A reconfiguration of priority areas for biodiversity might create synergies with objectives aimed at the provision of freshwater, thereby increasing the area of forest managed for water-related objectives with only a minimal reduction in species representation. Any trade-offs in biodiversity values might be compensated by increased funding for management.

The potential to create win–wins between biodiversity conservation and water services indicates a need to adapt management policies and priorities at the regional and local scales. Locatelli, Imbach and Wunder (2013) analysed spatial correlations between existing policies in Costa Rica (e.g. the network of protected areas and the National Forestry Financing Fund) and the status of ecosystem services. They found that biodiversity and water-related services were positively correlated with all other ecosystem services, including cultural ones (e.g. recreation and scenic beauty). This spatial overlapping can be seen as the result of a combination of policy solutions and biogeography factors: large forested national parks in mountainous areas of Costa Rica are biodiversity hotspots, but they also provide local people with hydrologic benefits such as cloud-water interception, water infiltration and soil protection. Zhang and Pagiola (2011) also found significant overlaps between the areas targeted for watershed protection and biodiversity conservation in Costa Rica, suggesting possible synergies in the implementation of joint payment mechanisms for ecosystem services.

Recreational and cultural co-benefits

Well-managed forests and rivers can provide opportunities for many types of recreation, such as fishing, boating, swimming, wading and hiking. The provision of predictable flows for boating is an important co-benefit of forest management to support water services. Recreational opportunities can generate economic benefits but also trade-offs. For example, white-water rafting generates local economic benefits of about AUD 6 million per year in a region in New South Wales, Australia (Buultjens and Gale, 2006). However, white-water rafting can also have negative impacts on water services, such as through bank trampling and littering, that need to be mitigated (Greffrath and Roux, 2011).

Forests managed for water can provide many cultural services such as aesthetic enjoyment, physical and mental health benefits, and spiritual experiences. The value of splashing in a river, gazing over a riverine landscape and other physical and mental activities associated with forests and water is difficult to overstate. Predictable and natural flow regimes are important in many customary practices, such as ceremonies and religious festivals that include ritual washing or the submersion of deities. Many religions hold particular rivers as sacred – for example a spring near the River Gave de Pau in Lourdes, France; the Jordan River in the Middle East; and the Ganges River in South Asia. In Hinduism, statues of Durga and Ganesh are immersed in rivers in the final stages of the Durga Puja and Ganesh Chaturthi festivals, respectively.

UNDERSTANDING TRADE-OFFS AND SYNERGIES

The hydrologic effects of forests have been the subject of public debate for a long time, and inaccurate assumptions about the forest–water nexus can lead to poor management and policy decisions (Brauman *et al*., 2007; Ellison *et al*., 2017). Land and water management practices play a significant role in how catchments respond to changes in forest cover, and effects can vary at multiple spatial and temporal scales. The analysis of trade-offs and synergies among ecosystem services and management options is therefore key to ensuring effective solutions and optimizing the role of forests in achieving the Sustainable Development Goals, including those related to water security and human health and well-being. The consideration of such trade-offs is of particular policy interest with respect to the following intersecting policy issues: climate (especially the role of carbon sequestration in standing forests and harvested wood products); bioeconomy (with the aim of decarbonizing the economy by substituting fossil-fuel-based materials with bio-based materials); and nature conservation (e.g. forest ecosystem restoration, including for biodiversity and multiple other ecosystem services).

Ellison, Futter and Bishop (2012) conceptualized the forest–water debate into two schools of thought: 1) demand-side (in which trees and forests are viewed mainly as consumers of water and therefore as competitors for other downstream water uses, such as agriculture, energy, industry and households); and 2) supply-side (which emphasizes the beneficial effects of forests on the hydrologic cycle and ultimately on water yield). A systematic review by Filoso *et al.* (2017) showed that most studies investigating forest cover and water yields were conducted at a small scale (i.e. catchments smaller than 10 km^2) and were short-term (i.e. less than ten years); these tended to report negative effects of forest-cover expansion on water yield, although many such studies mentioned the possible influence of temporal and spatial scales on outcomes as a potential limitation. The review by Filoso *et al.* (2017) also found that most existing studies focused on exotic tree species that are usually fast-growing and may not be water-use efficient in local conditions (Trabucco *et al.*, 2008; Cavaleri and Sack, 2010).

Although the demand-side school of thought stresses that upstream forest management can affect water supplies downstream, the supply-side school considers that forests can improve water availability at the regional and global scales by influencing downwind water supplies as a source of precipitation (Ellison, Futter and Bishop, 2012; Ellison *et al.*, 2017). Forests play an important role in regulating fluxes of atmospheric moisture and rainfall patterns through evapotranspiration, originating at least 40 percent of rainfall over land (Jasechko *et al*., 2013). Forest loss and degradation reduce evapotranspiration, with important implications for rainfall thousands of kilometres downwind (Debortoli *et al*., 2016). Widespread tropical deforestation has been predicted to cause up to a 30 percent decrease in rainfall (Lawrence and Vandecar, 2015).

In addition to influencing water availability, forests can affect water quality and temporal variability. For example, Knee and Encalada (2014) analysed water-quality data in five river systems in the Intag region of northwestern Ecuador. Comparing samples from different upstream land uses, they found that streams in protected forests tended to have better water quality than agriculture/pasture, urban development and mining, as well as the lowest concentrations of pollutants. Wang *et al.* (2013) assessed the effects of land-use type on surface water quality in the upper reaches of the Hun River, which provides more than 50 percent of the storage capacity of the largest reservoir for drinking water in northeastern China. They found that upstream land uses had different effects on water physicochemical parameters in different rainfall periods. In particular, forests were mostly associated with good water quality, reducing nutrient loadings through deposition and filtering and thereby decreasing the quantity of sediments carried in surface runoff. Other studies have found strong positive correlations between water-quality parameters and the proportion of upstream forest cover in a watershed, such as Huang *et al.* (2016) in the Three Gorges reservoir catchment in the upper reaches of the Yangtze River basin and de Mello *et al*. (2018) in southeastern Brazil. In examining about 600 watersheds in eastern Canada, Clément *et al.* (2017) found that, even in areas of intensive farming, streams with a forest area covering at least 47 percent of the watershed had higher water quality than those with less forest cover. The same authors also found that woodlands and wetlands located along streams and gullies with an edge density greater than 36 m^2 per ha had a positive impact on water quality; moreover, the shape and location of forested patches were important, with denser, more complex forest patches along streams and gullies more effective in ensuring water quality compared with large, uniform patches.

Spatial trade-offs

Spatial trade-offs in ecosystems may arise – on both the demand and supply side – as a consequence of management choices and biophysical factors (Rodríguez *et al*., 2006; Mouchet *et al*., 2014). Management choices and the use of water upstream impose externalities on those living lower in a catchment; therefore, the most common spatial trade-offs for water-based ecosystem services are between upstream and downstream users (Rodríguez *et al*., 2006).

Trees generally use more water than smaller vegetation because of their greater height and rooting depth. Tree plantations may also require additional nutrients, thus potentially creating trade-offs between carbon sequestration and timber production on the one hand and water yield and soil fertility on the other (Ellison, Futter and Bishop, 2012).

Because natural freshwater ecosystems are dynamic, they require a range of natural variation and disturbances to maintain viability and resilience; they have evolved to the rhythms of hydrologic variability (Baron *et al*., 2002). Water diversions for agricultural or municipal use, for example, and changes in natural nutrient and chemical conditions, can alter freshwater systems and ultimately their capacity to support fish and other aquatic species downstream. Cumming and Peterson (2005) reported that the cumulative effects of multiple nitrogen and phosphorus inputs by small farmers on the Mississippi and Atchafalaya rivers in the southern United States of America created a hypoxic (i.e. low-dissolved-oxygen) zone in the Gulf of Mexico, affecting populations of shrimp and fish species and ultimately local fisheries (Rabalais and Turner, 2019).

Land-use changes such as the conversion of forests to croplands can also affect downstream water-based ecosystem services. For example, Lorsirirat (2007) analysed sediment inflows from upstream areas in the catchment of the Lam Phra Phloeng Reservoir in northeastern Thailand for three periods. He found that the highest sediment volume (2.23 million m^3) occurred between 1970 and 1980, when forest cover declined in the catchment by 70 percent due to agricultural expansion for cash crops, which caused serious erosion (at a rate of 2.77 mm per km^2 per year). Increases in forest area between 1980 and 1990 (+1 percent) and between 1990 and 2000 (+5 percent) helped to significantly reduce both sediment volume (to 0.36 million m^3 per year) and erosion rate (to 0.44 mm per km^2 per year).

The settlement of sediment loads created due to land-use change such as deforestation can result in raised riverbeds, thus causing irregularities in stream dynamics and increasing the downstream flood risk. Conversely, forests (and their appropriate management) can support water management and moderate hydrogeological risks. For example, forest ecosystems in the watersheds of the Yangtze River (Hubei Province, China) regulate water discharge into rivers through canopy interception, litter absorption and soil-water and groundwater conservation. Forests decrease wet-season flows and enhance dry-season ones: Guo, Xiao and Li (2000) estimated that, as a result of waterflow regulation due to the presence of forests, the Gezhouba hydroelectricity plant on the Yangtze River could produce an additional 40 million kilowatt-hours per year, with a 2020 equivalent value of USD 3.2 million per year – which was more than 40 percent of the income generated by forestry in the region (and the value could amount to 220 percent of forestry when the plant is working at full power).

Box 3.13 describes China's huge reforestation efforts and how these have created synergies and trade-offs in water management.

BOX 3.13

Lessons from China's massive forest–water programme

The science that describes forest–water relations has advanced tremendously in the last 50 years, providing much-needed guidance on sustaining water-related benefits through forest management. Nevertheless, there is significant variability in the influences of forest management and many trade-offs and synergies among forest ecosystem services (Xiao *et al.*, 2013).

This case study analyses "managing forests for water" programmes in arid northwestern China as a case study of how hydrologic science and practice has progressed; it shows how forest management for water efforts can benefit local and downstream communities and presents lessons learned. Ultimately, ecohydrologic principles[11] must be adhered to and trade-offs among ecosystem services recognized to successfully implement forest–water programmes in diverse natural and socio-economic settings.

Sustainability of forest-based ecological restoration efforts

China's forest resources were depleted in the Second World War and later in the 1950s and 1960s when food production and industrialization became the country's highest priorities. Reforestation campaigns in China started in the 1970s with the Three-North Forest Shelterbelt Development Programme (hereafter called the Three-North Programme), the aim of which was to arrest the rising dust storms that threatened Beijing and other northern cities caused

Continued ...

[11] Ecohydrologic principles are the hydrological and ecological drivers that interact to control the structure and function of a forest. For example, sun light drives leaf growth, which controls evapotranspiration, which controls the hydrology (Dale *et al.*, 2000).

by severe soil erosion. Satellite imagery shows that the majority of China, especially in the arid Loess Plateau region, is "greening up" (Zhang *et al.*, 2017), thanks in part to reforestation efforts in the last two decades and also to climatic warming and an increase in precipitation (Xie *et al.*, 2015). Since the 1990s, China has invested USD 378.5 billion (in 2015 United States dollars) in land restoration programmes that covered 623.9 million ha of land and involved over 500 million people (Bryan *et al.*, 2018).

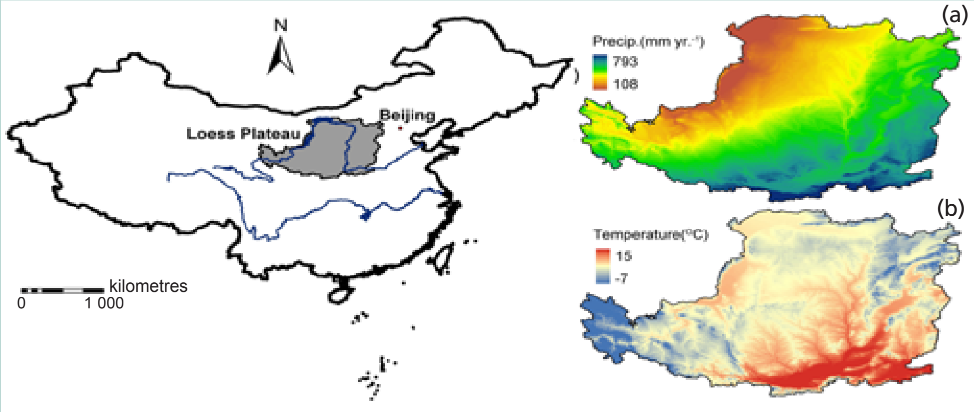
Today, China has the world's largest area of forest plantations – approximately 84 million ha, which is more than one-quarter of the world total of 293 million ha (FAO, 2020a). The total forest cover in China grew from about 11 percent of the land area in the 1980s to 23 percent in 2020 (FAO, 2020a). China has ambitious plans to further increase forest cover to 26 percent by 2035 and to 42 percent by 2050 as part of a policy to create an "ecological civilization". A large part of China's new forest estate is on the Loess Plateau (Figure 3.6), an area the size of France, where forest cover doubled from 5 million ha in 2001 to 10 million ha in 2016. The Three-North Programme spans about 400 million ha of arid and semiarid lands in the country's north, which is more than 42 percent of China's total land area (Xie *et al*., 2015). Under the programme, large areas have been planted with exotic trees and shrubs tolerant of arid conditions, including *Robinia pseudoacacia*, *Caragana intermedia*, *Amorpha fruticosa*, *Pinus tabuliformis*, *Populus davidiana*, *Ulmus pumila* and *Hippophae rhamnoides* (Cao, 2008).

There was a major change in the national policy on forest and grassland management in the late 1990s, when China suffered from flood disasters in the Yangtze and Yellow river basins, affecting more than 240 million people. The Natural Forest Conservation Programme, initiated in 1998, sought to halt logging and deforestation to protect natural forests for ecological and carbon benefits, and it encouraged afforestation by providing incentives for forest enterprises. Its target was to reduce timber harvests in natural forests from 32 million m^3 in 1997 to 12 million m^3 in 2003 and to afforest 31 million ha by 2010 through mountain closure (i.e. the prohibition of human activities such as woodfuel collection and grazing to allow regrowth), aerial seeding and artificial planting (Liu *et al*., 2008).

The Natural Forest Conservation Programme was followed by a series of ecological restoration programmes in recognition of serious environmental and ecological issues arising during an economic boom in the 2000s. For example, the Grain-for-Green Programme, launched in 1999, has been described as the developing world's largest land-retirement programme. It uses a payment scheme for ecosystem services to directly engage millions of rural households as core agents in programme implementation. In the period 1999–2008, the central government made a direct investment of RMB 192 billion (approximately USD 28.8 billion) in the Grain-for-Green Programme; under it, 120 million farmers converted 9.27 million ha of sloping croplands to forests (Lü *et al*., 2012).

Ecosystem service assessment studies suggest that these (and other) decades-long efforts in China have brought enormous benefits in improved local environments and people's well-being, including through erosion control, improved water quality, carbon sequestration and local economic development (Liu *et al*., 2008; Lü *et al*., 2012; Bryan *et al*., 2018). In a review of China's investment strategies for land-system sustainability, Bryan *et al.* (2018) found that the country's large-scale afforestation programmes had been successful, setting an example for the rest of the world in addressing the challenge of land restoration. Bryan *et al.* expressed caution, however, about negative unintended local (e.g. soil desiccation), watershed-scale and regional (e.g. river-flow reduction) water resource outcomes, as found by Sun *et al.* (2006) and Cao (2008). The hydrologic impacts of the reforestation may be substantial for the Yellow River (Asia's third-largest river), which has showed a declining trend in river flow and a 60 percent drop in sediment loading to the ocean since the 1980s, due mostly to vegetation recovery (Liang *et al*., 2015; Wang *et al*., 2016; Schwarzel *et al*., 2020). Scientists and policymakers in China are increasing concerned about water security and forest management (Feng *et al*., 2016; Cao, 2008; Zhang and Schwärzel, 2017).

FIGURE 3.6
Location of the Loess Plateau and average climate conditions: (a) precipitation and (b) temperature



Source: Lü *et al.* (2012).

FIGURE 3.7
Pine plantations in the Loess Plateau have reduced soil moisture and thus have relatively low functionality in protecting surface soils and biodiversity

Source: Yang *et al.* (2012).

Many lessons have been learned from the afforestation programmes implemented on the Loess Plateau, including the following:

- Water yields decrease in response to large-scale afforestation, soil conservation measures (e.g. check dams) and climate change (Sun *et al.*, 2006; Mu *et al.*, 2007; Zhang *et al.*, 2008).
- There are trade-offs among ecosystem services in planted forests (Lü *et al.*, 2012).
- Water yield, evapotranspiration, ecosystem productivity, carbon sequestration and sediment loading are closely coupled in anthropogenic–biological systems.
- The revegetation of China’s Loess Plateau is approaching sustainable water-resource limits (Feng *et al.*, 2016; Wang *et al.*, 2016).

Continued ...

- Reforestation at high tree densities using non-native pine species can cause soil desiccation (Yang *et al.*, 2012; Liu, Kuchma and Krutovsky, 2018; Liang *et al.*, 2018) and low light penetration to the forest floor, consequently reducing forest productivity and biodiversity (Figure 3.7).
- Planting trees in areas with limited precipitation (e.g. less than 400 mm per year) can damage soil physical properties, reduce infiltration capacity and promote overland flows and erosion (Chen *et al.*, 2010); when planting trees it is important to consider species and planting densities.

Historic vegetation patterns are a good guide for determining suitable vegetation for reforestation (or re-grassing) efforts. The selection of species for revegetation should be location-specific and not a "one size fits all" approach (Cao *et al.*, 2011). Many degraded ecosystems have remarkable ability to recover through natural processes. The human dimension (livelihoods and policy) must be factored into reforestation programmes to meet the multiple needs of nature and people (Cao *et al.*, 2009).

Temporal trade-offs and synergies

Many ecosystem processes, such as soil creation and changes in soil fertility and groundwater, occur at such slow rates that a long time is needed before significant effects can be perceived (Rodríguez *et al*., 2006). In a global analysis of 504 annual catchment observations, Jackson *et al.* (2005) found that afforestation tended to decrease streamflow within a few years of planting, especially in drier regions. Trade-offs between timber, carbon and water have also been identified in other studies. Cademus *et al.* (2014) found that water yields decreased in *Pinus elliottii* forests in Florida, United States of America, as biomass increased, but this trade-off varied over time and space depending on stand age, silvicultural treatment and site quality.

Chisholm (2010) investigated the economic viability of a possible expansion of *Pinus radiata* plantations in the Swartboskloof catchment in the fynbos biome in South Africa (one of the world's 25 biodiversity hotspots). Considering a range of economic scenarios, the marginal viability of afforestation coincided with a roughly linear trade-off between the values of carbon and water. For current economic values of water, carbon and timber and a mean fire interval of 32 years, afforestation was found to be economically unviable compared with the conservation of the biome. Given current timber prices, afforestation would become viable only if the price of a tonne of carbon dioxide was roughly 400 times the value of a cubic metre of water (Chisholm, 2010).

Ovando, Beguería and Campos (2019) analysed alternative management solutions for native pine and oak forests in Andalusia, southern Spain, with reference to long-term (to 2100) impacts on carbon sequestration and water provisioning services (surface discharge and groundwater recharge). They found that trade-offs between carbon sequestration and water supply were more likely than synergies in Mediterranean forests in the short to medium term (up to 2050), but synergies would arise in the longer term (2060–2100).

Although the dominant paradigm indicates trade-offs between forest cover/carbon sequestration and water yield, particularly in terms of groundwater recharge, Ellison *et al.* (2017) identified several caveats and biases and advocated more specific studies. In tropical areas in particular, the loss of forest cover can promote soil degradation and ultimately reduce soil infiltration, water-retention capacity and water quality, with major implications for rural households. This could be crucial, especially in low-income countries where the costs of installing and maintaining water-treatment systems in small communities might be unaffordable. Mapulanga and Naito (2019) analysed the effect of deforestation on household access to clean drinking water in northern

Malawi. This region has a historically high deforestation rate compared with the rest of the country attributed to its low population density and consequent difficulties in monitoring and regulating logging and woodfuel collection. Community members earn income from the sale of charcoal produced in local forests, but this activity has reduced the capacity of forests to ensure water quality and ultimately access to clean water. Deforestation that increases sediment loads reduces the feasibility of piping water because piped water systems require high-quality water sources; in this situation, the use of unprotected wells, rivers and ponds as sources of drinking water is likely to increase, making people more exposed to low-quality, unsafe water (World Health Organization, 2017). Mapulanga and Naito (2019) found that every 1 percent increase in deforestation implied a 0.93 percent decrease in access to clean drinking water. Based on this ratio, it was estimated that, in 2000–2010, deforestation in northern Malawi (a 14 percent loss of forest cover) had the same magnitude of impact on access to clean drinking water as would have been caused by a 9 percent decrease in rainfall.

Zongo *et al.* (2017) examined the impacts of forest loss and degradation on temporary ponds within and outside protected forest areas in eastern Burkina Faso. These ponds provide water for wild and domestic animals as well as for people in nearby villages. The authors found that water quality – in terms of both chemical and physical characteristics – in the temporary ponds was higher in protected areas than in unprotected ones, the latter being exposed to a greater risk of woodfuel collection and conversion to agriculture or grazing lands. Such uses ultimately cause higher turbidity in the ponds because rainwater runoff has a higher content of detritus and soil. Similarly, eutrophication was observed in ponds outside reserves due to higher concentrations of organic matter. Excess eutrophication can lead to the production of secondary metabolites that are highly toxic to animals and can pose health hazards to people.

Temporal trade-offs in ecosystem services can be identified in mangrove forests, which supply a wide range of such services, both locally and globally (Barbier, 2007). Many coastal communities in developing countries rely on the extraction of woodfuel and timber from mangrove forests for their subsistence and livelihoods. More than one-quarter of the world's mangrove habitats are overexploited and degraded (Valiela, Bowen and York, 2001). The unsustainable harvesting of mangrove wood not only affects ecosystem integrity and biodiversity, it can also have negative impacts on the nursery habitats of fish and shrimp species that are vital for the subsistence and livelihoods of coastal communities. Approximately 80 percent of the worldwide fish catch is estimated to depend directly or indirectly on mangroves (Ellison, 2008).

McNally, Uchida and Gold (2011) investigated the trade-offs among the provisioning services (woodfuel versus fishing) provided to local communities by mangrove forests in the Saadani National Park, United Republic of Tanzania. They found a trade-off between the short-run benefits of cutting mangrove forests for woodfuel and the potential long-run benefits of mangrove conservation. The extent of the trade-off differed depending on household wealth: mangrove protection would cause an immediate loss of income due to the curtailment of woodfuel collection, with richer households particularly affected. All wealth classes would likely benefit, however, from gains in the long-term sustainability of shrimping and fishing arising from mangrove protection. McNally, Uchida and Gold (2011) found that, on average, a 10 percent increase in mangrove cover in the Saadani National Park could increase shrimping income approximately twofold. Thus, the creation of a protected area would support a shift from uncontrolled mangrove cutting to mangrove conservation, provided there are gains in income in local villages as a result of the conservation of nursery habitats and biodiversity.

FOREST FIRES AND WATER

Natural disturbances can disrupt even the best-laid plans, and wildfires are potentially the most destructive and impactful of all such disturbances. Understanding how fire affects forests and can alter forest plans is crucial for the long-term health of forests and water resources. Forests are dynamic systems shaped by disturbances (Oliver and Larson, 1996). The loss of forests due to catastrophic fire is a major risk in catchments and to the water they produce. Forest management can help mitigate the risk by making forests more resilient to fire. Reducing forest density through thinning can both provide wood yields and improve tree health by reducing vulnerability to pests and diseases. Properly done, thinning can also reduce fuel volume and spatial arrangement to decrease the risk of large, hot fires capable of deforesting entire catchments.

The impacts of fire on water yield and quality are highly variable and complex (Neary and Leonard, 2015). Wildfire can have profound hydrologic impacts – it is the forest disturbance with the greatest potential to change watershed condition (DeBano, Neary and Ffolliott, 1998). Watershed condition, or the ability of a catchment system to receive and process precipitation without ecosystem degradation, is a good predictor of the potential impacts of fire on water supplies and other resources (e.g. roads, recreation facilities and riparian vegetation).

Forest fire management

Wildfires and prescribed fires can have a wide range of impacts on forested watersheds depending on interactions between fire severity and scale, slope, hydrologic condition, soil infiltration rates and postfire rainfall (Neary, 2019); these factors determine the degree of impact of fire and consequently the need for special postfire management. Fire can be a useful management tool, and the judicious use of fire should not require specific preparatory measures. Repeated uncontrolled forest fires, however, can lead to the serious deterioration of water services.

A low-severity prescribed fire in a small landscape unit with minimal fuel loading, slopes less than 10 percent and no water repellency is unlikely to reduce watershed condition and functions in all but heavy rainfall. On the other hand, a high-severity wildfire in a large area of heavy fuels with slopes greater than 100 percent and significant water repellency may result in serious deterioration with even moderate rainfall (Hallema *et al*., 2018). Soil management is unlikely to be needed in the former case and would be virtually impossible in the latter.

Fire severity. Fire severity – the commonly accepted term for describing the ecological effects of a specific fire – is a crucial concept for understanding the effects of forest fire on watershed conditions (Neary and Leonard, 2015); it describes the magnitude of the disturbance and therefore reflects the degree of change in ecosystem components. Fire severity integrates both the aboveground heat pulse and the heat pulse transferred downward into the soil (Borchers and Perry, 1990). It is dependent on the nature of the fuels available for burning, fire duration, climate, and the combustion characteristics that occur when vegetation and forest-floor fuels are ignited (Simard, 1991). Soils are affected by both the combustion of surface organic horizons (Byram, 1959) and the heat pulse into the mineral soil (DeBano, Neary and Ffolliott, 1998).

The effects and severity of wildland fire are strongly influenced by fuel loads – the total dry weight of fuel per unit surface area – and climate (DeBano, Neary and Ffolliott, 1998). Both live and dead vegetation contribute the biomass material comprising the fuel consumed in combustion; fuel load , which is usually measured as the mass per unit area, therefore, is a good measure of the energy that could be liberated by fire (Brown and Smith, 2000). Natural fuel loadings can vary from 0.5 tonnes per ha in light fuels to more than 400 tonnes per ha in heavy fuels (Neary and Leonard, 2015).

Brown and Smith (2000) described four types of severity-linked fire regimes that affect vegetation and watersheds: 1) understorey fire; 2) mixed-severity fire; 3) stand replacement fire; and 4) no fire. Understorey fires are generally non-lethal to the dominant vegetation and do not adversely affect watershed conditions. Such fires are usually low-severity ground fires typified by prescribed fires. Mixed-severity fires produce selective mortality in the dominant vegetation, depending on the tree species and the matrix of severities. Stand-replacing fires kill the aboveground parts of the dominant vegetation and usually have adverse effects on soils and watersheds. Most wildfires are a mix of all three fire regimes and may also contain areas classified as non-fire regimes.

Six fuel-related factors affect the intensity of fire and the severity of its impacts on vegetation, soils, watersheds and other ecosystem components: 1) temperature; 2) moisture; 3) position; 4) loading; 5) continuity; and 6) compaction (Neary, Ryan and DeBano, 2005). The temperature needed for fuel ignition ranges between 204 °C and 371 °C (DeBano, Neary and Ffolliott, 1998). Fuel moisture is determined by climate and weather, plant species and vegetation age. Wet weather increases fuel moisture, and vegetation age affects plant moisture (older plants are drier than younger ones). The moisture content of live fuels is also dependent on season and the presence of soil moisture and groundwater. The moisture content of dead fuels is a function of atmospheric humidity, air and biomass temperature, and solar radiation. The position of fuels relative to the ground (e.g. subsurface, surface or aerial) also affects the ease of ignition.

Subsurface fuels primarily comprise live and dead roots and organic layers, which are the last to ignite. Surface fuels consist of vegetation litter, grasses and other herbaceous plants. Aerial fuels are composed of shrub and tree biomass. Fuel continuity is the horizontal and vertical spacing of biomass (and is described as either continuous or patchy). The rate of combustion and the direction of fire movement are more predictable with continuous fuels. The ignition of patchy fuels is more dependent on spatial arrangement, and ignition and the direction of fire movement are therefore sporadic and uneven. Lastly, the temperature at which a fuel is susceptible to ignition decreases with increasing fuel compaction (DeBano, Neary and Ffolliott, 1998). Low atmospheric relative humidity contributes to vegetation desiccation. Low antecedent rainfall, low relative humidity, high air temperatures and high winds constitute a recipe for high-severity wildfire (Bradstock, 2010).

Litter or organic-matter fires burn at low speeds and intensities due to air-supply limitations; grass fires, on the other hand, burn at a high rate of spread, high intensity and low severity. Crown fires (i.e. fires in which the crowns of trees burn) burn at a high rate of spread, high intensity and high severity.

A low-severity fire may be useful in restoring and maintaining various ecological attributes that are generally viewed as positive; this is the case, for example, in the fire-adapted longleaf pine (*Pinus palustris*) and ponderosa pine (*P. ponderosa*) ecosystems. High-severity fires, on the other hand, have significant negative ecological – biological, chemical and physical – impacts, with the potential to alter the functioning of the soil and hydrologic systems for decades, centuries and even millennia.

Fire trends. Liu, Stanturf and Goodrick (2010) investigated trends in global wildfire potential under climate change and predicted significant increases in North America, South America, Central Asia, southern Europe, southern Africa and Australia. Relative changes are expected to be highest in southern Europe and smallest in Australia (which already has a high incidence of forest fire). The increased fire potential predicted by Liu, Stanturf and Goodrick (2010) was due mainly to projected warming in North and South America and Australia and a combination of warming and drying in the other regions. Some regions were predicted to experience moderate fire potential year-round, and the window of high fire potential will last longer each year. The analysis by Liu, Stanturf and Goodrick (2010) suggests dramatic increases in wildfire potential that will require increased future management efforts for disaster prevention and recovery.

In a similar study, Flannigan, Stocks and Wotton (2000) investigated the potential impacts of climate change on forest fire and the structure of North American forests. They found that seasonal severity ratings could increase by 10–50 percent over most of North America (although some regions might experience little change, or decreases) by the middle of the present century.

The implications for forest fire management of these and other studies are substantial. The risk posed by wildfire to water resources will increase markedly over large areas of temperate forest under climate change and, by necessity, will require the close attention of land and water management decision-makers.

The impacts of wildfire on water

Erosion. After the destruction of vegetation, erosion is the most visible and dramatic impact of wildfire. Increased stormflows after wildfire due to the loss of vegetation will also increase the rate of erosion. On the other hand, rehabilitation work can decrease postfire erosion to varying degrees, depending on the nature of the work and the timing and intensity of rainfall (Robichaud, Beyers and Neary, 2000). Fire management activities such as wildfire suppression, prescribed fire, the construction of firebreaks and postfire watershed rehabilitation can also affect erosion processes in forest ecosystems.

Natural erosion rates in undisturbed forests range from less than 0.01 tonnes per ha per year to 7 tonnes per ha per year (DeBano, Neary and Ffolliott, 1998); the upper limit of geologic erosion in highly erodible and mismanaged soils is 560 tonnes per ha per year. Differences in natural erosion rates arise due to site factors such as soil and geologic erosivity, rates of geologic uplift, tectonic activity, slope, rainfall amount and intensity, vegetation density and percent cover, and fire frequency. Landscape-disturbing activities such as mechanical site preparation (potentially causing an erosion rate of 15 tonnes per ha per year; Neary and Hornbeck, 1994), agriculture (560 tonnes per ha per year; Larson, Pierce and Dowdy, 1983) and road construction (140 tonnes per ha per year; Swift, 1984) can increase sediment loss in catchments.

Fire-related sediment yields vary considerably depending on fire frequency, climate, vegetation and geomorphic factors such as topography, geology and soils (DeBano, Neary and Ffolliott, 1998). In some regions, more than 60 percent of total long-term landscape sediment production is fire-related. Erosion rates vary from less than 0.1 tonnes per ha per year for low-severity wildfire to more than 1 500 tonnes per ha per year for high-severity wildfires on steep slopes (Neary *et al*., 2012). Sediment yields one year after a prescribed burn or wildfire range from very low in flat terrain and the absence of major rainfall events to extreme in steep terrain affected by high-intensity rainfall. Erosion typically declines in a burnt area over subsequent years as the site stabilizes (e.g. ground vegetation and a litter layer is re-established), but the rate of recovery varies depending on fire severity and vegetation recovery.

Water quality. Fire can have a major effect on catchment hydrology, geomorphology and water quality in fire-prone regions (Shakesby and Doerr, 2006). Turbidity can increase after fire due to the suspension of ash and silt-to-clay-sized soil particles in flood streamflow; turbidity is often the most visible water-quality effect of fire (DeBano, Neary and Ffolliott, 1998). Less is known about turbidity than sedimentation generally because it is difficult to measure, highly transient and extremely variable. Extra-coarse sediments such as sand, gravel and boulders eroded in burnt areas (due to higher peak flows in storms) can also adversely affect aquatic habitats, recreation areas and reservoirs. Postfire sediment yields vary widely depending on fire severity, topography, fuel type and climate. The highest soil erosion rates are usually associated with intense rainfall on steep terrain (Moody and Martin, 2001; Neary, Ryan and DeBano, 2005).

The nitrogen forms most commonly studied as indicators of fire disturbance are nitrate, ammonia and organic nitrogen, but hydrologists and watershed managers tend to focus on nitrate because it is highly mobile. The potential for an increase in nitrate in streamflow after fire is due mainly to accelerated mineralization and nitrification (DeBano, Neary and Ffolliott, 1998) and reduced plant demand. This results from the conversion of organic nitrogen to available forms, mineralization, and mobilization by microbial biomass through the fertilizing effect of ash nutrients and improved microclimates. These postfire effects are short-lived, however (usually only one year or so).

Water quantity. Annual streamflow discharges in catchments burnt by wildfire have been highly variable in Australia, Europe and North America (DeBano, Neary and Ffolliott, 1998). Helvey (1980) found substantial increases in discharge in a watershed in which wildfire killed nearly 100 percent of vegetation in a mixed-conifer forest. Differences between the measured (burnt) and predicted (unburnt) streamflow discharge varied from 107 mm in a dry year to about 477 mm in a wet year.

Annual streamflow discharge from watersheds in fire-prone chaparral shrublands in the southwestern United States of America increases (by varying magnitudes) at least temporarily as a result of high-intensity wildfire (Baker *et al*., 1998). The combined effects of loss of vegetative cover, decreased litter accumulation and the formation of water-repellent soils following fire are the presumed reasons for such streamflow increases (Hallema *et al*., 2018).

Average annual streamflow discharge increased by about 10 percent (to 120 mm) in a forested watershed in the Cape region of South Africa following a wildfire that consumed most of the indigenous fynbos (sclerophyllous) vegetation (Scott, 1993), resulting in more stormflow on a severely burnt watershed compared with a watershed that was only moderately burnt.

Lavabre, Gaweda and Froehlich (1993) found that streamflow discharge increased by 30 percent to nearly 60 mm in the first year after a wildfire in a watershed in southern France, where the pre-fire vegetation was primarily a mix of maquis, cork oak and chestnut. They attributed the increase to a reduction in evapotranspiration due to a corresponding decrease in basal area in woody vegetation caused by the fire.

In general, changes in annual watershed yields after wildfire, as measured by numerous wildfire investigations, are the result of changes in vegetation characteristics, soil conditions and climate. Reductions in the density of woody vegetation and basal area affect postfire evapotranspiration (DeBano, Neary and Ffolliott, 1998). The loss of organic-matter soil horizons and the development of water repellency lead to higher rates of runoff and erosion. Meanwhile, land surfaces blackened by fire absorb more heat and lead to increased thunderstorm activity, and therefore precipitation rates and intensities are frequently higher after wildfires (Neary, 2019).

Convection, rainfall intensity, and precipitation amounts increase dramatically under the right meteorological conditions. Even historical normal precipitation rates can produce excessive runoff due to the combined fire effect on vegetation, litter, and soil conditions (DeBano, Neary and Ffolliott, 1998). Risks for elevated precipitation amounts and subsequent flooding are greatest within the first year after wildfire but can continue for 10 to 20 years due to fire modification of the pre-fire environment.

The impacts of prescribed fire on water

Erosion. Soil erosion following prescribed fire ranges from less than 0.1 tonnes per ha per year to 15 tonnes per ha per year. Slope, severity and climate are the major factors in determining the amount of sediment yielded during rainfall following prescribed fire.

Water quality. Wright, Churchill and Stevens (1976) demonstrated the effect of slope on water quality after prescribed fire in a study in juniper stands in Texas, United States of America. The annual sediment loss due to prescribed fire ranged from about 0.029 tonnes per ha per year on flat ground (i.e. 0 percent slope) to 8.443 tonnes per ha per year on slopes of 43–54 percent (the sediment loss on comparable terrain was 0.013 and 0.025 tonnes per ha per year, respectively, in unburnt paired catchments).

Water quantity. Streamflow responses are smaller in magnitude for prescribed fire than for wildfire. It is generally not the purpose of prescribed burning to completely burn forest litter and other decomposed organic matter on the soil surface (DeBano, Neary and Ffolliott, 1998). The retention of at least some of this litter and organic matter reduces the likelihood of drastic alterations in streamflow discharges that are common after severe wildfires.

A burn prescribed to reduce accumulated fuel loads in a 180-ha watershed in the Cape region of South Africa resulted in a 15 percent increase (to 80 mm) in average annual streamflow discharge (Scott, 1993). Most of the fynbos shrubs in the watershed were undamaged by the prescribed fire. The immediate effectiveness of the fire in reducing fuel loads was less than anticipated due to the unseasonably high rainfall at the time of burning.

A prescribed fire in a grassland community in Texas, United States of America, resulted in a large (1 150 percent) increase in streamflow discharge compared with an unburnt watershed in the first year after burning (Wright, Churchill and Stevens, 1982). The increased postfire streamflow discharge was short-lived, however, with streamflows returning to pre-fire levels shortly after the fire.

The burning of logging residues (slash) in timber harvesting operations, of competing vegetation to prepare a site for planting, and of forests and woodlands in the process of clearing land for agricultural production are common practices in many parts of the world. Depending on their intensity and extent, fires for these purposes may cause changes in streamflow discharge. In analysing the responses of streamflow discharge to prescribed fire, however, it is difficult to isolate the effects of treatments from the accompanying hydrologic impacts of timber harvesting, site preparation and the clearing of forest vegetation.

Fire management and water considerations

Planning. In planning prescribed-fire treatments, forest managers should:

- consider prescription elements and ecosystem objectives at the appropriate catchment scale in determining the optimum and maximum burn unit size, total burn area, burn intensity, disturbance thresholds for local downstream water resources, the area or length of water resources to be affected, and contingency strategies;
- consider the extent and severity of fire disturbance, and the recovery afterwards, that a watershed has previously experienced to evaluate cumulative effects and re-entry intervals;
- identify those environmental conditions favourable for achieving the desired condition or treatment objectives of the site while minimizing detrimental mechanical and heat disturbances to soils and water resources;
- develop burn objectives that avoid or minimize the creation of water-repellent soil conditions to the extent practicable considering fuel loads, fuel and soil moisture levels, fire residence times and potential burn severity;
- use low-severity prescribed burning when fire is the only practicable means for achieving project objectives on steep slopes and highly erodible soils;
- set targets for desired levels of ground cover after burning based on slope, soil type and risk of soil and hillslope movement;
- where practicable, plan burn areas using natural or in-place barriers – such as roads, canals, utility rights-of-way, barren or low-fuel-hazard areas, streams, lakes, and wetland features – to reduce or limit fire spread and minimize the need for firebreak construction;
- identify the type, width and location of firebreaks in the prescribed fire plan;
- use locations for ignition and control that minimize potential effects on soil, water

quality and riparian zones; and
- use prescribed fire in riparian zones only when this will help achieve long-term ecological conditions and management objectives for such zones.

Best management practices. Prescribed fires should be conducted using available guidelines on best management practices to achieve the burn objectives outlined in the planning process (Neary, 2014). Safety zones, access routes and staging areas should be identified and located near project sites but outside riparian zones, wetlands and areas with sensitive soils. Staging areas (i.e. areas designated for the gathering of people, vehicles and equipment in preparation for a fire) should be kept as small as possible while allowing safe and efficient operations. Ignition-device fuels should be stored away from surface water bodies and wetlands. Suitable measures are needed to minimize and control concentrated waterflows and sediments from staging areas. Staging areas should be restored and stabilized after use. Prescribed fires should be managed to minimize the residence time of fire on soils while meeting burn objectives.

North America. North America has the world's most extensive literature on wildland fires and water. Summaries of case studies are available in DeBano, Neary and Ffolliott (1998), Neary, Ryan and DeBano (2005), Neary and Leonard (2015) and Hallema *et al.* (2017).

South America. Wildfire and prescribed fire have become significant issues in South America for the maintenance of water resources and other ecological values in the context of climate change, land clearance and intensive plantation forestry (Sanford *et al.*, 1985; Di Bella *et al.*, 2006; Úbeda and Sarricolea, 2016; Liu, Stanturf and Goodrick, 2010).

Europe. There has been an increase in the frequency of wildfires in Europe in the past several decades – particularly in the Mediterranean region (Liu, Stanturf and Goodrick, 2010) but, in recent years, even in boreal forests. These trends pose risks to water supplies and natural hydrologic regimes (Smith *et al.*, 2011; Robinne *et al.*, 2018). Drought is an important factor in the increase in fire frequency, but human activities are also implicated (Turco *et al.*, 2017). Wildfire increases have added complexity to fire management in Europe – including the need for additional suppression resources (Tedim, Xanthopoulos and Leone, 2015) – as well as to forest–water management.

Australia. Wildfires burn large areas of forest in Australia each year, including potentially in catchments important for the supply of potable water, such as for the cities of Adelaide, Brisbane, Canberra, Melbourne and Sydney (Smith *et al.*, 2011). Australia suffered its worst fire season in history in 2019–20, with an estimated 10.2 million ha burnt, including 8.19 million ha of native forest (the remainder comprising agricultural croplands and grasslands, forest plantations, other forest, periurban lands, and native grasslands, heath and shrublands) (Davey and Sarre, 2020). Wildfires in 2003, 2009 and 2020 threatened or disrupted water supplies in several major metropolitan areas.

OTHER DISTURBANCES WITH IMPACTS ON WATER

The impacts of climate change are expected to increase throughout the twenty-first century (IPCC, 2014a). Increasing climate variability is likely to mean increases in flooding, heatwaves and drought, with major implications for water management. Floodwaters are often laden with sediment that can deposit in watercourses and thus increase the risk of future flooding and disruptions to the hydrologic cycle (Bathurst *et al.*, 2017). Heatwaves increase the rate of forest evapotranspiration (Guerrieri *et al.*, 2016); increases in tree water demand lead to decreases in soil moisture and streamflow, even if precipitation rates do not change. Drought directly affects forest water yield by decreasing precipitation input to soils (McNulty, Boggs and Sun, 2014). Vegetation has first access to soil water through its root systems; water will flow in forest streams only after plant water demand has been met. Therefore, trees may experience limited stress in drought conditions but streams may run dry (Vose *et al.*, 2016).

Invasive species can have impacts on forest water yield. For example, invasive insects may cause widespread tree defoliation and mortality, reducing plant water demand and increasing stream water yield (Tamai *et al*., 2020). Conversely, invasive plant species can increase the total forest leaf area, which will elevate plant water demand and decrease stream water yield (Dye and Poulter, 1995). Fire can be a benefit or a bane to forest–water management.

Understanding how biotic and abiotic stressors interact with forest management is vital for forest–water sustainability. Such stressors can result in the decline or death of forest trees, with impacts on the hydrologic cycle and the potential to increase soil erosion and landslides that affect water quality (IPCC, 2014a). Carbon dioxide is the primary contributor to climate change, but other pollutants can also affect forest and hydrologic processes. Emissions of nitrogen and sulphur compounds from the burning of fossil fuels have decreased over the past 30 years in many parts of the Northern Hemisphere but are increasing in East Asia (Aas *et al*., 2019). Most forests are nitrogen-deficient, and deposited aerosol nitrogen acts as a fertilizer, increasing leaf area, forest growth and water use (Carter *et al*., 2017). In some forests, however, the quantity of input nitrogen is excessive to the point of toxicity, creating conditions of nitrogen saturation (Aber *et al*., 1998) and causing declines in forest health and subsequent increases in water yield and decreases in water quality (in the form of excess nitrite released into streams) (McNulty *et al*., 2017). Excessive atmospheric sulphur compounds can acidify forest soils, resulting in soil aluminium toxicity and leading to forest decline, increased water yields and reduced water quality (Sullivan *et al*., 2013).

These stressors affect water quantity, quality and timing in various ways but all involve changes in forest cover (and associated changes in forest root mass). Generally, as forest waterflow increases, the quality of water decreases because the percentage of overland versus belowground flow increases. Soils act as filters that purify water. Conversely, overland waterflow can dislodge soil particles and transport them into streams and thus increase stream turbidity while causing soil erosion. The opposite is true as forest cover expands. Root mass increases with increasing forest leaf area, which in turn better secures the soil. Also, precipitation hits the forest canopy before proceeding to the soil surface. The amount of energy contained in a raindrop that has fallen from 30 m in a forest canopy is much less than the energy from precipitation that has fallen several hundred (or thousand) metres from a cloud onto exposed soil.