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Foreword

Forested watersheds provide 75 percent of our accessible freshwater supply and are therefore integral to our water security. Landscape transformations due to growing populations, increasing urban sprawl and shifts in land use and climate ultimately affect hydrology, including the quantity, quality and timing of water. Tree loss and watershed degradation increase the risk of erosion, forest fires and water stress. Yet only 12 percent of the world's forests are managed with water as a primary objective.

Managing forests to provide healthy water functions does not need new management tools. Rather, it requires the application of existing tools through a lens that considers ecosystems, the locations of those ecosystems in the landscape, other management objectives, and scale.

Numerous resources provide information on forest–water relationships. The present publication, *A Guide to Forest–Water Management*, however, is the first comprehensive global publication on the monitoring, management and valuation of forest–water interactions. It was developed to stimulate discussions on strategic forest management and governance for water and to provide general guidance on forest–water monitoring, management and valuation at multiple scales.

Because of the importance of context in forest–water relationships, this publication does not provide comprehensive and detailed guidance for all situations. It does, however, examine certain specific forest ecosystem types as examples to illustrate how sustainable forest management can support hydrologic functions and services at different scales, from local to landscape.

A Guide to Forest–Water Management is the product of collaboration among numerous experts worldwide, supported by FAO, the European Commission, the United States Forest Service, the International Union of Forest Research Organizations' Task Force for Forests and Water, and the European Commission Joint Research Centre.

Ensuring the functionality of landscapes and the delivery of ecosystem services requires effective management and monitoring that focuses on water. Despite uncertainty around integrated forest–water management, it is imperative that water receives much more attention in forest management as the world faces the consequences of climate change and other pressures. We hope and expect that the guidance provided here will encourage stakeholders to prioritize water in forest management and governance.

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Abbreviations and acronyms

AUD	Australian dollar(s)
BTT	Blue Targeting Tool
DEM	digital elevation model
EUR	euro(s)
FAO	Food and Agriculture Organization of the United Nations
FL-WES	Forest and Landscape Water Ecosystem Services
FRA	Global Forest Resources Assessment
GIS	geographic information system
ha	hectare(s)
HU	hydrologic unit [United States of America]
km	kilometre(s)
kWh	kilowatt-hour(s)
m	metre(s)
mm	millimetre(s)
MXN	Mexican peso(s)
NASA	National Aeronautics and Space Administration
PES	payments for ecosystem services
PWS	payments for watershed services
RFA	recorded forest area [India]
RMB	Chinese renminbi
SEPAL	System for Earth Observation Data Access, Processing and Analysis for Land Monitoring
TMCF	tropical montane cloud forest
TOC	total organic carbon
USD	United States dollar(s)
VDT	variable-density thinning
VND	Vietnamese dollar(s)
VHR	very high resolution
WUE	water-use efficiency
WWF	World Wide Fund for Nature

Executive summary

Many people worldwide lack adequate access to clean water to meet basic needs, and many important economic activities, such as energy production and agriculture, also require water. Climate change is likely to aggravate water stress. As temperatures rise, ecosystems and the human, plant and animal communities that depend on them will need more water to maintain their health and to thrive.

Forests and trees are integral to the global water cycle and therefore vital for water security – they regulate water quantity, quality and timing and provide protective functions against (for example) soil and coastal erosion, flooding and avalanches. Forested watersheds provide 75 percent of our freshwater, delivering water to over half the world's population.

The purpose of *A Guide to Forest–Water Management* is to improve the global information base on the protective functions of forests for soil and water. It reviews emerging techniques and methodologies, provides guidance and recommendations on how to manage forests for their water ecosystem services, and offers insights into the business and economic cases for managing forests for water ecosystem services.

Intact native forests and well-managed planted forests can be a relatively cheap approach to water management while generating multiple co-benefits. Water security is a significant global challenge, but this paper argues that water-centred forests can provide nature-based solutions to ensuring global water resilience.

Monitoring and reporting

Standardized global methods for monitoring forest–water relationships are lacking – likely because of the highly contextual nature of forests and water, resource and capacity limitations, regional research bias, and the prioritization of other forest ecosystem services such as carbon sequestration and biodiversity conservation.

Forest–water interactions are context-specific, and major issues exist in defining riparian zones and determining how best to monitor and manage them. In this paper we build on current knowledge to present a new approach for the monitoring of riparian forests with available data and software. This is a significant step in addressing forest–water relationships, biodiversity and other ecosystem services at the watershed, landscape and national scales.

New tools and citizen science can be used to advance forest–water monitoring and thereby improve policy and management decisions. Developments in remote sensing and user-friendly image-processing technologies such as the System for Earth Observation Data Access, Processing and Analysis for Land Monitoring – SEPAL, the availability of decision-support tools such as Forest and Landscape Water Ecosystem Services – FL-WES, and the increased use of citizen science (e.g. the Blue Targeting Tool) are enabling scientists, government agencies, practitioners and managers to close major gaps in forest–water monitoring.

There is a need to address the contextual nature of forest–water interactions through approaches that combine global observations and national monitoring databases. Mixed approaches that include remote sensing and field methodologies provide a way forward for the accurate assessment of forest–water interactions.

Managing forests for water

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. Trade-offs may be required, but there may also be synergies; for example, water quality is closely linked to soil conservation, a priority of sustainable forest management for timber production.

The quantity of water flowing 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. 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 (at higher latitudes and altitudes) and – by influencing soil health – the amount of water stored in soils. These types of impact can alter the seasonal timing of flows. Monitoring is essential for ensuring that management practices do not cause negative impacts on water timing.

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 silvicultural practices can help maintain or improve water values, with their application varying depending on factors such as forest type, other forest management objectives, forest condition, the resources available for management, time of year, and desired future condition. The impacts of commonly used management practices such as the construction and maintenance of road infrastructure, harvesting, and forest regeneration on forest water resources are examined, along with key means to minimize these.

Ecosystem management tools are available to assist in managing forests to benefit water quantity, quality and timing, and many examples exist of effective forest management for the timely delivery of clean drinking water to cities. Conversely, poor forest management can have long-term negative impacts on forest health and water resources.

Valuing water from forests

The global provision of water services decreased by nearly USD 10 trillion per year between 1997 and 2011.

The valuation of ecosystem services is the starting point for managing forests and all the benefits they provide. Several methodologies have been put in place for recognizing the value of the ecosystem services provided by forests. The value of an ecosystem service can be derived from information provided by market transactions relating directly or indirectly to that ecosystem service, or from hypothetical markets that may be created to elicit values.

Payments for watershed services (PWS) are a promising mechanism for benefit-sharing and cooperation among the forest and water sectors, especially in the absence of legislative frameworks or functioning local governance. Nevertheless, PWS should be seen as part of a broader process of local participatory governance rather than as a market-based alternative to ineffective government or community management.

Networks and collaborative approaches at the local level are a common characteristic of successful PWS schemes, in which regulators, private companies, local authorities and technical and civil-society organizations share their expertise – through matched funding – to deliver high-level forest watershed schemes.

The two most common PWS schemes in the forest–water domain are water fees (utility-led) and multiple-benefit partnerships. Schemes that apply fees for water use are usually based on a defined normative background. National governments may incentivize these schemes through appropriate regulations; examples are provided.

There is value in employing a communication strategy as a means to increase the effectiveness of forest–water initiatives. Properly developed and deployed, it will assist in gaining political and public support and funding; strengthen the morale and internal organization of institutions and partnerships involved in the initiative by providing a broader vision and mission; engage more beneficiaries and buyers and thereby help spread the word; and build trust and relationships with new users, including ethnic minorities, women and youth.

Based on an analysis of communication strategies for existing forest–water projects and nature tourism, we propose a nine-step process for designing a communication strategy as a means to enhance community engagement, policy commitment and willingness to invest.

Key ecosystems for forest–water management

We examine four forest types of particular importance in forest–water management and provide guidance for optimizing their roles.

Mangroves. There are approximately 13.8 million hectares of mangrove forests worldwide; they provide many essential ecosystem services and play important roles in climate-change mitigation and adaptation. An estimated 30–35 percent of mangroves has been lost since the 1980s, and about one-quarter of remaining mangroves is considered to be moderately to severely degraded. Forest width is the most important factor determining the mitigation potential of mangrove forests against tsunamis and storm surges. Integrating mangroves in disaster risk reduction strategies and coastal management planning can help reduce the risk of coastal disasters.

Peatland forests. Wetland forests growing on peat soils play crucial roles in water regulation (flood and drought mitigation) and the maintenance of water quality at the catchment level. Unlike other forest types, there is a synergistic relationship between the water and carbon services provided by peatland forests. Peatlands are the world's most carbon-dense terrestrial ecosystems; their conservation is one of the most cost-effective ways to decrease greenhouse-gas emissions.

Peatland drainage dramatically increases the risk of fire, and it is estimated that one-quarter of the world's peatland forests disappeared between 1990 and 2008. Effective peatland ecosystem restoration would help ensure the delivery of water-filtering and regulating services and also provide sustainable livelihoods options in wet peatlands while reducing forest and peat fires and land degradation and loss.

Tropical montane cloud forests (TMCFs). TMCFs are among the most valuable terrestrial ecosystems for their role in the hydrologic cycle because they influence the amount of available water and regulate surface and groundwater flows in watersheds while maintaining high water quality. The high water yield of TMCFs arises from their location in areas with high rainfall, additional inputs of cloud-water capture by canopies, and low evaporative losses.

TMCFs are rare; area estimates range from 1 percent to 14 percent of tropical forests globally. Approximately 55 percent of the original area of TMCFs has been lost. The conservation of remnant mature TMCF forests needs strengthening and their conversion to agricultural land uses should be avoided.

Low-intensity selective logging in secondary TMCFs conforming with low-impact logging guidelines is strongly recommended to mitigate the deleterious effects of logging on soils, water yields and biomass. In restoring TMCFs, efforts should be made to plant mixtures of native water-use-efficient species. Payment schemes for the water services of TMCFs could help compensate landowners, maintain forest cover and counteract deforestation and water scarcity. Research is needed to better understand the hydrologic impacts of climate change on TMCFs.

Dryland forests. There are 1 079 million hectares of forests in drylands, supporting the livelihoods of millions of people globally. Dryland forests and trees survive and grow on limited water resources, but they also influence various components of the water cycle and water availability.

Climate-change projections indicate an expansion towards more arid dryland ecosystems, altering the ecological space of tree species and affecting hydrologic processes. Management strategies for dryland forests, such as canopy opening, pruning and species selection, might help combat local water scarcity by increasing soil and groundwater recharge. Given the complexity of multi-objective management and the intrinsic variability of dryland forests and other dryland systems with trees, more effort is needed to quantify and value the goods and ecosystem services produced in these systems and the management options available. The reuse of wastewater can help in maintaining dryland ecosystem services in the face of water scarcity.


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1 Introduction

Key points

- Forests and trees are integral to the global water cycle and are therefore vital for water security. Forested watersheds provide 75 percent of our freshwater, delivering water to over half the world's population.
- Water security is a significant global challenge. A water-centred approach to forest management can provide a nature-based solution for increasing global water resilience.
- Changes in tree cover mean changes in hydrology; watersheds with significant tree-cover loss are at greater risk of soil erosion, water stress and forest fire.
- Our understanding of forest–water relationships has increased significantly in recent decades. This knowledge can now be applied to how forests are monitored, measured and managed.

The importance of integrated forest–water management has gained recognition since the Shiga Declaration on Forests and Water in 2001 (Springgay *et al.*, 2019). A thematic study on forests and water was carried out in 2008 within the framework of FAO's Global Forest Resources Assessment (FRA) (FAO, 2008), but advances have been made since then in understanding forest–water relationships. Several scientific reviews have addressed these, notably the International Union of Forest Research Organizations' Global Forest Expert Panel report on forests and water (Creed and van Noordwijk, 2018). FAO (2013) summarized the key recommendations of several international fora, calling for policies and practices that incorporate an integrated science-based approach. Those recommendations, which are presented in Box 1.1, were reiterated in Creed and van Noordwijk (2018) and by a group of experts in the forest and water sectors (Springgay *et al.*, 2018).

BOX 1.1

Summary of recommendations from Forests and Water – International Momentum

Process understanding and research

- Conduct interdisciplinary research to improve understanding of forest and water interactions as a function of the seasons, climatic zones, geological conditions, stand development stages, native versus non-native species, natural versus planted forests and forest management practices.
- Develop long-term monitoring systems and tools on qualitative and quantitative changes of water resources within and from forested catchment areas.

Cooperation, policy and institutional development

- Develop innovative, cross-sectoral and, if appropriate, transboundary institutional mechanisms and policy proposals to enhance collaboration between the forest and

Continued ...

water sectors. These should be based on an understanding of existing legislations, policies and institutional mechanisms related to forests and water, including lessons learned, critical issues and knowledge gaps, as well as challenges and opportunities that can hinder or propel join management.

Economic incentives and mechanisms

- Analyse existing experiences and explore the potential for new and innovative economic mechanisms, incentives and benefits with regard to forest and water management. Conduct cost–benefit analyses in specific management areas to explore the financial viability of payment schemes for water-related forest services. Define the legal instruments for the development of such schemes and test them through the implementation of pilot field projects.
- Develop and foster collaboration with the private sector.

Climate-change mitigation and adaptation

- Consider forest and water relationships as an integral part of the development of national climate-change mitigation and adaptation strategies, disaster risk management plans and integrated approaches in planning processes.
- Promote forest and water issues in international climate-change-related dialogues and negotiations, with particular reference to the United Nations Framework Convention on Climate Change and the World Water Forum. Assess the impacts of other drivers of change on forest and water interactions, such as the energy crisis and changes in production and consumption patterns.

International dimension

- International organizations are encouraged to provide technical support to countries, for example through the organization of technical workshops and seminars for the exchange of national experiences on joint forest and water management. International organizations are encouraged to facilitate the strengthening of existing or the development of new transboundary institutional mechanisms related to forests and water.

Awareness-raising, capacity development and communication

- Develop and implement training programmes on the various aspects of integrated forest and water management that are able to develop the capacities of concerned technicians and decision-makers up to the highest levels.
- Develop and widely disseminate awareness-raising and communication materials related to forests and water and their links to food security. Scientists are encouraged to contribute to awareness-raising, capacity development and communication by "translating" research findings into applied and policy-relevant key messages.

Forest and water management

- Ensure, in forest and water management, that the benefits of forests for water quantity and quality are optimized. Carefully balance the trade-offs between water consumption by trees and forests and the protection functions, as well as other environmental services, provided by forests and trees.
- Apply an integrated and landscape approach to forest and water management at the local, national and transboundary levels. Ensure the links to other land uses and communicate the important contribution of forest and water management to food security and livelihood improvement.

Source: FAO (2013).

Advances in scientific knowledge should be reflected in how forests are monitored, measured and managed for the provision of their water-related ecosystem services (abbreviated hereafter to water services). Thus, FAO decided to conduct the present study to complement FRA 2020[1] by exploring the importance of forests in the hydrologic cycle and presenting information on maintaining and restoring their water services. Ultimately, the aim is to improve the information base on forest–water management and provide guidance for:

- improving forest–water monitoring and reporting;
- taking water more fully into account in forest management, including through examples of successful forest management for water; and
- providing a business case for managing forests for their water services.

THE IMPORTANCE OF FOREST–WATER RELATIONSHIPS

Forests and trees are integral components of the water cycle (Creed and van Noordvijk, 2018), regulating water quantity, quality and timing and providing protective functions against (for example) soil and coastal erosion, flooding and avalanches.

Forests are vital for water security: forested watersheds (Box 1.2) contribute 75 percent of the world's accessible freshwater, providing water to over half of the world's population (Millennium Ecosystem Assessment, 2005a). Forests provide water to over 85 percent of the world's major cities; on average, the source watersheds of the largest 100 cities are 42 percent forests, 33 percent cropland and 21 percent grassland, including both natural and pastureland (McDonald and Shemie, 2014). As tree cover changes in a landscape, however, so too does the hydrology. Major watersheds that experience more than 50 percent tree-cover loss are at greater risk of erosion, forest fire and base water stress (World Resources Institute, 2017). Changes in tree cover due to deforestation, forest growth, reforestation and afforestation all affect water services. It is estimated that land conservation and restoration, including forest protection, reforestation and agroforestry, and/or reducing forest fuel loads could lead to a reduction of 10 percent or more in sediments and nutrients in watersheds, with the potential to improve water quality for more than 1.7 billion people living in large cities at a cost of less than USD 2 per person per year (World Bank, 2012; MacDonald and Shemie, 2014; Abell *et al*., 2017).

BOX 1.2

Defining a watershed

A watershed is a functional land definition describing the basin influencing a stream or river network above a certain point in the landscape. It is a multiscalar concept with no fixed spatial scale. Any upstream area that is hydrographically linked to a point in a stream or river is part of the watershed that influences water supply at that point. Watersheds, therefore, are nested. Many small watersheds of headwater streams are contained within the watersheds of larger downstream rivers or other bodies of water such as lakes and deltas. The term "basin" often describes a large watershed of a named river (e.g. the Amazon River basin).

Water availability is a major factor constraining humanity's ability to meet future global food and energy needs (D'Odorico *et al*., 2018), and water is expected to become an even more scarce resource in the future. Human demands for water, energy and food are projected to increase by 30–50 percent; under a business-as-usual climate scenario, the world will face a 40 percent global water deficit by 2030 (The 2030 Water Resources

[1] FRA 2020 (FAO, 2020a) was the result of a collective effort by FAO, FAO member countries and institutional and resource partners. It involved more than 700 individuals, including national correspondents and their teams, who provided detailed country reports. In addition to the main FRA 2020 report, several thematic studies have been prepared, of which this is one.

Group, 2009; WWAP, 2015). Comprehensive, integrated water and land management plans are needed to tackle the problem of water quality and availability.

Many people worldwide lack adequate access to clean water to meet basic needs. The majority of the estimated 4 billion people with insufficient access to clean water live in areas with low forest cover and depend on engineered infrastructure to redistribute water across watershed boundaries. Intact native forests and well-managed planted forests can be a cheaper approach to water management while generating multiple co-benefits (Creed and van Noordwijk, 2018). In the United States of America, for example, national forests supply water to approximately 50 percent of the country's population. There is an urgent need, therefore, to address the role of forests in the provision of water and to manage forests in ways that increase water security.

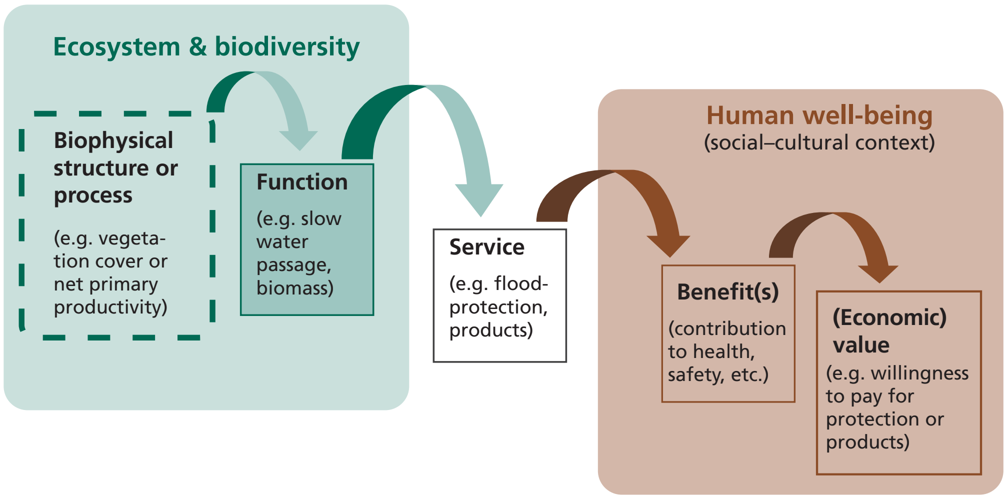
Climate change is likely to aggravate water stress. As temperatures rise, ecosystems and the human, plant and animal communities that depend on them will need more water to maintain their health and to thrive. Many important economic activities, such as energy production and agriculture, also require water. The volume of accessible water may reduce as the planet warms (Melillo, Richmond and Yohe, 2014).

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). Understanding the close relationship between forests and water is essential for effective forest and water management practices and policies; science, therefore, should inform management strategies for the world's forests in the face of ongoing climate change and its consequences for forests and people. Moreover, taking the forest–water nexus into consideration will contribute to achieving the Sustainable Development Goals and other globally agreed objectives. On the other hand, a failure to ensure a robust science-based approach, as well as a lack of coordination among multiple needs, goals and policies, will have consequences that likely will be unevenly distributed geographically, socially, economically and politically (Creed *et al*., 2019).

Water services provided by forests

Ecosystems are the "planet's life-supporting systems, for the human species and all other forms of life", and ecosystem services are the "multiple benefits provided by ecosystems to humans" (Millennium Ecosystem Assessment, 2005b). Figure 1.1 depicts the connection between ecosystem services and human well-being (TEEB, 2010). The functions derived from biophysical structures and processes express the potentiality of ecosystems to deliver services; the services, therefore, are the potential contributions of ecosystems to human welfare. This welfare, in turn, is built on what are called benefits, which can be measured to obtain the economic value of ecosystem services. The spatial distribution of function and benefit is also crucial to understand – that is, where the function occurs, where the provision of the service can be assessed, and ultimately where the benefits are appreciated (TEEB, 2010).

FIGURE 1.1
Connection between ecosystem services and human well-being



Source: Adapted from TEEB (2010).

There have been many attempts to classify ecosystem services. The Millennium Ecosystem Assessment (2005b) divided such services into four main categories:

1. supporting services (which create the conditions for the other services to exist);
2. provisioning services (the generation of products and materials);
3. regulating services (responsible for the regulation of ecosystem processes); and
4. cultural services (intangible benefits that enrich lives).

As a fundamental component of ecosystems, water has a key role in all these categories (Millennium Ecosystem Assessment, 2005a). The focus of this publication, however, is on the water services provided by forests. Brauman *et al.* (2007) defined hydrologic services as the "benefits to people produced by terrestrial ecosystem effects on freshwater" and proposed the five water services shown in Table 1.1.

TABLE 1.1
Classification of water services

Brauman *et al.* (2007) category	Millennium Ecosystem Assessment (2005b) category	Description of service
Improvement of extractive water supply	Provisioning	Effects on water extraction for municipal, agricultural, commercial, industrial and thermoelectric power generation uses
Improvement of instream water supply	Provisioning	Effects on *in situ* water use for hydroelectricity, recreation, transportation and the supply of fish and other freshwater products
Water damage mitigation	Regulating	Effects on reduction of flood damage, dryland salinization, saltwater intrusion and sedimentation
Provision of water-related cultural services	Cultural	Provision of religious, educational and tourism values
Water-associated supporting services	Supporting	Water and nutrients to support plant growth and habitats for aquatic organisms, and the preservation of options

Sources: Adapted from Brauman *et al.* (2007); Masiero *et al.* (2019).

MANAGING FORESTS FOR WATER

The FRA takes into account forest management as it relates to water in a single indicator – “total area of forests managed for soil and water conservation as a primary management objective”. On its own, this indicator is insufficient for understanding the extent to which forests are managed for soil and water services; information is also required on the types of forests managed for these purposes, the ways in which they are managed and where they are located. It is generally assumed that forests that are protected for certain other management priorities (e.g. biodiversity) will also provide water services; it is also often assumed that water services are a default byproduct of sustainable forest management (e.g. minimizing soil compaction and erosion during timber harvesting). To a certain extent, this may be true. Nevertheless, as discussed in this report, maintaining and optimizing forest-based water services generally requires water-centred management – and where such forests are located in a landscape matters.

With increasing pressure on water resources due to a growing human population, expanding urban centres, widespread land degradation and climate change, water security looms as a major challenge for the planet. Forest management can provide a nature-based solution.

Given the importance of water for all aspects of life and for domestic, agricultural and industrial purposes, a strong argument can be made that maintaining and enhancing the water services of forests should not only be a conscious management decision but also a high management priority. What would that mean for forest management? What would managing forests for water look like? This report aims to answer these questions (among others).

Advances in remote sensing and rapid field assessments are making it easier to assess the extent to which forests are delivering water services. After reviewing the fundamental roles of riparian forests in forest–water relationships, Chapter 2 of this report shows the importance of triangulating remote sensing data with field methods. The chapter, which is especially relevant to technicians involved in national forest monitoring and managers interested in ensuring water services, also provides guidance on implementing forest–water monitoring frameworks, including establishing baselines.

Forest management has focused on biomass production since the early twentieth century (Parde, 1980). The protection of forests for biodiversity conservation has been perceived mainly as the maintenance of a “natural” state and therefore requiring little active management. Sustainable forest management for multiple uses has become more prevalent in recent decades, with water services usually supplied as a byproduct. Nevertheless, there are circumstances in which water services should be a management priority. Chapter 3, which is most relevant to forest managers, advocates more conscious management for water-related objectives, taking into account both spatial and temporal scales.

It is important to understand the trade-offs and synergies involved in sustainable forest management. Chapter 4 considers the value of forest-related water services and how to develop a business case for managing forests for water. This chapter is likely to be especially useful for policymakers, economists and foresters engaged in national or subnational forest management, including watershed management.

Chapter 5 brings together the various concepts explored in chapters 3 and 4 by showcasing forest ecosystems in which management for water services is particularly important and which are highly vulnerable to climate change, deforestation, land degradation and land-use change.

2 Monitoring and reporting on the forest–water nexus

Key points

- This chapter builds on current knowledge to present a new approach for the monitoring of riparian forests with available data and software. This is a significant step in addressing forest–water relationships, biodiversity and other ecosystem services at the watershed, landscape and national scales.
- New tools and citizen science can be used to improve forest–water monitoring and thereby improve policy and management decisions.
- Forest–water interactions are context-specific, and major issues exist in defining riparian zones and determining how best to monitor and manage them.
- Although the remote sensing-based monitoring of forest–water interactions is improving rapidly, major limitations still exist related to, for example, image resolution, the availability of field-level data, and access to models and technology for handling such data.
- Developments in remote sensing and user-friendly image-processing technologies and the increased use of citizen science are enabling scientists, government agencies, practitioners and managers to address major gaps in forest–water monitoring.
- There is a need to address the contextual nature of forest–water interactions through approaches that combine global observations and national monitoring databases. Mixed approaches that include remote sensing and field methodologies provide a way forward for the accurate assessment of forest–water interactions.

The purpose of forest monitoring and reporting is to provide the information needed to understand the extent, condition, management and use of forest resources and to adapt management accordingly to ensure that forest-related goals are met. The monitoring and reporting process involves standardizing definitions and procedures to provide a means for comparison.

FAO has been providing globally compiled information on forests and their resources since 1948. The FRA process combines national data collated via a global network of officially nominated national correspondents with remote sensing and other sources to provide a wide range of information on forests that governments, civil society and the private sector can use in developing forest-related policies, objectives and priorities. The FRA is integral to the monitoring of Sustainable Development Goal 15 ("life on land") by collecting information for and reporting on indicators 15.1.1 and 15.2.1 and contributing to indicator 15.4.2. The FRA has reported on forests managed for soil and water conservation since 2005.

This chapter presents pragmatic, readily available methodologies and tools for forest–water monitoring and reporting, including remote sensing, modelling and field-based methods. These methods and tools can be adapted and applied at the local level by combining remote sensing with field methods. The benefits and limitations of each tool and method are discussed, and case studies are provided.

The purpose of the chapter is not to impose a standardized global indicator or method or to provide an exhaustive list of methods and tools (other methods and tools exist in addition to those presented here). Rather, the objective is to raise awareness of the forest–water nexus and to promote the inclusion of water in forest resource monitoring and reporting, thereby encouraging informed management and policy decision-making that addresses synergies and trade-offs in multipurpose sustainable forest management.

THE GLOBAL SITUATION

Standardized global methods for monitoring forest–water relationships are lacking – likely because of the highly contextual nature of forests and water, resource and capacity limitations, regional research bias, and the prioritization of other forest ecosystem services such as carbon sequestration and biodiversity conservation.

The interrelationships between forests and water are explicitly mentioned in two SDG targets (6.6 and 15.1; Table 2.1), but indicators and methods are lacking for quantifying these relationships and informing policy and practice (FAO, 2018). FAO (2018) proposed two potential global datasets to address this gap: change in the extent of tree cover in major global watersheds over time based on the Global Forest Watch Water database (World Resources Institute, 2017); and the proportion of forests managed for soil and water conservation as a key objective (based on FRA data).

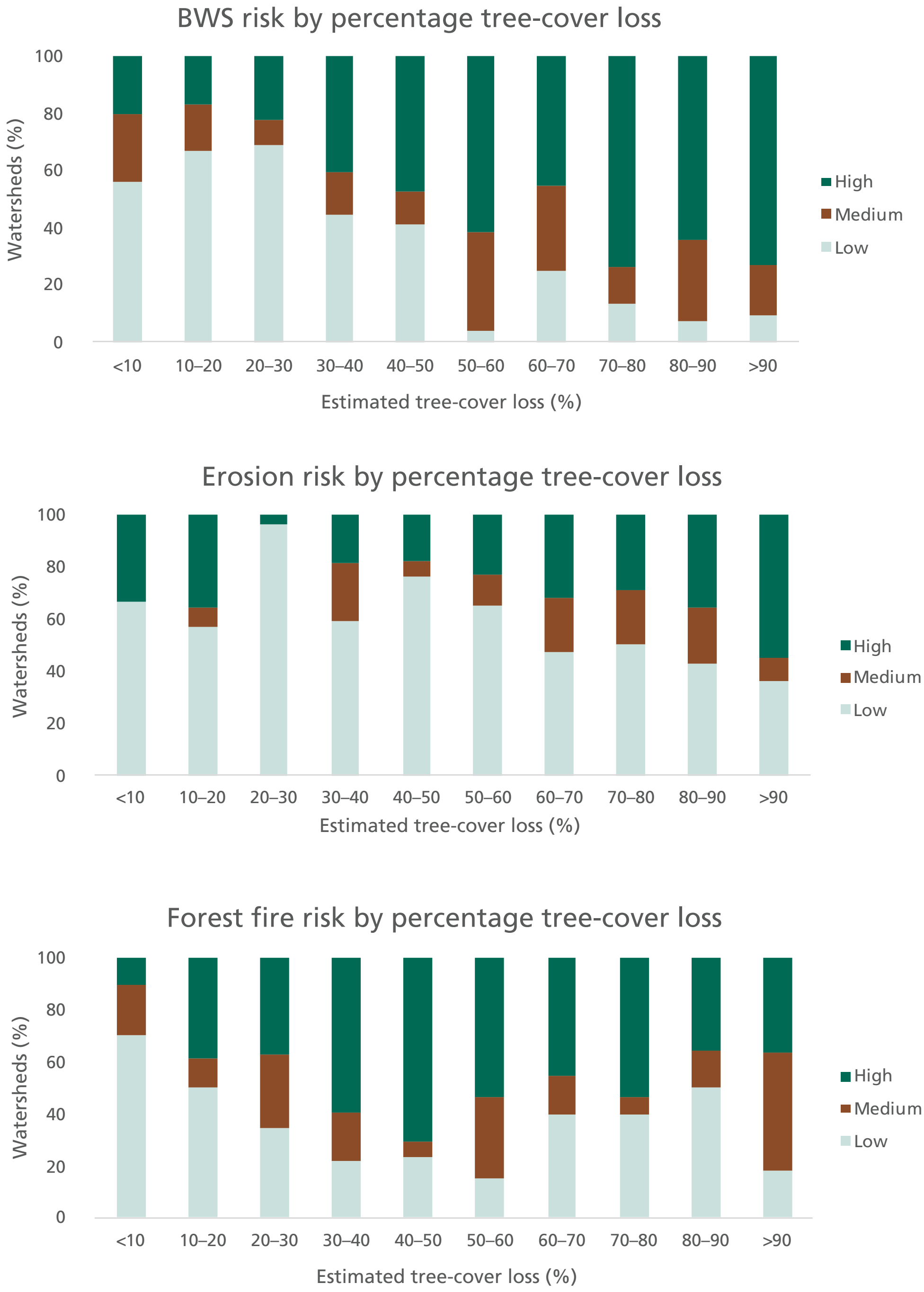
TABLE 2.1
Sustainable Development Goal targets related to forests and water

Sustainable Development Goal	Target
6 – clean water and sanitation	6.6 – By 2020, protect and restore water-related ecosystems, including mountains, forests, wetlands, rivers, aquifers and lakes
15 – life on land	15.1 – By 2020, ensure the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services, in particular forests, wetlands, mountains and drylands, in line with obligations under international agreements

It has been estimated that tree cover in major watersheds averaged 67.8 percent historically[2] but that this had declined to only 30.7 percent by 2000 (World Resources Institute, 2017). This tree-cover loss (i.e. forest loss and the loss of trees outside forests combined) has generally resulted in an increased risk of erosion, forest fire and baseline water stress. Of the 230 major global watersheds that had lost more than 50 percent of their original tree cover by 2015, there is a medium to high risk of erosion in 88 percent, of forest fire in 68 percent and of water stress in 48 percent (Figure 2.1).

[2] Historical tree cover refers to the estimation of tree cover for the decades before 2000; it has been calculated based on potential forest cover, tree cover and climate zones (Qin *et al.*, 2016; World Resources Institute, 2017).

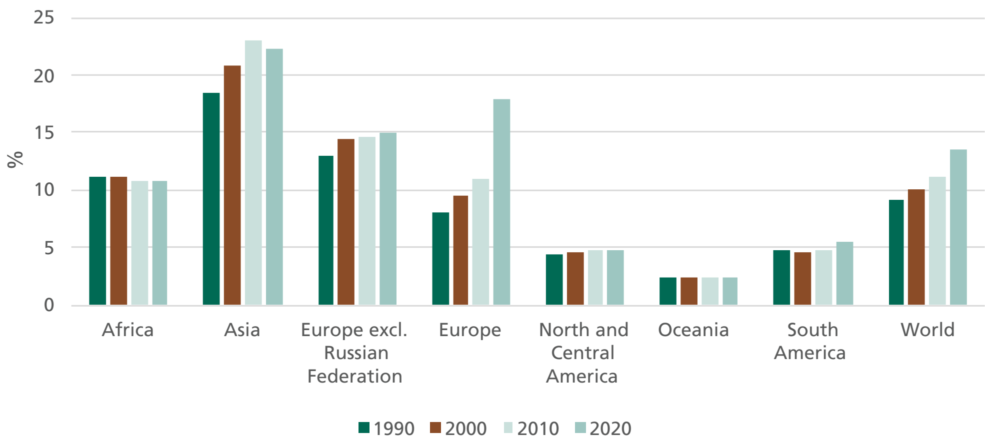
FIGURE 2.1
Potential relationship between tree loss and the risk of erosion, forest fire and baseline water stress



Note: BWS = baseline water stress.
Source: Adapted from the Global Forest Water database (World Resources Institute, 2017).

The FRA includes the indicator, "total area of forests managed for soil and water conservation as a primary management objective".[3] According to FAO (2020a),[4] 398 million hectares (ha), or 12 percent of the total forest area globally, is designated primarily for the conservation of soil and water, up by 119 million ha since 1990. Europe (including the Russian Federation) has the largest total area, at 171 million ha (18 percent of the region's total forest area), but Asia has the largest proportion of forests designated primarily for soil and water conservation, at 22 percent of the region's total forest area (132 million ha). All the main regions globally show positive trends in the area of forests designated primarily for soil and water conservation except Africa and Oceania, where there was little change in the area so designated between 1990 and 2020 (Figure 2.2).

FIGURE 2.2
Proportion of total forest area designated primarily for the conservation of soil and water, by region



Source: FAO (2020a).

Table 2.2 shows the top ten countries globally for the proportion of total forest area designated primarily for soil and water conservation (FAO, 2020a). All ten are either island nations or mainly comprise mountainous terrain or drylands and have experienced high levels of degradation and desertification. All these countries are highly vulnerable to disasters, and their forests offer increased resilience and an ability to maintain high-quality water supplies.

3 The FRA also provides data on the area of forests designated primarily for biodiversity conservation, and it can be assumed that such areas are likely to also provide water services. It cannot be assumed, however, that water was a consideration in the selection or management of these areas or will be factored into management in the future.

4 FRA 2020 (FAO, 2020a) received information on the area of forest designated primarily for soil and water conservation from 141 countries and territories representing 82 percent of the world's total forest area of 4.06 billion ha. In 2015, the area of forest so designated represented 31 percent of the forest area of the reporting countries and only 121 countries reported on this indicator. FRA 2015 adopted a slightly different approach to other FRAs, in which the variable referred to the total forest area managed for the protection of soil and water (other FRAs refer to the forest area designated primarily for soil and water conservation). Therefore, the data for FRA 2015 are excluded from this comparison.

TABLE 2.2
Top ten countries and territories for the proportion of total forest area designated primarily for soil and water protection

	Country/territory	Area (1 000 ha)	% of total forest area
1	Kiribati	1.2	100
2	Kuwait	6.3	100
3	Cabo Verde	44.7	98
4	Kyrgyzstan	1 212	92
5	Tunisia	627	89
6	Wallis and Futuna Islands	5.1	87
7	Bahrain	0.6	86
8	Uzbekistan	2 532	69
9	Mongolia	9 192	65
10	Kazakhstan	2 160	63

Source: FAO (2020a).

HOW TO MEASURE FOREST–WATER RELATIONSHIPS

Forests and water interact at various spatial scales, from continental – in the case of major river basins and moisture recycling through evapotranspiration – to local, for example in small forest stands and riparian forests along streams. This wide range of interactions means that, if it is to provide reliable evidence for science-based policies and management, forest–water monitoring must take site-specific interactions into account at differing spatial scales.

The temporal scale is also important because forest management decisions can have short- and long-term impacts. For example, removing forests and trees may lead to an increase in water quantity in the short term but a decrease in water quantity, quality and timing (also called “water values” in this report) in the long term (Springgay *et al*., 2019; FAO, 2008). Moreover, the impacts of restoration efforts may take months or years to manifest and may therefore be hard to measure in the short term. This poses a challenge because decision-makers may need to wait several years to see significant results – and even longer at larger spatial scales.

Thus, depending on its purpose, the monitoring of forest–water interactions needs to happen at different spatial and temporal scales, requiring the use of different monitoring tools and approaches. For example, national monitoring to measure the effectiveness of national policies and for reporting on international commitments may best be done using a combination of remote sensing and national networks of monitoring stations, requiring significant investments in capacity development, planning and funding. Conversely, at the local level, forest managers need simple, low-cost monitoring tools that enable them to make decisions almost in real time and to alert them to significant changes in an ecosystem or landscape that may require immediate action.

Regardless of scale, effective evidence-based forest–water management and monitoring requires suitable indicators: major global data and knowledge gaps exist partly because of a lack of appropriate forest–water indicators (Springgay *et al*., 2019). Local authorities, forest managers and communities need to develop forest management plans that take into account forest–water interactions and include appropriate measuring and monitoring protocols. This is challenging but, as shown below, monitoring and management tools are now being developed for these purposes.

Monitoring methodologies

Remote sensing. The development of a wide range of remote sensing products has increased the ability of governments, researchers and forest managers to monitor change in forest ecosystems over time. Remote sensing-based products and models can be particularly useful for monitoring disturbances and their impacts, thus assisting management decision-making and emergency responses in real time. This was the case, for example, during Australia's historically significant 2019–20 fire season, when earth-observation technologies and modelling were used as part of the preparation phase to assess fire risk and later in the emergency-response and postfire phases (Bushfire Earth Observation Taskforce, 2020; USGS, 2020). In addition to assessing fire risk and burnt areas and detecting the location of settlements, remote sensing-based products were used to monitor the availability and quality of water, which can be severely affected by ash and debris during and after fire (USGS, 2020).

A number of important forest–water variables can be measured using remote sensing-based products, such as forest leaf area and vegetation indices, thereby providing information on tree water use and content, soil water, surface soil moisture, water levels, water quality, the presence of water bodies and land cover (Hunt, Ustin and Riaño, 2015; Copernicus, 2020). Technologies that combine satellite and unmanned aerial vehicle (drone) images are increasingly able to collect information at fine spatial scales.

Remote sensing methodologies, tools and models for forest and water monitoring continue to develop rapidly (Box 2.1). Even at coarse spatial scales, these can be highly cost-effective and accurate (Box 2.2).

Although capability is advancing in the use of remote sensing for monitoring forest–water interactions and accuracy is increasing, very-high-resolution (VHR) images and field data are still required to validate and finetune models. Models often include assumptions that oversimplify complex forest–water interactions, which vary spatially and temporally. Outcomes may be inaccurate and misleading if such models are not calibrated and triangulated with relevant field data, supplemented with data from other ecosystems, leading, in turn, to poor management decisions. It is important, therefore, for forest and water managers and decision-makers to work with scientists and others to develop better decision-support systems that use the best available science and data from both remote sensing and field monitoring.

BOX 2.1

FAO's state-of-the-art tool for everyone

The development of online cloud computing has enabled a paradigm shift in access to and the processing of large amounts of remote sensing and ancillary geographic data. Nevertheless, Google Earth Engine requires programming skills that are not always readily available in water and forest agencies.

To address this, FAO has developed the cloud-computing forest monitoring platform, SEPAL ("System for Earth Observation Data Access, Processing and Analysis for Land Monitoring"). This user-friendly platform offers developing countries unparalleled access to granular satellite data and supercomputing power, enabling users to query and process satellite data quickly and efficiently, tailor their products to local needs, and rapidly produce sophisticated and relevant geospatial analyses.

The modular nature of SEPAL enables users to implement virtually any processing chain of remote sensing data written in commonly used programming languages (e.g. C++, Python, Javascript and R), with the option of not interacting with the scripts. Thus, SEPAL's user-friendly interface provides the public with easy access to the processing chain, enabling wide usage by academics, researchers and institutions.

Harnessing cloud-based supercomputers and modern geospatial data infrastructures (e.g. Google Earth Engine), SEPAL provides access to and enables the processing of historical satellite data and newer data from Landsat and higher-resolution data from the European Union's Copernicus programme.

The SEPAL interface enables non-specialists to, among other things, create cloud-free mosaics from a range of satellites for a given region on given dates; develop random stratified sample schemes that can then be entered into FAO's Open Foris Collect Earth visual interpretation utility; analyse phenological trends in a given region; and create thematic classifications for large regions. SEPAL requires a stable Internet connection but does not need high bandwidth.

SEPAL is paving the way for more accessible monitoring, such as that developed using drones – which are used increasingly for fine-scale monitoring, data validation and the refining of models. Commercial drone processing software is expensive, however, and licences are restrictive, presenting a barrier to the use of drones for forest–water monitoring, especially in developing countries. Open-source drone processing software is available and effective but requires significant computer power to run efficiently – another barrier to use in developing countries. The SEPAL platform can run open-source drone software using cloud computing and a user-friendly interface. This allows SEPAL users to process drone imagery anywhere in the world without worrying about a lack of computing resources and storage. Drone imagery can also easily be integrated into existing workflows for forest and land monitoring using other satellite data. SEPAL has been used successfully to process drone imagery in several countries for projects on forest monitoring and indigenous community mapping.

More information: FAO (undated).

BOX 2.2
Atlas of India's wetlands

Given the importance of wetlands in India's forests and the emphasis placed on wetland conservation, the Forest Survey of India inventorized the country's wetlands in recorded forest areas (RFAs). The Space Application Center in Ahmedabad mapped wetlands from 2006 to 2010 using the Linear Imaging Self-Scanning Sensor 3 (LISS III) and released the *National Wetland Atlas* in 2011 – the most recent information on the spatial distribution of wetlands in India.

An overlay analysis of the wetland layer over the RFA/green-wash layer was carried out to determine the number and extent of wetlands in various categories in RFAs in each state and union territory. The analysis showed that, among the large states, Gujarat has the largest area of wetlands in RFAs, followed by West Bengal. Among the smaller states and union territories, Puducherry has the largest area of wetlands in RFAs, followed by the Andaman and Nicobar Islands. Nationwide, there are 62 466 wetlands in RFAs covering 3.8 percent of the area; 8.13 percent of all wetlands are in RFAs.

Field methods. Field methods are qualitative or quantitative forms of data collection that aim to observe, interact and understand the natural environment. They provide real-life observations of changes in forest–water relationships due to direct and indirect influences, such as changes in land-use and climate change. Field-based approaches are also useful for validating ("ground-truthing") model-based methods. They have two main functions: providing data on parameters that remote sensing is unable to collect;

and validating data collected through, for example, remote sensing and desk studies. Decision-making support tools that enable the measurement of indicators of forest–water interactions and their monitoring can help forest managers take such interactions into account in their sustainable forest management plans. Two of these tools, described below, are Forest and Landscape Water Ecosystem Services (FL-WES) and the *Field Guide for Rapid Assessment of Forest Protective Function for Soil and Water,* both developed by FAO.

The Forest and Landscape Water Ecosystem Services tool

The FL-WES tool was developed by FAO with the aim of improving forest and water monitoring and addressing knowledge gaps. The four main objectives of the tool are to:

1. make monitoring more accessible to non-academic project developers and policymakers by providing an interactive, online platform that adapts automatically to the needs of users;
2. improve forest management decisions by making more explicit the link between forestry and hydrological dynamics through the provision of relevant indicators based on recent science and preferred methodologies;
3. provide practitioners with tools to collect, aggregate and visualize data from specific projects over time that cover a wide range of contexts globally; and
4. support the collection of data that can be used to inform management guidelines by providing users with ways to interpret data as a cost-effective alternative to publications.

The FL-WES tool guides users to appropriate forest–water indicators and monitoring methodologies. It is based on a monitoring framework developed with inputs by scientists and practitioners from several applicable disciplines. It includes six indicators, 16 subindicators and more than 130 methodologies covering quantitative and qualitative aspects of forest–water relationships and their potential impacts on societies and the environment.

Based on an initial guidance survey that takes into account environmental context, management objectives and existing human and financial resources, the tool provides users with context-specific methodologies for measuring indicators related to their management or project objectives. The FL-WES tool also provides guidance on additional indicators that should be measured and methodologies to consider in monitoring practices. The tool will be updated as science and monitoring practices evolve.

Figure 2.3 shows indicators and subindicators for measuring physical and chemical attributes of forest–water interactions (indicators 1–3) and socio-economic aspects (indicators 4–6), as listed by the FL-WES tool. Indicators 1, 2 and 3 and their subindicators are measured mainly through fieldwork and widely used, peer-reviewed quantitative methodologies. The tool identifies appropriate desk studies and models. Data for socio-economic indicators and subindicators are expected to be collected through qualitative methodologies, such as questionnaires and desk studies.

The FL-WES tool has downloadable data-collection templates for numerous variables related to indicators 1, 2 and 3. Sample questionnaires are available for indicators 4, 5 and 6. Both the data-collection sheets and questionnaires are customizable – an attribute that enables the use of the tool in any context.

EarthMap has been integrated into the FL-WES tool to help users with their data-collection needs. EarthMap is a user-friendly web application that can be used to conduct geospatial analysis for selected FL-WES project locations, such as land use, precipitation and temperature.

The FL-WES tool is most useful for national and subnational forest, water and environmental agencies and technicians. Many national agencies in charge of forest and water monitoring lack appropriate frameworks for integrating the forest–water nexus into policies and management practices; the FL-WES tool, combined with capacity

FIGURE 2.3
Forest monitoring framework outlining indicators and subindicators in the Forest and Landscape Water Ecosystem Services tool

INDICATOR 1

The status of water supply (quantity and timing) within and from forested areas in comparison to reference conditions

Subindicators

1.1: Change in surface flows
1.2: Change in groundwater recharge and soil water storage
1.3: Change in evapotranspiration
1.3.1: Change in catchment evapotranspiration
1.3.2: The impact of changes in forest cover from upwind catchments on local cloud cover and precipitation

INDICATOR 2

The status of water quality within and from forested areas in comparison to reference conditions

Subindicators

2.1: Change in water chemistry
2.2: Change in water body morphology

INDICATOR 3

Capacity of forests to provide water-related ecosystem services

Subindicators

3.1: The impact of changes in forest cover on water-related protective functions
3.2: The impact of changes in forest cover on biodiversity
3.3: The impact of changes in forest management on water-use efficiency
3.4: The impact of changes in forest cover on soil erosion in forested areas
3.5: Changes in water stress within a forest landscape

INDICATOR 4

The use of integrated forest–water management in practice

Subindicators

4.1: Conservation and sustainable forest management practised to enhance water-related ecosystem services

INDICATOR 5

The enabling environment for integrated forest–water approaches

Subindicators

5.1: Use of legal frameworks to support water-related forest services
5.2: Use of institutional frameworks to support water-related forest services
5.3: Use of economic frameworks to support water-related forest services

INDICATOR 6

The impact of water-related forest management on the provision of socio-economic benefits

Subindicators

6.1: Social and cultural benefits derived from forests managed for water-related purposes
6.2: Economic costs and benefits associated with forests managed for water-related purposes
6.3: Cost-effectiveness of forests managed for water-related purposes
6.4: Change in community water access and distribution due to forests managed for water-related purposes

development on the forest–water nexus, can help close this gap.

Recognition that capacity development is an essential element of improving forest–water monitoring and management is at the core of the FL-WES tool. Therefore, it is a key component of, and closely linked to, another FAO product, *Advancing the Forest and Water Nexus – A Capacity Development Facilitation Guide (Eberhardt et al., 2019).* The aim of this module-based facilitation guide is to help facilitators train stakeholders – from communities to politicians and practitioners – on the forest–water nexus, the importance of water considerations in forestry, the measurement and monitoring of forest–water interactions, and how to create forest–water action plans and follow up on them. Sessions, activities and case studies include work with the FL-WES tool.

Field Guide for Rapid Assessment of Forest Protective Function for Soil and Water

Trees, forest litter, undergrowth and forest soils contribute to the regulation of water quantity and the quality and timing of waterflows. They can reduce erosion, act as filters for pollutants, help moderate peak flows, prolong base flows and recharge groundwater and also contribute to soil organic matter and nutrients (FAO, 2008; Ilstedt *et al*., 2016; Pardon *et al*., 2017).

The *Field Guide for Rapid Assessment of Forest Protective Function for Soil and Water* is a pocket-sized product to support data collection on the soil and water protective functions of forests and thereby help forest managers and policymakers integrate forest and water objectives into management plans and forest, water and disaster-risk-management policies. The data collected using the methodology outlined in the guide can also easily be integrated with national inventories and national and global forest resources assessments, thereby improving the reporting capacity of countries and supporting evidence-based decisions and policies.

The methodology records data on forest canopy and ground cover and evidence of erosion. It requires a certain amount of monitoring knowledge, but forest managers can easily be trained in its application. The data allow users to address, for example:

- the conditions necessary for forests to provide soil and water protective functions;
- indicators for determining when interventions may be necessary for the protection of soil and water resources;
- critical topography for the protection of soil and water;
- the role of forest canopies in soil and water protection; and
- the critical level of forest canopy and ground cover for best management practices in soil and water protection.

RIPARIAN FORESTS – A NEW GLOBAL MEASURE FOR MONITORING FORESTS AND WATER

Riparian forests showcase the challenges, opportunities and data gaps in monitoring forest–water interactions. Riparian forests – forests located in riparian zones – provide important ecosystem services, but their monitoring and management are challenging, even in data-rich areas (Riis *et al*., 2020). Advances in remote sensing-based systems and rapid field assessment tools that support forest management decisions at local scales provide opportunities to create tools and methodologies that improve riparian-area management. This section reviews definitions of riparian forests; the challenges of implementing a given definition at a global scale; the potential of remote sensing technologies; available databases and methods; validation methods; and key limitations and gaps.

Defining riparian forest

Definitions of riparian forest have long been debated, and there are three broad categories: 1) those that consider geomorphological aspects; 2) those that consider the functions of riparian forests; and 3) those used for policy purposes. Riparian areas are highly varied, and they often comprise various types of vegetation that may not all fall within the definition of forest (Clerici *et al*., 2011). Thus, recommendations or regulatory stipulations for the width of "riparian forest" or the delineation of "riparian zones" may depend on the definition used. Choosing the right definition is important, therefore, for monitoring and managing riparian areas to ensure the provision of ecosystem services, especially in mixed-use landscapes where riparian forests might compete with other land uses, such as agriculture.

Historically, river dynamics have been explained broadly in terms of waterflow and sediment transport, erosion and deposition (Gurnell and Grabowski, 2015; Osterkamp, Hupp and Stoffel, 2011). Recently, however, it has become recognized that vegetation is an important driver of channel and floodplain morphological processes (Gurnell and Grabowski, 2015), and it has a direct impact on the provision of ecosystem services. Riparian zones are considered transitional ecosystems that occur between terrestrial and aquatic ecosystems and which vary in their characteristics with distance from the water channel or river. Riparian vegetation also varies in structure and function with bioclimatic regime, which drives water quantity and timing; the morphological patterns of river channels, affecting the type of vegetation and the stress and disturbance regimes; and land use, such as whether forested or agricultural land. An example of classifications based on such parameters is that of Gurnell *et al.* (2015), which provides a classification of riparian zones for European rivers.

Riparian zones are complex, and their interactions with adjacent terrestrial and aquatic areas give rise to processes that need to be taken into account in their delineation. Definitions to address this complexity have focused on riparian-zone functionality to account for the impact of riparian vegetation on hydrology, water quality, biodiversity, landscape connectivity and other ecosystem services (Luke *et al*., 2019). This functionality is crucial because it may change with disturbances such as dams, water diversions and climate change. Human disturbances may cause significant alterations to riparian vegetation through their direct impacts on bioclimate, morphology and land use; thus, riparian zones are socio-ecological systems, the specific characteristics of which depend on biophysical and anthropogenic factors (Dufour and Rodríguez-Gonzáles, 2019).

At least theoretically, a broad definition of riparian zones would enable the local tailoring of management approaches to account for their complexity and variation. This poses challenges for policymakers, however, because developing, implementing and enforcing policies that depend on local-scale interactions may be difficult. Thus, some governments have opted to, for example, establish minimum limits for riparian-zone widths based on a particular ecosystem type and applied nationally, independent of local river geomorphology. Setting a minimum width may mean, however, that important parts of some riparian zones are omitted from protection, with negative consequences for the ecosystem services they provide (Fernández *et al*., 2012).

The issues of definition and delineation are even more complex for monitoring. Countries and monitoring agencies have invested considerable human and financial resources in developing remote sensing methodologies in their jurisdictions. The benefits are clear: data can easily be gathered and analysed at various spatial scales across large areas without the need for fieldwork. This is tricky for riparian zones, however, because remote sensing products measure parameters based on physical attributes that can be highly variable in riparian zones. For example, digital elevation models (DEMs) can help determine the shape of channels and surrounding topography, which often dictates the extent of a riparian zone, especially in steep valleys with narrow channels. This is not the case everywhere, however, because riparian zones also exist in flat areas. In such cases, DEMs combined with information on flooding might give a better idea of the area under the influence of water (Fernández *et al*., 2012) – but such information may be unavailable. Models to delineate riparian zones could include vegetation presence, hydrology and biodiversity, among other variables, but the same issue of a lack of information arises (Fernández *et al*., 2012). Studies suggest that geographic information system (GIS) models and remote sensing images can be used to delineate riparian zones based on geomorphology and hydrology, but local-level calibration is needed to finetune the models, which are also limited by the spatial resolution of the DEMs (Fernández *et al*., 2012).

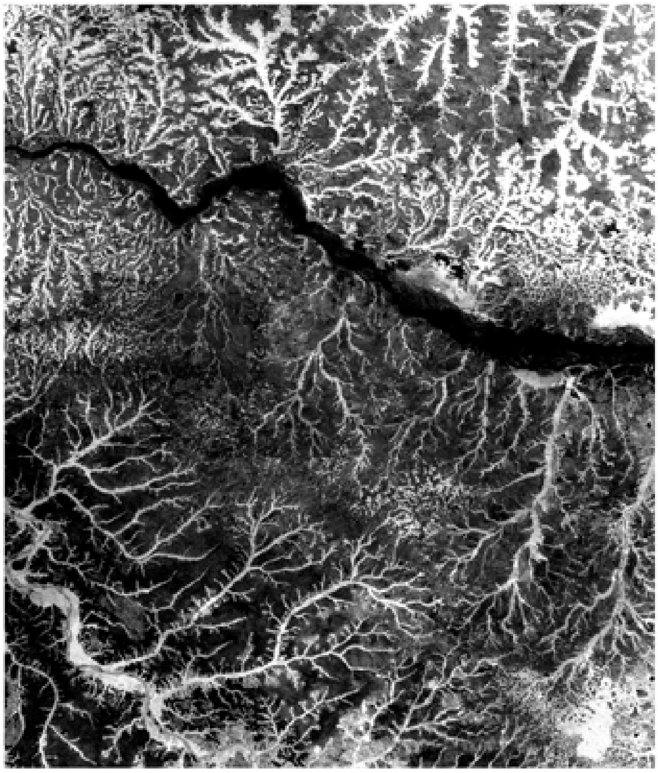
To summarize, riparian zones are complex in nature and affected by anthropogenic and biophysical factors. Their importance for both aquatic and terrestrial ecosystems cannot be overstated, and they therefore require careful management. Their delineation, management and monitoring should be context-specific, and the aim should be to maximize all the functions of these areas for the provision of ecosystem services. Nevertheless, there may also be opportunities to apply knowledge beyond site-specific dynamics, such as in the delineation of riparian zones using a combination of GIS and remote sensing technologies (e.g. Clerici *et al*., 2013; Weissteiner *et al*., 2016), together with site-level monitoring for finetuning. Below, we examine how riparian-zone management can be improved with mixed approaches.

Challenges in the global monitoring of riparian forests

The mapping of riparian forests at the global scale has two purposes: 1) to obtain a global overview of riparian forests and current dynamics; and 2) to provide data and methods to enable national forest agencies to monitor change in riparian forests, validate data and act on information.

The key questions for establishing a global indicator for riparian forests are as follows:

- At what scale do riparian forests need to be mapped – is there a minimum mapping unit? This is crucial at the global scale (see photo below).
- Is there a suitable classification scheme for riparian forests that can be mapped at a global scale?
- What parameters are to be extracted (e.g. the area of riparian forests, change, and change in the interface with anthropogenic zones)?
- Are accumulated totals of each class of riparian forests required? It would also be necessary to distinguish between natural changes (due to river flow) and anthropogenic changes. Can a global product respond to these requirements?
- How can a riparian forest be "detected", mapped and validated?



This derived product – shadow index (Rikimaru, Roy and Miyatake, 2002) – from the Sentinel-2 satellite highlights woody vegetation (forests) along the river network, which appear brighter. The darker areas are savannas and agriculture. A large riparian forest can be seen on the lower right corner of the image.

Official definitions of riparian forest vary depending on national environmental laws and goals. Definitions of riparian zones in the literature for large-area mapping are often based on a "buffer distance" from a watercourse (often between 10 m and 200 m) (Broadmeadow and Nisbet, 2004; De Oliveira Ramos and dos Anjos, 2014). The challenge for global monitoring is to set a clear definition without oversimplifying. The legal definition and functionality of such forests varies between ecosystems and countries, making a coherent global approach difficult. A two-tier system may be considered: a general global assessment that provides an overview, and a nationally relevant database to enable the development of appropriate management strategies. The two tiers can be compatible, with national monitoring systems nesting within the global one.

Similarly, there are difficulties in monitoring different ecosystems using the same methodology. This is illustrated by work in the Democratic Republic of the Congo (see the case study on page 27), in which two ecozones were reviewed – one (equatorial) with full forest cover and the other (subtropical) in a forest–savannah ecosystem (see Figure 2.11 on page 27).

Remote sensing as a tool for monitoring riparian forests

Remote sensing provides a synoptic tool for monitoring land cover and land-cover change over large and often inaccessible areas. When appropriately employed, the methods are robust and repeatable, giving a homogeneous product whereby quantitative measures can be compared across countries and regions. The georeferenced results of image processing can rapidly be ingested into a GIS to produce maps and statistics for land management and modelling scenarios. A number of major institutions (e.g. FAO, Brazil's Instituto Nacional de Pesquisas Espaciais, the European Commission Joint Research Centre and the University of Maryland) have been using remote sensing for many years to produce forest-distribution and forest-change maps and statistics.

There are two components to mapping riparian forests: mapping forest cover (and change); and identifying riparian forests within the forest-cover layer. Various datasets, methods and tools are available to implement a globally valid, locally relevant monitoring system.

For riparian areas in most parts of the world, images are unavailable at the necessary resolution for adequate monitoring. Thus, riparian-forest monitoring and management efforts that rely on satellite-based remote sensing are limited and must be combined with more accurate methodologies, such as remote sensing data obtained through the use of drones, and field monitoring. Box 2.3 and Box 2.4 present examples of the use of both remote sensing technology and field methods.

BOX 2.3

The Blue Targeting Tool for the rapid assessment of riparian habitat

The Blue Targeting Tool (BTT) is an example of how countries have started to implement tools that can easily be applied by any citizen to improve the management of riparian zones. These ground-level initiatives can complement government-led remote sensing-based approaches. The result is the more comprehensive monitoring and management of riparian zones and increased awareness of forest and water resources among citizens and industries and greater participation in their management.

The BTT was developed by the World Wide Fund for Nature (WWF) and Swedish forest owners' associations with the aim of including water management considerations in forest planning. The target audience comprises private and smallholder forest owners and managers.

The BTT was developed initially for small streams (10 m wide) in boreal and Scandinavian conditions (Henrikson, 2018). It consists of a scorecard-type survey that can be applied to stream sections and which requires little technical knowledge to complete. The survey evaluates four key aspects of a stream section: 1) conservation values; 2) impact; 3) sensitivity; and 4) added values (Henrikson, 2018). Based on the score obtained,

Continued ...

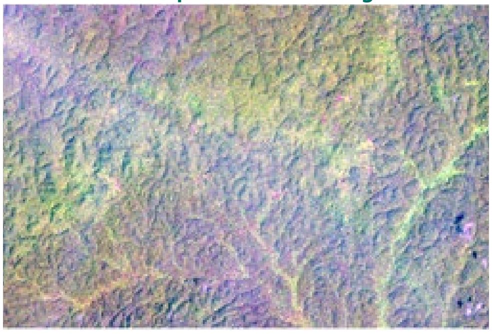
the BTT ranks stream sections into "blue target classes", which set out the actions needed with respect to, for example, the width of the riparian area, the use of protection measures and the management of stream-adjacent forests (Henrikson, 2018).

The BTT is supported by an enabling environment and platforms developed over many years. The forest sector has traditionally influenced forest management, policy and legislative actions in Sweden (Lindahl *et al.*, 2017). Policy has also evolved, resulting in a model in which production, the environment and conservation have the same weight, with private actors bearing much of the responsibility for finding this balance in management. Sweden's 1993 review of its Forestry Act contributed to a new wave of restoration efforts focused on landscape-scale management challenges, including forest management that takes into account water resources and considers the importance of multistakeholder participatory processes. The need for this was reinforced by the European Union's Water Framework Directive, which recognizes the role of the forest sector in water management and the need for further measures (Eriksson *et al.*, 2018).

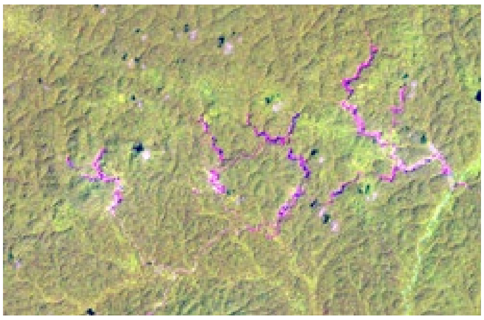
The successful implementation of the BTT in Sweden has led to its adaptation and implementation in other countries. The European Union's Interreg project, Water Management in Baltic Forests (WAMBAF), started in 2016 with the aim of reducing the export of nutrients and pollutants from forests to streams, lakes and the Baltic Sea. The project also set out to improve knowledge and coordination among Baltic countries, agencies and other stakeholders and to create efficient tools for managing riparian forests, forest drainage and beaver activity (Interreg Baltic Sea Region, 2020). The BTT was included in the project as a tool for managing riparian forests. The project provided training to test the BTT and other tools, involving more than 600 people, including representatives of private and state-owned forest enterprises, planners, landowners, hunters, authorities, non-governmental organizations and scientists (Swedish Forestry Agency, 2020). Demonstration areas were set up, and the BTT has now been adapted and translated for implementation in Finland, Latvia, Lithuania and Poland (WAMBAF, 2020). A new follow-up project, the WAMBAF Toolbox, aims to scale up the use of these tools. The BTT is being adapted for use in other ecosystems, including boreal forests in the Russian Federation and tropical forests in Brazil (Taniwaki *et al.*, 2018).

Using remote sensing to assess change in riparian forests. Various activities – such as industrial mining (Figure 2.4), hydroelectricity projects, small-scale agricultural expansion and large-scale agricultural projects – can cause changes in riparian forests, with impacts on, for example, forest cover, water flux and water quality. Many such activities bring national and local economic benefits; nevertheless, it is important to document the changes they cause in riparian forests, as well as forest–water relationships more generally, and to monitor and, if necessary, take steps to mitigate impacts on resource quality and to support forest and water governance.

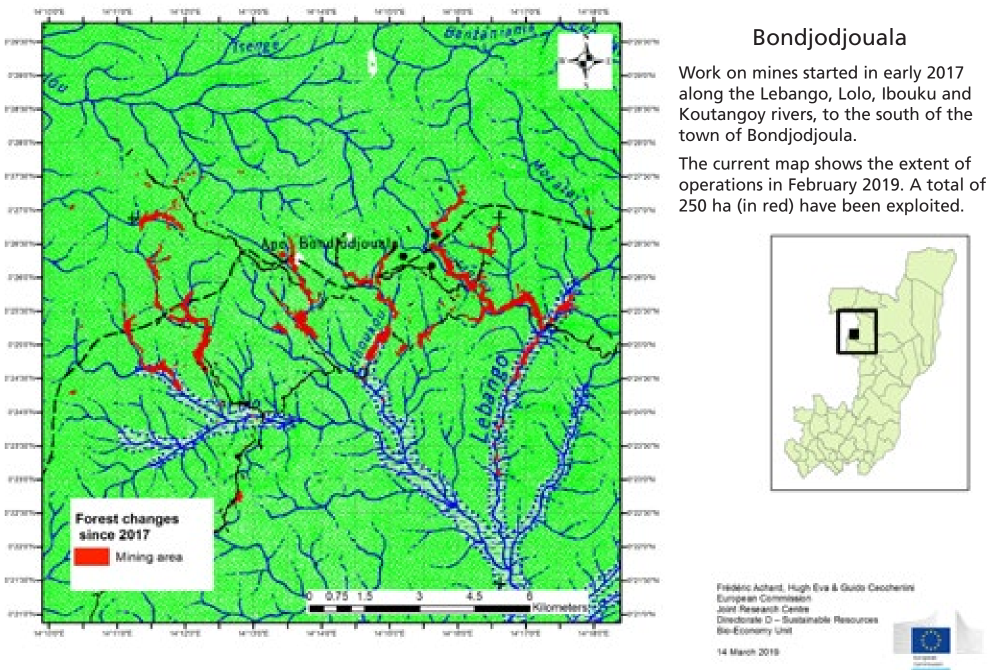
FIGURE 2.4
Sentinel-2 optical data showing the development of mining along a river network in the north of the Republic of the Congo



a) Sentinel-2 image (2016) before mining development



b) Sentinel-2 image (2018) showing mining areas along rivers



c) Cartographic product – approximately 250 ha of riparian forests along 25 km of river course was removed; the change was mapped using image segmentation and superimposed on the topographic map

Source: Eva *et al.* (2020).

BOX 2.4

Riparian zones: where green and blue networks meet

The European Union's Riparian Zones initiative was carried out in 2016 to identify and map riparian zones across the (then) 28 European Union countries, plus some cooperating countries (Figure 2.5 shows an example). The initiative was based on a methodology developed at the European Commission Joint Research Centre (JRC) (Clerici *et al.*, 2011; 2013) and relied on a set of databases – such as EU-HYDRO, EU-DEM, JRC Flood Hazard Risk, Corine Land Cover and the High Resolution Forests Layer. These were combined in a complex spatial modelling approach based on fuzzy logic and object-based image analysis. The model ultimately was capable of delineating potential, observed and actual riparian zones. Given the extent of the area and its complexity, the product's level of detail is unprecedented.

For the present publication, the authors posed the question – "can we upscale from the Riparian Zones initiative?" (Clerici *et al.*, 2011; Weissteiner *et al.*, 2016). The initiative was carried out in a data-rich environment, which does not exist at the global level. Nevertheless, given the increasing availability of satellite-derived data and new image-processing techniques, it is now feasible to produce a global dataset using proxies to meet the requirements of the European Union approach.

FIGURE 2.5
An example of the modelled Riparian Zones product

Note: Permanent water is in blue, riparian zones are in green.

Source: Clerici *et al.* (2011).

Available databases for implementing riparian forest monitoring

Global datasets on river networks and derived products are available to support the distinguishing of riparian forests from upland forests (Pekel *et al.*, 2016), including the 3 Arc Seconds Digital Elevation Model derived from the National Aeronautics and Space Administration (NASA)'s Shuttle Radar Topography Mission. This model enables the creation of useful products such as flow accumulation, which defines the quantity of upstream area (measured in number of cells) draining into discrete downstream areas, which can be used to form the basis of riparian buffer zones in a river network (Box 2.5).

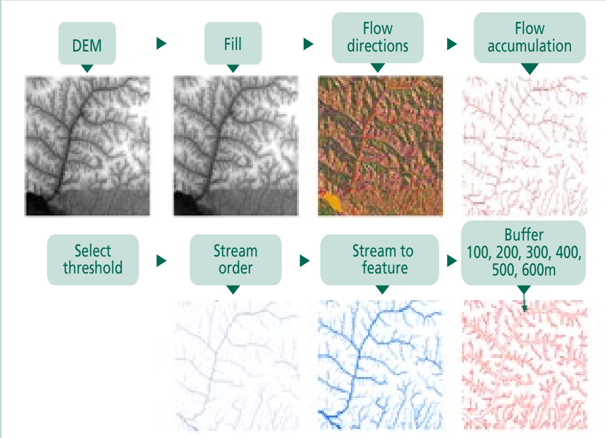
BOX 2.5

Potential methods for defining riparian zones

A potential method for defining riparian zones is to use the 90 m Shuttle Radar Topography Mission’s digital elevation model (DEM) to derive an estimated accumulated waterflow layer, as shown in Figure 2.6.

FIGURE 2.6

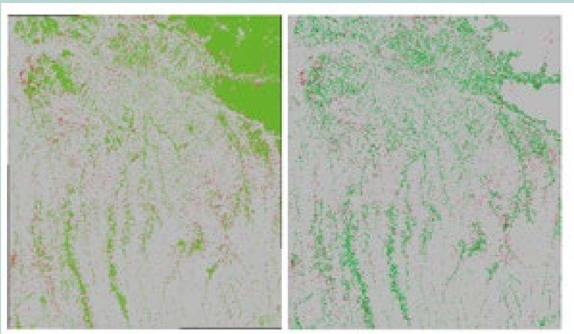
Process for identifying riparian buffer zones using accumulated waterflow



Another approach is to use global forest-change products such as Tropical Moist Forest with water masks (Pekel *et al.*, 2016) and to apply fragmentation algorithms to separate core blocks from gallery forests (Figure 2.7). The water masks are used to restrict the processing to areas within water-affected areas. This approach returns improved results with more precise locally produced forest and water masks.

FIGURE 2.7

Tropical Moist Forest product (original and after fragmentation analysis)



Note: On the right-side image, riparian forest have been extracted (green) and changes from forest to non-forest are showing in red

Source: Pekel *et al.* (2016).

Remote sensing analysis using suitable indices supported by image segmentation can generate an adequate riparian layer in some ecosystems. The databases generated in this way need to be assessed to ensure the robust delineation of riparian forests for various ecosystems. Similarly, morphological spatial pattern analysis (MSPA) (Soille and Vogt, 2009) generates maps and statistics on patch size and connectivity using input forest base maps. Although developed to support ecological studies on species distributions and movements, MSPA is useful for highlighting forest patterns that can also help in discriminating riparian forests. Online and stand-alone tools are available for this approach.

Off-the-shelf forest databases and available remote sensing images (tier 1).[5] One off-the-shelf database on forest cover is Global Forest Change, which gives percentage tree cover and changes from 2000 at a resolution of 30 m (Hansen *et al*., 2013). A similar product with the same resolution and timescale is Tropical Moist Forest, although this only covers evergreen forest belts (Vancutsem and Achard, 2017). A global database of mangroves based on ALOS PALSAR and Landsat data is available[6] for the baseline year of 2010 (Bunting *et al*., 2018), with change from this baseline for six epochs between 1996 and 2016. Annual maps derived from this database are planned from 2018 onward.

Pre-processed satellite images from the Landsat and Sentinel satellites, open-access images of which are available either as downloads or processed online, can be used to create forest-cover and forest-cover-change maps for any selected area using single-date images in an appropriate season or time composite. These medium-resolution (10–30 m) images are suitable for mapping riparian forests at global scales. Global coverage now exists for VHR (5 m) image data, which can be used to validate maps derived from medium-resolution satellites. Open-source tools are available – both stand-alone (e.g. IMPACT – Simonetti, Marelli and Eva, 2015) and online (e.g. SEPAL) – that enable users to process satellite images to cloud-free mosaics and maps and to extract statistics and validate products using finer-scale VHR satellite data (Box 2.6).

BOX 2.6

Very-high-resolution satellite data for product validation

Commercial satellite companies have started putting in place constellations of very-high-resolution (VHR) satellites capable of providing daily near-global data coverage, such as RapidEye (5 m) and Planet (3 m). The global coverage means that, although wall-to-wall mapping with such data remains a challenge due to data volume and cost, statistical sampling schemes can be employed for validation purposes. Norway's International Climate and Forest Initiative recently entered into a contract with KSAT, Airbus and Planet to provide universal access to high-resolution satellite monitoring in the tropics to support efforts to reduce tropical deforestation. New cloud-free mosaics from Planet data with a spatial resolution of 3 m will be available each month, free of charge for two years. Historical archives (from 2015 onwards) will also be available, covering all tropical countries where deforestation and forest degradation are occurring.

This dataset, which will be accessible through FAO's cloud-computing open-source SEPAL platform, will complement near-real-time alert systems to enable the precise validation of deforestation and degradation in riparian forests.

Such VHR data are valuable for validating rapid changes in landscapes. Figure 2.8 shows riparian forest in the Democratic Republic of the Congo in May 2019, October 2019 and March 2020. A new clearing in the forest (detected automatically by the Tropical Moist Forest algorithm) is visible in October 2019, but vegetation regrowth has largely obscured this clearing by March 2020. This example shows the need for high-cadence imagery to validate automatically detected tree-cover disturbances, even in very localized

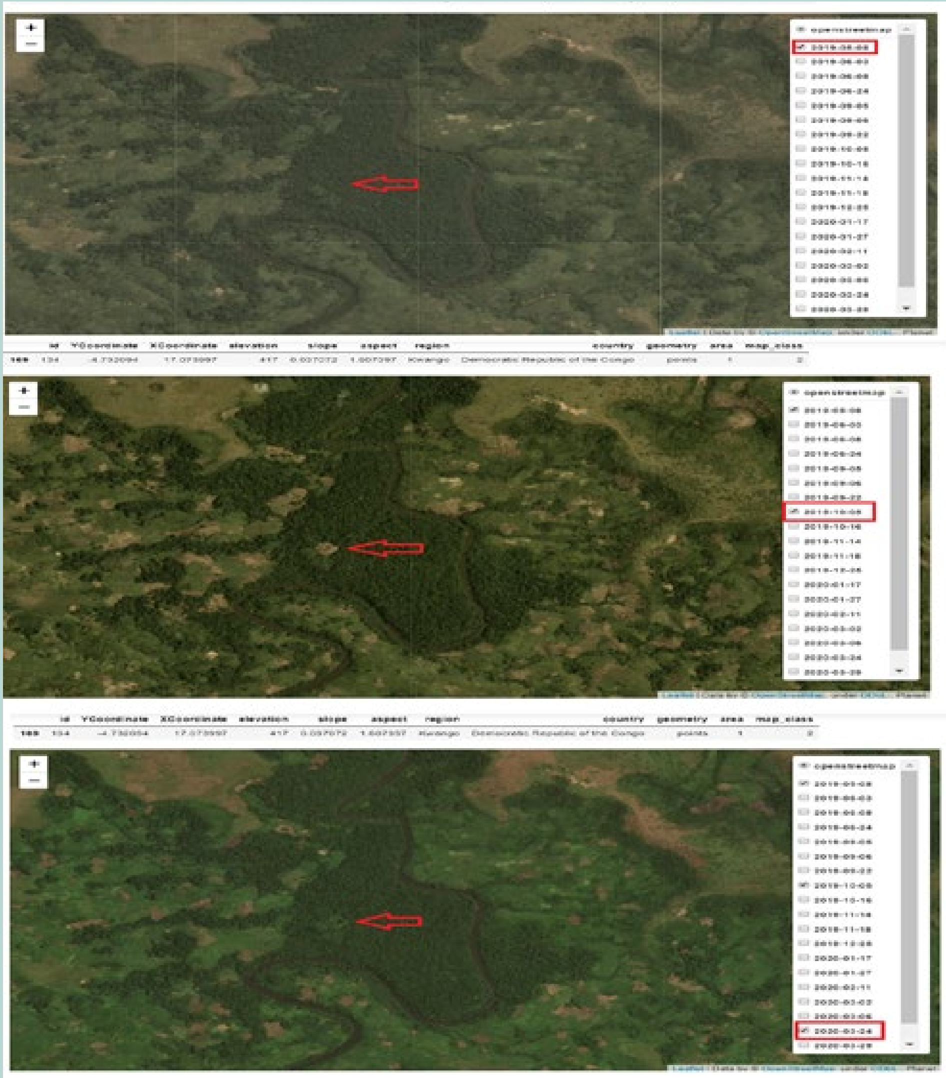
[5] The Intergovernmental Panel on Climate Change has classified the methodological approaches in three tiers according to the quantity of information required and the degree of analytical complexity (IPCC, 2006).

[6] www.globalmangrovewatch.org

ecosystems such as riparian forests. High-resolution data need field validation to ensure quality as well as acceptance by national forest agencies.

FIGURE 2.8

Change in riparian forest cover at a site in the Democratic Republic of the Congo, May 2019–March 2020



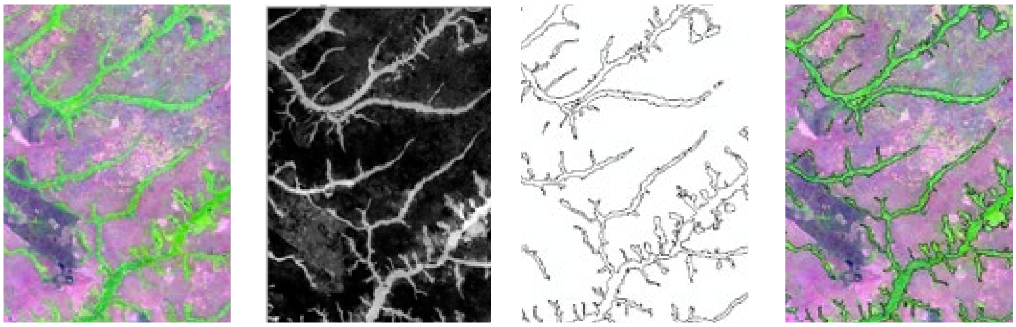
Note: Top – May 2019; middle – October 2019; and bottom – March 2020.

Source: www.nicfi.no/current/new-satellite-images-to-allow-anyone-anywhere-to-monitor-tropical-deforestation

Methods for processing remote sensing images for riparian forests (tier 2). Most of the literature on the mapping of riparian forests is limited to North America and Europe (e.g. Klemas, 2014; Clerici *et al*., 2011). A recent study of 428 peer-reviewed papers on the mapping of riparian forests with remote sensing found that 79 percent focused on the Northern Hemisphere and 14 percent focused on tropical and subtropical ecosystems (Huylenbroeck *et al*., 2020), the remaining studies being in tundra and desertic ecosystems. The remote sensing-based mapping of mangroves in the tropics is more studied, with efforts using optical sensors, synthetic-aperture radar and a combination of the two (Kuenzer *et al.*, 2011; Bunting *et al*., 2018; Thomas *et al*., 2018). For optical instruments, a wide range of techniques (e.g. spectral indices, supervised and unsupervised classifications, and decision-tree classifiers) and sensors has been employed for riparian forest mapping, depending on the scale and extent of the study area and available imagery (see the review by Huylenbroeck *et al*., 2020).

Data extraction can be done at the pixel level or using image segmentation, based either on band reflectance or derived indices, which enables the use of minimum mapping units (Raši *et al*., 2011). Pixel-based classifiers, although efficient, tend to create a "salt and pepper" effect that needs to be removed using filtering. In savannah zones, segmentation has the potential advantage of providing a strong contrast for riparian forests (Figure 2.9). In the equatorial zone, however, where full forest cover is more common, the detection of riparian forests requires a combination of forest detection and supplementary sources to delimit riparian areas.

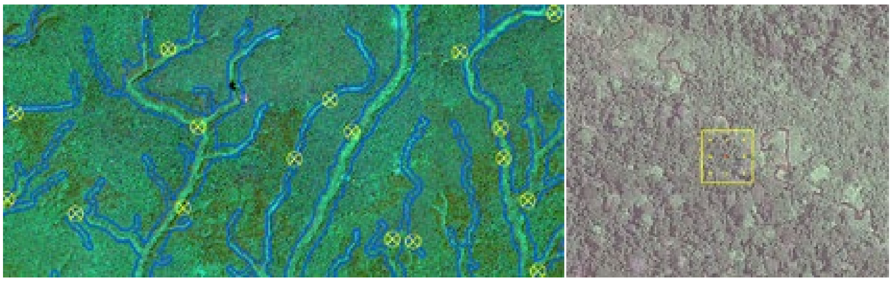
FIGURE 2.9
Example of the use of spectral indices in conjunction with segmentation to highlight riparian forests in the forest–savannah domain



Sentinel-2 image of riparian forest in the forest–savannah domain | Shadow index of Sentinel-2 image | Segmentation of shadow index | Segmentation overlaid on original image

Validation. To maintain quality and confidence in results, it is essential to validate remote sensing products using finer-spatial-resolution data and, where possible, field inventories (Olofsson *et al*., 2013. Sampling schemes such as stratified random samples of validation points can be generated for target classes (e.g. riparian forest area and change) using SEPAL or other tools. The points generated can then be reviewed in visualization tools such as Open Foris Collect Earth using VHR data (Figure 2.10). Where these data are not available, mosaics from Landsat and Sentinel-2 can be employed as surrogate confirmation. Crucial for the validation exercise is determining a validation mapping unit (point, area) and criteria to enable interpreters to obtain consistent results. The interpretation of forest and forest change poses few problems, but the concept of confirming whether a forest is riparian is more challenging. The results of the validation serve not only to build confidence in the product; they can also be used in a correction phase to adjust statistics on riparian forest area and change (Tyukavina *et al*., 2013). Several countries have 3-m-resolution optical Planet data available.

FIGURE 2.10
Example of how tools such as SEPAL and Collect Earth can be used to validate remote sensing observations

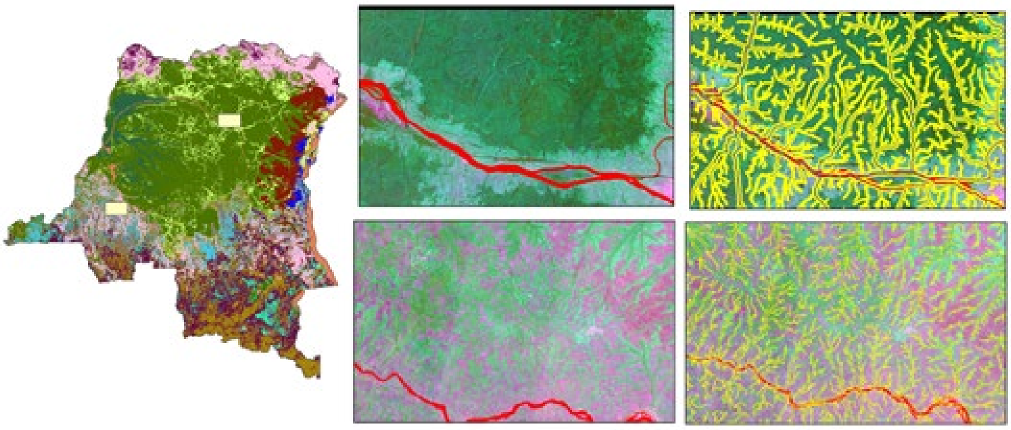


Notes: A set of validation points (left) generated in SEPAL using the 30 m Tropical Moist Forest product, and a box interpretation (right) in Collect Earth. Because riparian forests are a narrow target, the co-location of validation points generated from a medium-resolution dataset (e.g. Landsat at 30 m resolution) and fine-resolution validation images may pose difficulties.

Case study: mapping riparian forests in the Democratic Republic of the Congo

Figure 2.11, which shows two ecosystem types (moist tropical forest and savannah) in the Democratic Republic of the Congo, demonstrates the diverse nature and challenges of monitoring riparian zones. Full-canopy forest presents challenges in discriminating riparian from upland forests. The savannah ecosystem type is less problematic because riparian forests are usually easily distinguished from surrounding grasslands; on the other hand, they are often surrounded by areas of shifting cultivation, which may include shrub forms (e.g. cassava) that can be difficult to differentiate from other woody species.

FIGURE 2.11
Riparian zones in the closed-forest and savannah ecosystems, Democratic Republic of the Congo

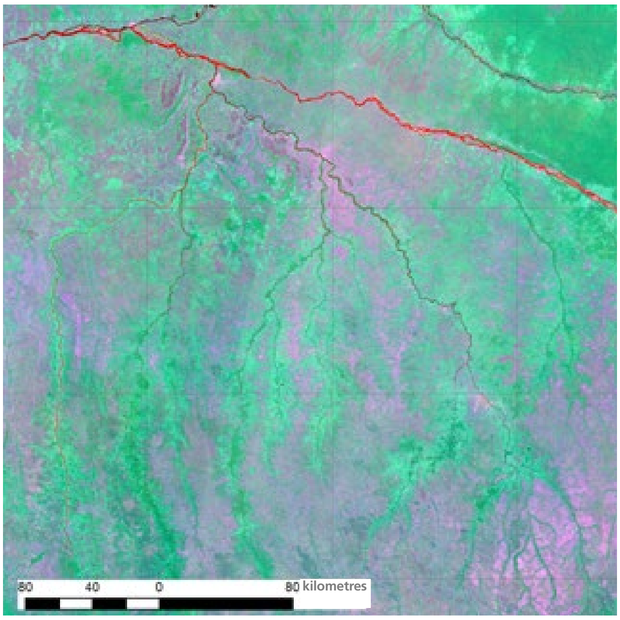


Note: The map on the left shows the location of the two study areas; the top-centre Sentinel-2 image shows closed-canopy forest, and the bottom-centre Sentinel-2 image shows savannah. The images on the right show the same locations overlaid with vectors of riparian forests derived from accumulated waterflow.

A review of savannah riparian forests. The savannah study area (see Figure 2.11) encompasses 157 620 km², of which the majority is a savannah–gallery forest complex (Figure 2.12). To the northeast of the study area, a large area (1 400 000 ha) of dense forest dominates, with logging activities present – this region was removed from the analysis because virtually no changes to forest cover were occurring along rivers within this intact forest area.

FIGURE 2.12

Study area in southern Democratic Republic of the Congo at Bandunu, showing intact forests in the northeast and gallery forests in savannah to the south of the Kasai River



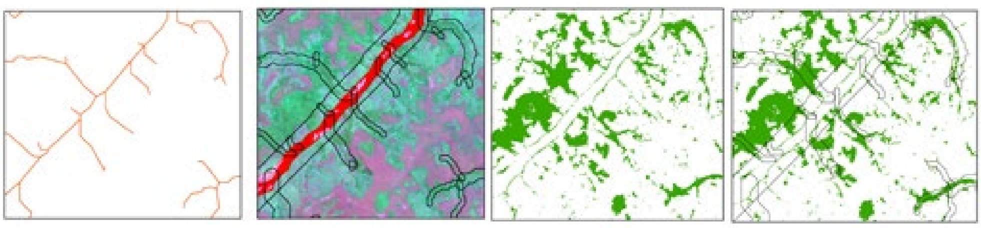
Source: Sentinel-2 false colour composite (SWIR, NIR, RED) courtesy of the Copernicus programme.

Forest-cover data were obtained from the Tropical Moist Forest database (Vancutsem and Achard, 2016), which collates information from the Landsat archive spanning 37 years. The data were assigned to three classes: 1) forest; 2) non-forest; and 3) forest-cover change between December 2015 and December 2018. An assessment showed that, although the product is acceptable for delineating riparian forests, estimates of forest-cover change in this ecozone are far less reliable due to the small scale of changes around the gallery forests. A certain amount of "noise" (i.e. false change detections) were observed in the data.

To separate riparian from upland forests in this ecozone, a simple approach was used in which a stream-order mask was created from the Arc 3 second DEM with a buffer of 200 m. This was cross-tabulated with the Tropical Moist Forest map to calculate the area of, and changes in, riparian forest (Figure 2.13). Riparian forests accounted for 80 percent of the 1 845 500 ha of forest in the savannah domain, but 48 600 ha was being lost per year – a deforestation rate of just over 3 percent. This high rate is due to a combination of the relatively small area of riparian forests, their open access, and potentially their naturally irrigated soils. The results can be disaggregated by stream order.

FIGURE 2.13

Combining river networks with forest data, savannah, Democratic Republic of the Congo



Notes: From left to right: 1) the river network; 2) the stream order derived from the river network, overlaid on Sentinel-2 data; 3) the Tropical Moist Forest map; and 4) forest lying in the stream-order layer classed as riparian.

A product such as the Tropical Moist Forest map can highlight hotspots of deforestation and where these are occurring in the stream order. By combining historical deforestation and field data, such maps can help target suitable areas for restoration.

Gaps in the monitoring of riparian forests through remote sensing

Limitations of data and systems. The low spatial resolution of water datasets can limit the accuracy of gallery-forest detection. As seen in the above case study of savannah in the Democratic Republic of the Congo, stream-order masks generated from a DEM do not always correspond with actual forest cover.

The spatial and temporal resolution of remotely sensed forest datasets can limit their accuracy in detecting rapid change in forest cover, which can also be hindered by persistent cloud cover (especially in the tropics). Soil moisture maps can be derived and regularly updated from remotely sensed datasets on forest cover (Ali *et al*., 2015), but these are limited to areas outside the forest zone because the publicly available C-band sensors (Sentinel-1) cannot "see" through tree canopies (Frolking *et al*., 2009).

Masks of national riparian network. Few national databases exist of riparian networks. Global datasets, notably DEMs, need to be reviewed to produce artefact-free layers that can be used at the national level and edited to provide the required information. As shown in the Democratic Republic of the Congo case study, riparian forests in savannah ecosystems stand out against grasslands and agricultural land use but are spectrally inseparable from upland forests in full-forest-cover environments. This means that rulesets based on elevation and buffering need to be developed and tested.

Easily available data of historical and present forest cover. Although two available databases (Global Forest Change from 2000 and Tropical Moist Forests from 1987) contain historical Landsat-based data on tropical forest cover, both have drawbacks, especially in dry ecosystems. Pre-2000 data are unavailable for many regions due to cloud cover and the low number of acquisitions from outside then-existing receiving stations.

Automatic alert system to detect potential changes. The Global Land Analysis and Discovery lab at the University of Maryland (Hansen *et al*., 2016), supported by Global Forest Watch, is a Landsat-based alert system providing weekly alerts on tree-cover loss at 30-m resolution in the tropical and subtropical belt.[7] The system is integrated into SEPAL and can be complemented by tailored near-real-time alert systems based on various time-series data analysis approaches (e.g. Breaks For Additive Season and Trend – BFAST, Continuous Change Detection and Classification – CCDC, and Bayesian approach to combine multiple time series for near-real-time forest-change detection – BAYTS).

Tailormade interface to inspect, verify and confirm changes. Effective monitoring requires an interface to enable the rapid and robust verification of ongoing change, which could be based on the Collect Earth interface. Key to the success of such an interface is direct access to recent high-resolution (5 m) data. Given the persistent cloud cover in many regions, such a system should also provide access to all-weather SAR data. Currently, Sentinel-1 data are available, but at a lower-than-optimal resolution. The forthcoming NASA–Indian Space Research Organisation Synthetic Aperture Radar satellite (Stavros *et al*., 2018), due to be launched in 2022 equipped with L band (suitable for forest monitoring), will acquire data 4–6 times per month at a resolution of 3–10 m and will also improve the global DEM database.

Geographic information system analysis to assess impacts on waterflow. To support national agencies in their efforts to monitor and maintain riparian forests, a

[7] This alert system will soon be complemented by similar products from the Copernicus Sentinel-1 and Sentinel-2 satellites.

suite of GIS modules could be developed to enable the rapid analysis of remote sensing data and the production of cartographic alerts and statistical products. These could also provide information on the relative potential impacts of change in riparian forests along watercourses, from headwaters to estuaries, and help in prioritizing restoration efforts. A coordinated effort among experts would be required to refine the modules and their implementation in open-source software and to provide accompanying examples and tutorials.

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:

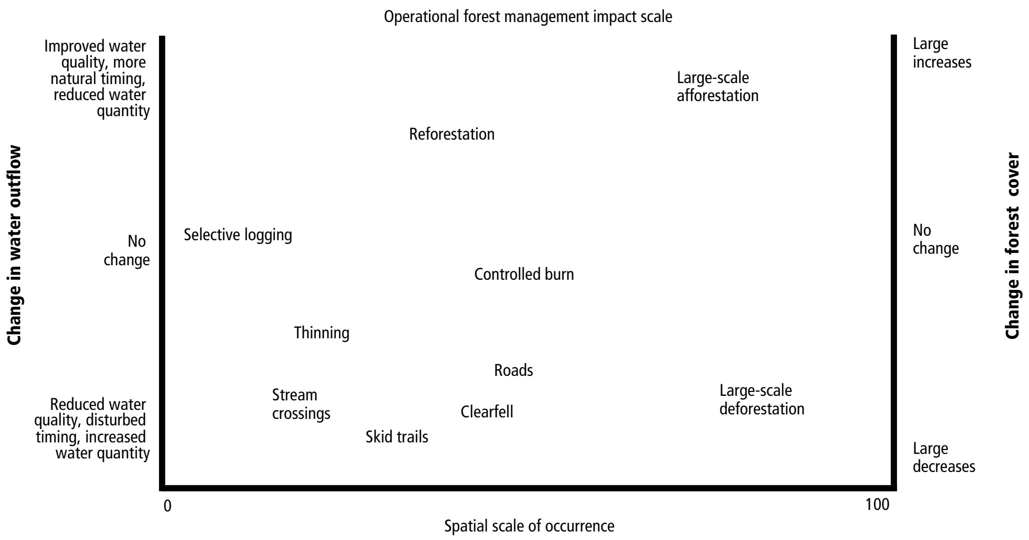
- identifying, describing and mapping all streams, wetlands and other water features, as well as slopes and soil types, and taking these fully into account in road planning;
- pre-logging planning of roads, skid trails and landings to provide access to the forest and the trees to be harvested while minimizing 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).

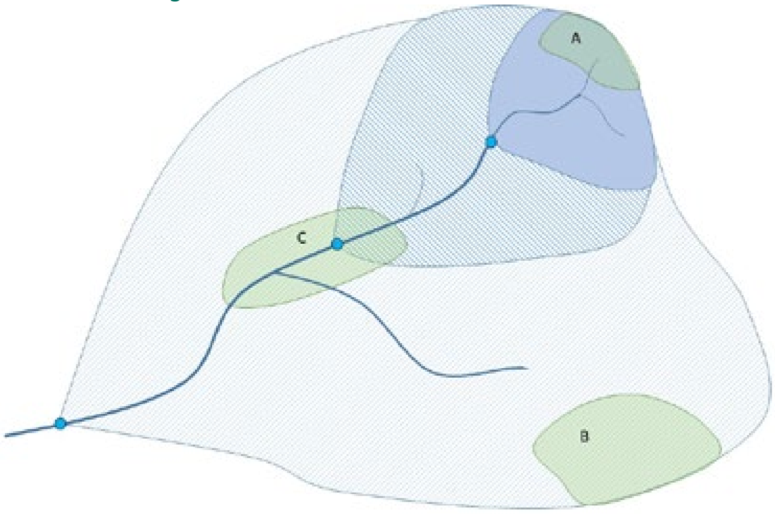
WATERSHED-BASED FOREST MANAGEMENT

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.

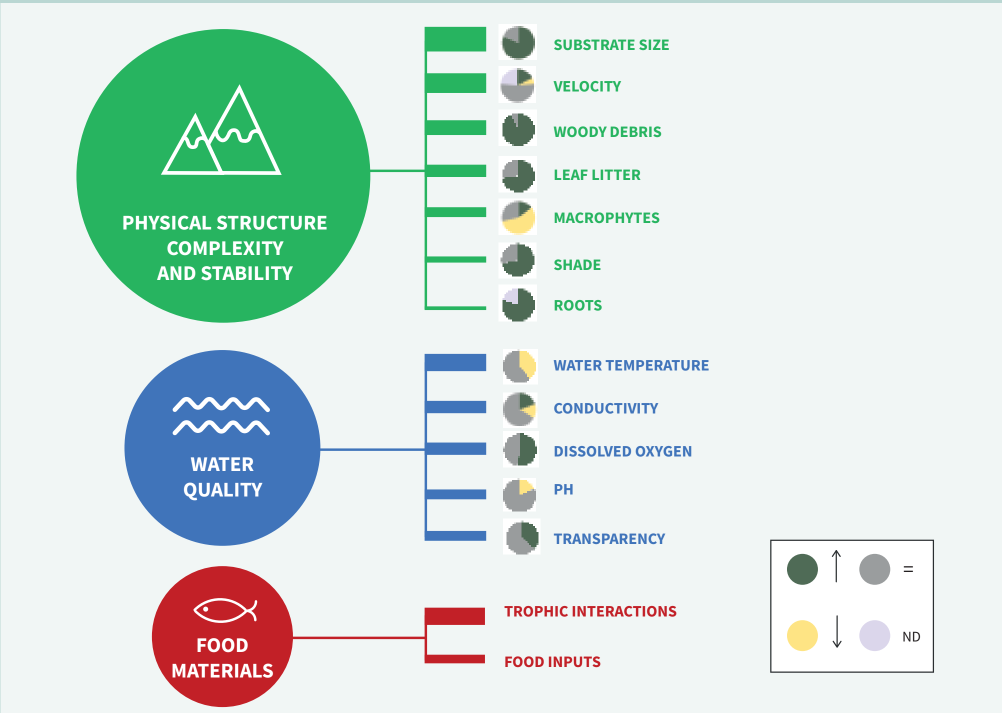
Healthy aquatic biodiversity and fisheries require a predictable supply of clean water. High levels of sediment, for example, interfere with the gills and respiration of aquatic animals, particularly at younger life stages and in sensitive species. Sediments can also physically smother eggs and juveniles 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).

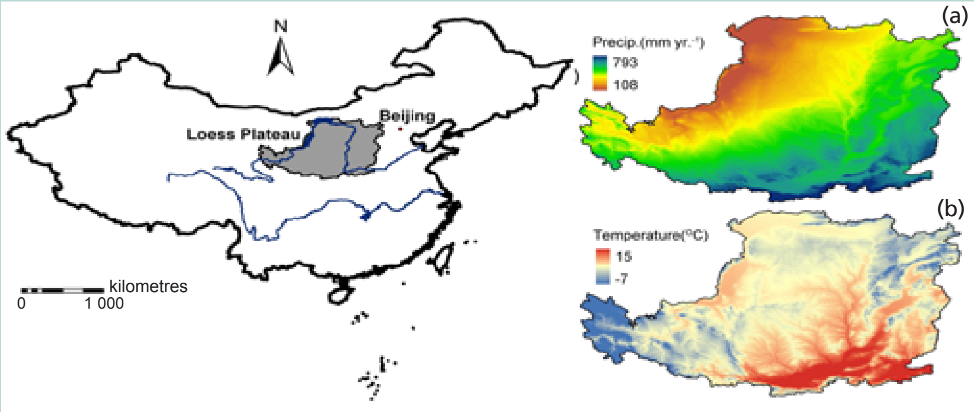
Today, China has the world's largest area of forest plantations – approximately 84 million ha, which is more than one-quarter of the world total of 293 million ha (FAO, 2020a). The total forest cover in China grew from about 11 percent of the land area in the 1980s to 23 percent in 2020 (FAO, 2020a). 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.

4 Valuing water from forests

Key points

- The global provision of water services decreased by nearly USD 10 trillion per year between 1997 and 2011.
- The valuation of ecosystem services is the starting point for managing forests and all the benefits they provide.
- Several methodologies have been put in place for recognizing the value of the many ecosystem services provided by forests. The value of an ecosystem service can be derived from information provided by market transactions relating directly or indirectly to that ecosystem service, or from hypothetical markets that may be created to elicit values.
- Payments for watershed services (PWS) are a promising mechanism for benefit-sharing and cooperation among the forest and water sectors, especially in the absence of legislative frameworks or functioning local governance.
- PWS should be seen as part of a broader process of local participatory governance rather than as a market-based alternative to ineffective government or community management.
- Networks and collaborative approaches at the local level are a common characteristic in successful PWS schemes, in which regulators, private companies, local authorities and technical and civil-society organizations share their expertise – through matched funding – to deliver high-level forest watershed schemes.
- The two most common PWS schemes in the forest–water domain are water fees (utility-led) and multiple-benefit partnerships. Schemes that apply fees for water use are usually based on a defined normative background. National governments may incentivize these schemes through appropriate regulations; examples are provided.

Well-informed management and policy decisions on the forest–water nexus require an understanding of the true value of forest–water relationships, trade-offs and synergies. Recognition has increased in recent decades of the importance of forests and trees in the provision of ecosystem services such as biodiversity conservation, carbon sequestration and water provision. Estimating the value of these services in economic terms helps bring them into political discourses and planning, although such valuations are difficult.

Payment schemes for ecosystem services are becoming more prevalent. Those for water-related services, which constitute the largest and most rapidly growing type of scheme, increased in value from USD 6.7 billion in 2009 to USD 24.7 billion (in 62 countries) in 2015 (Salzman *et al*., 2018).

Land and water management practices play significant roles 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 key, especially within the framework of policies related to climate change (e.g. those promoting carbon sequestration in standing forests and harvested wood products), bioeconomy (in which the aim is to decarbonize economies by substituting fossil-fuel-based materials with bio-based materials), and nature conservation (e.g. forest ecosystem restoration for biodiversity and multiple other benefits) – policies that all interact.

This chapter explores the valuation of forest–water ecosystem services as well as trade-offs and synergies and how to manage these.

ESTIMATING THE VALUE OF FOREST–WATER ECOSYSTEM SERVICES

How much are forest–water ecosystem services worth?

An impressive database has been developed comprising 1 350 case studies estimating the value of 22 ecosystem services in several biomes (van der Ploeg, de Groot and Wang, 2010) (see Box 4.1 for two other databases).[12] Using this database, a mean value was estimated for each ecosystem service per ha of biome (de Groot *et al*., 2012) and aggregated to obtain a global estimate of the value of forests for ecosystem services, with values converted to a common set of units (Costanza *et al*., 2014). Table 4.1 presents the results for water services (in 2020 international USD), where it can be seen that coastal wetlands – mangroves and tidal marshes – are valued much more highly per unit area than other forests. The table shows that, at the global level, the annual value of forest-related water services decreased by nearly USD 10 trillion between 1997 and 2011 due to declines in forest area. The estimated values assume a linear link between forest loss and services loss. This is a simplification as there may be different (i.e. non linear) relationships at work in reality.

TABLE 4.1
Estimated average and aggregate values of various water services, selected biomes, 1997 and 2011

Biome	Total land area		Value of ecosystem service								
	1997	2011	Water regulation	Water supply	Erosion control	Waste treatment	Habitat	Cultural	Total	Total (area in 1997)	Total (area in 2011)
	(million ha)		(2020 USD per ha per year)							(billion USD per year)	
Tropical forests	1 900	1 258	90	34	419	149	49	1 082	1 826	3 469	2 297
Temperate/ boreal forests	2 955	3 003	158	238	51	149	1 073	1 232	2 902	8 575	8 714
Tidal marshes/ mangroves	165	128	6 661	1 515	4 891	201 825	21 335	2 730	238 958	39 427	30 587
Total	**5 020**	**4 389**								**50 853**	**40 971**

Source: Adapted from Costanza *et al.* (2014).

Valuation is only the first step in the integrated analysis of the contributions of forest ecosystem services to human well-being. Several other actions should follow, as described below.

[12] The complete database is available at www.es-partnership.org/services/data-knowledge-sharing/ecosystem-service-valuation-database. It provides useful insights into the monetary value of specific ecosystem types and other spatially defined areas (e.g. parks, watersheds and regions) and can also help in analysing the effects of different land-use options using both empirical research and value-transfer approaches; notwithstanding their limitations, the latter are an increasingly attractive option for policymakers with time and budget constraints (de Groot *et al.*, 2012).

BOX 4.1

Databases and tools on the valuation of ecosystem services

Environmental Valuation Reference Inventory. Once logged in, it is possible to navigate through geographical regions and methods to find relevant case studies, which are updated frequently. *www.evri.ca/en*

Envalue. This database (Morrison, Groenhout and Moore, 1995) enables users to explore case studies by method or ecosystem. It is useful for locating older (to 2002) studies. *http://environmentaltrust.nsw.gov.au/envalueapp*

InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs). This software provides a suite of models for mapping and valuing the ecosystem goods and services that sustain and fulfil human life. It makes use of a geographic information system and is relatively simple to use.

Method Navigator. This site guides you in choosing the best method through the selection of variables, providing a good starting point for navigating through the various valuation methods. *www.aboutvalues.net/method_navigator/policy_areas*

Practical tips for the valuation of ecosystem services

The valuation of ecosystem services is the starting point for managing forests and all the benefits they provide. To increase the impact of a valuation, consider the following aspects before valuation (Pierrot-Maitre, 2005):

- **The purpose of the analysis and how the results will be used.** Ecosystem service assessments are always part of larger decision-making processes that should end with the adoption of policies and market-based instruments that redress the imbalances highlighted by the valuation.
- **Budget and timeline.** Methods differ in their cost, but benefit transfer is usually considered the cheapest, and market-value methods are generally less expensive than demand-curve methods.
- **Most appropriate method.** This may depend partly on the budget but also on the ecosystem service to be valued and the values that characterize a particular ecosystem service. Each method has advantages and disadvantages, and these should be weighed carefully before a choice is made. See Masiero *et al.* (2019) and Chapter 5 of TEEB (2010) for more on the advantages and disadvantages of various valuation methods for forest ecosystem services.

Methodologies for estimating ecosystem services

The value of an ecosystem service can be derived from information provided by market transactions relating directly to that ecosystem service, but such information is frequently unavailable. Prices might also be derived from parallel market transactions associated indirectly with the good(s) to be valued. If both direct and indirect price information on ecosystem services is absent (Box 4.2), hypothetical markets may be created to elicit values (TEEB, 2010).

BOX 4.2

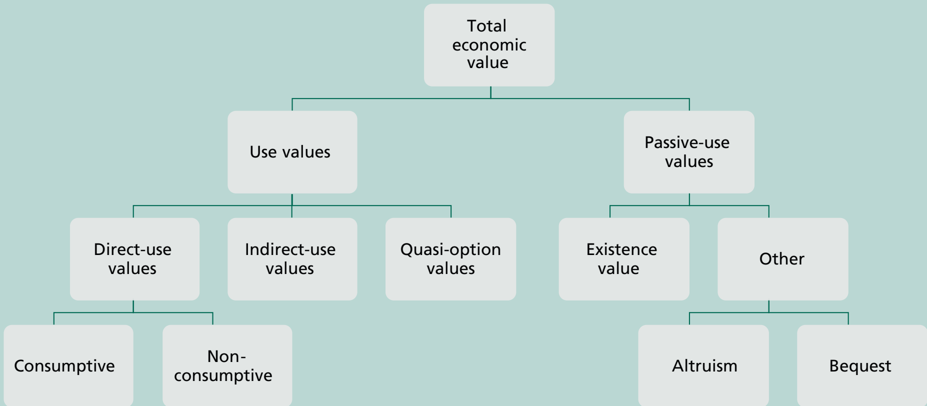
Total economic value

Economic values can be categorized broadly as either use or non-use (or passive-use) (Masiero *et al.*, 2019), and the sum of both provides the total economic value (Figure 4.1).

Use values may be direct or indirect. Direct-use values comprise those benefits derived from the actual direct use of an ecosystem (such as a forest that has an effect on water); they are usually distinguished as either consumptive (or extractive, such as the extraction of drinking water) or non-consumptive (or non-extractive, such as recreation activities). Indirect-use values refer to the benefits derived from an ecosystem's functions without direct interaction with it – such as protection against floods. Quasi-option values are those benefits derived from the option of directly or indirectly using forests in the future.

FIGURE 4.1

The components of total economic value



Source: Masiero *et al.* (2019).

Passive-use values, such as existence value, are values not associated with actual use and comprise the benefits derived from knowledge of the existence of an environmental characteristic, such as biodiversity. Other types of passive-use values include the benefits derived from placing a value on the conservation of a certain environmental feature on behalf of other people (altruism) and of future generations (bequest) (Masiero *et al.*, 2019). The set of relevant components of total economic value differs by ecosystem service: quasi-option, bequest and altruism values apply to all ecosystem services, whereas provisioning services are generally linked with direct use and regulating services are more linked with indirect use. Cultural services usually comprise all types of value (Masiero *et al.*, 2019), and supporting services are valued through various other categories of ecosystem service (Price, 2014). Each valuation method addresses a certain set of values and is therefore suitable for assessing specific ecosystem services (TEEB, 2010). For example, revealed-preference methods usually apply to use values and are therefore used to estimate services characterized by use (such as recreation). Stated-preference methods give information on both use and non-use values and are usually used for valuing biodiversity.

Market-value approaches. The market price represents the meeting point between supply and demand (being the amount at which the consumer/user is willing to buy and the supplier/producer is willing to sell). This price is an adequate representation of the value of ecosystem services with pre-existing markets, assuming that the market is not distorted (e.g. by monopoly power) and that therefore the price is freely attributed by the market. Market price might be an adequate mechanism for tariffs on drinking water; in most cases, however, market values do not exist for the ecosystem services provided specifically by forests.

Where there is no direct market, two methods may be applied.

1. **Opportunity cost.** This refers to the income that would be lost by choosing to deliver an ecosystem service (the object of the estimation) instead of another product or service with a market value. For example, opportunity cost can be used to quantify the amount that forest managers should be paid in compensation if they were required to follow specific management practices to improve water quality that mean they must forgo income they would otherwise receive (Masiero *et al*., 2019).
2. **Production function.** This refers to how much a given ecosystem service (e.g. a regulating service) contributes to the delivery of another service or commodity that is traded in an existing market (TEEB, 2010). For example, forests provide water-infiltration services and increase water availability for hydroelectricity, ultimately supporting an increase in the production of energy (Box 4.3).

BOX 4.3

Hydroelectricity production in Hubei Province, China

Basic data

Ecosystem services	Provisioning Improvement of in-stream water supply Hydroelectricity production
Valuation method	Production function
Area	Xingshan County, Hubei Province, China (231 600 ha)
Year	2000

Source: Guo, Xiao and Li (2000).

Forests may have substantial economic value for their waterflow-regulation services in local watersheds. Because of the distance between the ecosystem service at the source and the realization of its benefits, however, forests tend to get little recognition for their role. The objectives of a study by Guo, Xiao and Li (2000) were to:

- develop an integrated approach for valuing forests for waterflow regulation using simulation models and a geographic information system (several variables were used to model forest capacity in different combinations of vegetation types, soil types and slopes);
- estimate the economic value of waterflow regulation provided by forest ecosystems to increase the output of the Gezhouba hydroelectricity plant (a relatively small increase in waterflow in the Yangtze River would raise electricity production in the plant); and
- provide a model for economic compensation in which profit is distributed between the hydroelectricity plant and forest landowners by calculating the most efficient amount of water to be regulated and the corresponding benefit for landowners.

Table 4.2 shows that the estimated economic value of the waterflow regulation services of forests and other vegetation complexes in Xingshan County is USD 916 million per year. The model used to make this estimate also indicates how to identify the most efficient combination of water released and timber sold, enabling partnerships among actors to share the benefits of this ecosystem service.

TABLE 4.2
Total waterflow regulation by 90 types of vegetation–soil–slope complexes in the dry and rainy seasons, and its economic impact

Description	Unit	Dry period	Rainy period	Total (year)
Water released	million m^3	80.7		
Water retained	million m^3		868	
Flow increased	m^3 per second	10.4		
Flow decreased	m^3 per second		112	
Increase in the output of hydroelectricity plant	million kWh	27.4	13.0	40.4
Economic value	RMB million per year			5 050
	USD million per year			916

Source: Guo, Xiao and Li (2000).

Cost can also be used to estimate the value of an ecosystem service, with the value equal to the cost of producing (or reproducing) the service. **Replacement cost** refers to the cost of restoring a damaged asset to its original state or replacing it with artificial measures (TEEB, 2010). For example, the treatment costs incurred in the absence of the purification services provided by forests can be used to estimate the value of those services (Elias *et al*., 2014; Box 4.4).

BOX 4.4
Public water supply in Alabama, United States of America

Basic data

Ecosystem services	Provisioning Improvement of extractive water supply Drinking-water quality
Valuation method	Replacement cost
Area	Converse Reservoir watershed, Alabama, United States of America (31 600 ha)
Year	2010

Source: Elias *et al.* (2014).

Elias *et al.* (2014) estimated the economic value of the ecosystem service provided by a forested landscape in mitigating total organic carbon (TOC), a contaminant of drinking water. The study used robust hydrologic models to simulate watershed and reservoir nutrient processes under progressive urbanization scenarios to evaluate the effects of forest land conversion on reservoir TOC concentrations and therefore the cost of TOC removal during water treatment (i.e. replacement cost).

A simulated change from forest to urban land use caused an increase in monthly median predicted TOC concentrations at the source of water intake between May and October of 33–49 percent. Additional drinking-water treatment is necessary when raw water TOC concentration is greater than 2.7 milligrams per litre between May and October. Using 1992 data for pre-urbanized land use, the simulation indicated that drinking water needed to be treated with powdered activated carbon on 47 percent of days. Under simulated urbanization, the model indicated that drinking water needed continuous additional treatment. Table 4.3 shows that the cost of treatment increases substantially as urbanization spreads.

TABLE 4.3
Estimated increase in treatment cost due to change from baseline (forested) conditions to urban land use, Converse Reservoir, Alabama, between 1992 and 2004

USD per day (52 Km^2)	Volume of treated water registered	
	minimum	**maximum**
baseline (1992)	1100	1360
urbanized (2004)	5560	5920

Notes: Adjusted to 2020 United States dollars. A range of minimum and maximum volumes was maintained to account for the variability registered in the quantity of water treated, which in turn is dependent on annual rainfall.

Source: Adapted from Elias *et al.* (2014).

Although the results shown in Table 4.3 are specific to the Converse Reservoir, the methodology can be applied elsewhere to estimate values for the ecosystem services associated with various water-quality parameters.

Such studies can be useful for planning public interventions in which a fixed percentage of the income derived from a tariff is paid to forest owners who commit to sustainably managing forests (and therefore water).

Defensive expenditures are expenses incurred to avoid or reduce the effects of a negative externality or to reduce or compensate for damage arising from such an externality. For example, the money spent by coastal communities to upgrade their houses to protect against the increasing frequency and severity of cyclones and storm surges could be considered a defensive expenditure and thus used to estimate the protection service provided by mangroves (Masiero *et al.*, 2019). Box 4.5 provides an example of the damage-cost valuation method.

BOX 4.5

Flood damage mitigation in Manadia National Park, Madagascar

Basic data

Ecosystem services	Regulating Water damage mitigation Flood protection
Valuation method	Damage cost
Area	Watershed of the Vohitra River, Manadia National Park, Madagascar (26 800 ha)
Year	1997

Source: Kramer *et al.* (1997).

Tropical forests have a strong impact on flood dynamics. There is mounting concern that increasing rates of deforestation are causing greater flooding in the eastern half of Madagascar, where monsoon rains are particularly severe.

The aim of a study by Kramer *et al.* (1997) was to estimate the economic benefits of reduced flooding arising from the establishment of the Manadia National Park. The analysis followed a three-step method, evaluating:

1. Environmental quality (extent of flooding) and the human interventions (land-use practices, particularly deforestation) that affect it. Remote sensing techniques retraced deforestation patterns and hydrologic analysis identified the effects of deforestation.
2. Human uses of the environment (agriculture) and the dependence of people on environmental quality (intensity of flooding and damage). Several parameters were modelled – area, depth, duration, seasonality and frequency of inundation.
3. Changes in economic welfare because of a change in use of the environment (loss in producer surplus). The monetary value of the loss in producer surplus was estimated using an average price, net of production costs.

Table 4.4 shows the results of the study. It proved useful for demonstrating the full impact of establishing a protected area and the importance of keeping it over time. Without the protection of the park, the forests within its boundaries were projected to have disappeared within 46 years.

TABLE 4.4

Net present value of loss of agricultural yield over the life of the park due to low and high intensity flooding

	Minimum volume of water (flooding)	Maximum volume of water (flooding)
	USD	
Without park	83 127	1 090 982
With park	81 680	887 224
Difference	1 447	203 758

Note: Adjusted to 2020 United States dollars.

Demand-curve approaches. The demand-curve method, which has a long tradition in economics, is based on the estimation of hypothetical markets. It is useful for valuing ecosystem services when market values are unavailable and approaches based on benefits and costs are infeasible or impractical. The method involves inferring the value of a service (defined as a consumer's willingness to pay for it), either by observing behaviours ("revealed preferences") or by asking respondents to state hypothetical preferences ("stated preferences") (TEEB, 2010).

Revealed-preference techniques are based on the observation of individual choices in existing markets related to the ecosystem service being valued. Such parallel markets may be:

- The **expenses incurred** to reach a recreational site (i.e. travel cost). In this approach, the willingness to pay to visit a site is estimated based on the number of trips tourists make and their associated travel costs (Masiero *et al.*, 2019).
- The environmental attributes of marketed commodities such as houses (i.e. **hedonic pricing**). These attributes – for example the proximity of a house to a forested park – are reflected in the price of the commodity, and changes in the quality of such attributes influence the price in a way that can be assessed (TEEB, 2010).

Stated-preference methods establish that, when a parallel market cannot be found, it can be simulated through surveys about hypothetical changes in the provision of ecosystem services (TEEB, 2010). In particular, **contingent valuation** uses questionnaires to ask people how much they would be willing to pay to increase or enhance the provision of an ecosystem service or, alternatively, how much they would be willing to receive to compensate for its loss or degradation.

The aim of **choice modelling** is to model the decision processes of individuals in a given context. In this method, individuals need to choose between two or more alternative means for providing attributes of the ecosystem services to be valued (one of the attributes being the money people would have to pay for the service) (TEEB, 2010). This method has been used, for example, to estimate the value of protecting groundwater from contamination in the drinking-water sector in Denmark compared with treatment to purify the water (Hasler *et al.*, 2005), with survey respondents asked to choose between alternatives where the levels of drinking-water quality, surface-water quality and price are varied systematically. The study found that the estimated willingness to pay for groundwater protection was higher than the willingness to pay for purified water, supporting the current Danish groundwater policy.

Benefit transfer. Benefit transfer comprises methods that rely on the use of research results from pre-existing primary studies at one or more sites to predict estimates for other, typically unstudied sites (Rolfe *et al.*, 2015).

POLICY AND MARKET-BASED INSTRUMENTS TO INCENTIVIZE FOREST HYDROLOGIC SERVICES

A "governance gap" exists between land-use and water planning (Bates, 2012), affecting the capacity to set integrated policies and market tools connecting the forest and water sectors. Water utilities, hydroelectricity plants and households are often "free riders" of the water services provided by sustainable forest management, benefiting from them without compensating forest owners and managers (Obeng, Aguilar and Mccann, 2018).

Governments and public agencies have financial ("carrots"), regulative ("sticks") and informational ("sermons") instruments at their disposal to meet the increasing demand for forest ecosystem services. Here, the focus is primarily on those policies and market-based instruments that can be classified as carrots, such as rewards, incentives, payments and investments to increase the provision of water services from forests.

Carrot-based policies and instruments include payments for ecosystem services (PES), defined as the "transfer of resources between social actors, which aims to create incentives to align individual and/or collective land-use decisions with the social interest in the management of natural resources" (Muradian *et al.*, 2010). Payments for watershed services (PWS) represent a subcategory of PES in which forest owners or managers are compensated for the provision of water services.

PWS is a promising mechanism for benefit-sharing and cooperation between the forest and water sectors, especially in the absence of a legislative framework or functioning local governance (Schomers and Matzdorf, 2013). Nevertheless, in practice, the approach and appropriateness of PWS in a given context should be evaluated carefully (Engel, 2016) and, if adopted, should be implemented not as a stand-alone solution but, rather, as part of a policy mix of incentives, legal restrictions and awareness-raising actions (Barton *et al.*, 2017). PWS, therefore, should be seen as part of a broader process of local participatory governance rather than as a market-based alternative to ineffective government or community management (Van Hecken and Bastiaensen, 2010).

Types of payment scheme for water services from forests

PWS mechanisms may be classified depending on the role played by the public sector, which can be as a buyer of water services (e.g. a public water utility) and as a legal actor providing a legal framework within which users may – or are obliged to – compensate or pay for water services (e.g. by imposing taxes on hydroelectricity plants). Figure 4.2 classifies the four main types of PWS governance model depending on the role of the state: 1) user- and non-government-financed payments; 2) government-financed payments; 3) compliant payments; and 4) compensation payments (Leonardi, 2015).

FIGURE 4.2
Types of payment scheme for ecosystem services, by role of the state

STATE INVOLVED AS A BUYER	STATE INVOLVED AS A REGULATOR: NO	STATE INVOLVED AS A REGULATOR: YES
NO	User-financed (Coasean approach) and non-government financed payments, e.g. Vittel case study	Compliant payments, e.g. mitigation and wetland banking in the United States of America
YES	Government-financed payments (Pigouvian approach), e.g. agri-environmental schemes in the European Union	Compensation payments for legal restrictions, e.g. groundwater protection areas payments

Source: Leonardi (2015), modified from Matzdorf, Sattler and Engel (2013).

Table 4.5 shows the main PWS typologies and their subtypes, based on their voluntariness (if demand and supply are voluntary or made compulsory by regulation); directness (of the benefit transfer between the beneficiary and supplier); aims and drivers (e.g. compensation for damage, avoiding impacts such as the use of chemicals, or providing additional ecosystem services by improving and maintaining existing resource conservation status); and financing mechanisms employed (Leonardi, 2015).

TABLE 4.5
Types of payment scheme for watershed services

Programme typology	Subtype	Major drivers	Descriptions in water-related forest services	Examples
Public – non-voluntary	Compensation for legal restrictions	Increased acceptance of legal restrictions through compensation for opportunity costs	Schemes used by governments to compensate farmers or forest owners for their opportunity costs in applying certain restrictions on their agricultural/forest management practices in a catchment. This approach is often used to improve the acceptance of regulations or due to equity concerns	This type of programme is relatively common in Europe and where strong environmental legislation exists; many national payment schemes in Latin America, such as in Costa Rica, are considered to be in this category (Pagiola, 2008)
Public – regulated	Agroforestry-based schemes	Provision of public goods, and may partially cover the adoption of management practices	This type is relatively common in Australia, Europe and the United States of America, dating to the 1970s. It typically involves national-scale incentive schemes, with little targeting or additionality; such schemes may incentivize tree-planting, the maintenance of tree hedgerows, fire control and sustainable forest management for water quality	90% of European Union funding for forests comes from the European Agricultural Fund for Rural Development (EAFRD). In the 2007–2013 programming period, approximately EUR 5.4 billion was allocated from the EAFRD budget to co-finance forestry measures, of which some are water-related (European Commission, 2020)
	Public bilateral agreements	Local provision of public goods	These are schemes enforced by public bodies on behalf of taxpayers, in which public or private suppliers participate in an agreement on a voluntary basis. Agreements are managed mainly by municipalities or public utilities. The funding mechanism is direct budget allocation/transfer, without the use of any innovative financial mechanism or policy	The New York City Watershed Agreement is an example of a public entity that directly establishes an agreement with farmers and forest owners. The funding mechanism is a simple budget allocation to a watershed programme driven by the City itself (Grolleau and McCann, 2012). China's Sloping Land Conversion Programme, which has been running since 1999, is the world's largest payment scheme for ecosystem services; its aim is to reduce soil erosion, and nearly USD 69 billion has been allocated to it through the central budget (Leshan *et al.*, 2017)
	Water charge – public bilateral agreements	Investing in water quality – customers are charged fees for water use	This funding mechanism is based on the charging of a fee for the use of water, at least some of which is distributed to upstream "suppliers". Schemes of this nature are reasonably common in all regions	Viet Nam's Payments for Forest Environmental Services scheme involves charging hydroelectricity plants and water utilities for their water usage. Most water-related payment schemes for ecosystem services in Latin America use fees for water use as their main funding source
	Regulated trading initiatives	Regulatory compensation/ offsetting	These are schemes that establish water-trading systems by allocating abstraction rights that can be sold among users, creating efficient allocation	These schemes are rarely applied in the forest sector. The main examples are water-trading schemes in Australia and the United States of America in the agriculture sector, and they are usually applied at the scale of river basins (Heberling, García and Thurston, 2010; Mariola, 2012)
	Guaranteed funds	Incentivizing investments in green infrastructure with below-market interest rates	These public funds agree to step in to cover a borrower's financial obligations to repay a lender under certain scenarios. A guarantee can also be provided by a third party to enable a borrower to access a loan. This can incentivize investments in less-profitable ventures, such as green infrastructure	The European Investment Bank's Natural Capital Financing Facility is backed by a European Union guarantee (Box 4.9). Other specialized financial mechanisms that can provide blended or concessional financing for green infrastructure projects are the Global Environment Facility, the Green Climate Fund and Climate Investment Funds

Continued ...

Programme typology	Subtype	Major drivers	Descriptions in water-related forest services	Examples
Private voluntary	Corporate social responsibility (CSR) offsetting	CSR water footprint voluntary compensation	Many private corporations fund water–forest projects to help "green" their images and to implement their CSR policies. Many such projects lack a clean methodology for compensation; many may be characterized as ad hoc or one-off interventions	Such schemes often involve private beverage companies, such as Coca Cola and Bionade
	Multiple-benefit partnerships	Improve the provision of water services through the maintenance and improvement of natural capital	Such schemes often work through a partnership model involving private companies, public regulators, non-governmental organizations and local authorities. The partnerships are usually managed by intermediary organizations that collect funds from beneficiaries and pay the service providers directly or implement restoration projects. Partnership agreements are made at the catchment level, where conservation objectives are aligned among actors with different interests. Usually a set of actions (e.g. for forest restoration) is implemented to provide multiple benefits related to, for example, water quality, biodiversity conservation and climate adaptation	In Kenya, the Lake Naivasha Basin Integrated Water Resources Action Plan Project is a partnership among the World Wide Fund for Nature, CARE, water-user associations around the lake and upstream communities. All these actors have committed to an action plan, and upstream communities are paid to restore forests and avoid the use of fertilizers with the aim of increasing water quality in the lake (WWF, 2015)
	Investment funds	Cost savings in operational costs through investment in green–grey infrastructure	These are private funds, such as environmentally focused bonds, funded by impact or philanthropic investors that invest in green–grey infrastructure projects to fulfil their impact-oriented missions while also expecting a return on their investment arising from reduced operational costs	Forest resilience bonds, green bonds and climate bonds. For example, the Climate Bonds Standard and Certification Scheme is a labelling scheme for bonds, including a section for green water infrastructure projects

In practice, each PWS scheme is a unique combination of institutional settings, local regulations, key actors, forest management practices and financial mechanisms used to transfer funds from beneficiaries to the suppliers of the ecosystem service(s).

The two most common schemes in the forest–water domain are water fees and multiple-benefit partnerships. Schemes that apply fees for water use are usually based on a defined normative background; they have proved to be both long-lasting and capable of mobilizing consistent quantities of funds at the subnational and national scales. Multiple-benefit partnerships are considered to be relatively resilient because of their capacity to value co-benefits, including social aspects and livelihoods, and to align multiple actors in a catchment-wide approach to forest–water management (Bennett, Nathaniel and Leonardi, 2014; UNECE and FAO, 2018). In addition to water fees and multiple-benefit partnerships, a trend is emerging towards incentivizing investment in forest–water infrastructure.

Water fees. Users of water services such as public and private water utilities and hydroelectricity plants usually depend directly on natural resources such as aquifers, water catchments and forests. The degradation of forests and associated increase in pollutants and sediments can directly increase their operational costs related to water treatment and the removal of sediments (Arias *et al*., 2011; Bennett *et al*., 2014). Addressing this equitably requires bringing upstream communities and downstream beneficiaries together. Upstream communities are often marginalized rural people, who contribute to catchment degradation in extracting a living through agriculture and forestry. Downstream communities would benefit from improved land management practices upstream.

To reduce degradation, governments set regulations designed to modify the activities of upstream communities with the aim of protecting downstream urban populations. Such regulations, however, often exacerbate the poverty, marginalization and illegal practices of upstream communities. An alternative is to charge a water levy or fee as part of household and industrial water and electricity bills, thereby providing a basis for bilateral contracts to pay upstream communities to improve their agricultural and forest practices and to compensate them for their forgone income (Figure 4.3).

In many countries, governments have integrated existing water- or forest-related legislation with the use of water fees. In some countries, such as Viet Nam, governments collect the fees and use the revenues to fund national forest management and water protection programmes (Box 4.6). Fee-based schemes involving the forest and water sectors exist in Asia (Bennett, 2016), Europe (Bennett and Leonardi, 2017), Latin America (de Paulo and Camões, 2020) and the United States of America (Bennett *et al*., 2014). There are relatively few such schemes in Africa, although they are increasing (South Africa has one of the continent's longer-standing examples; Box 4.7). The Nature Conservancy's Water Fund Toolbox supports the establishment of fee-based PES mechanisms and provides regional examples.

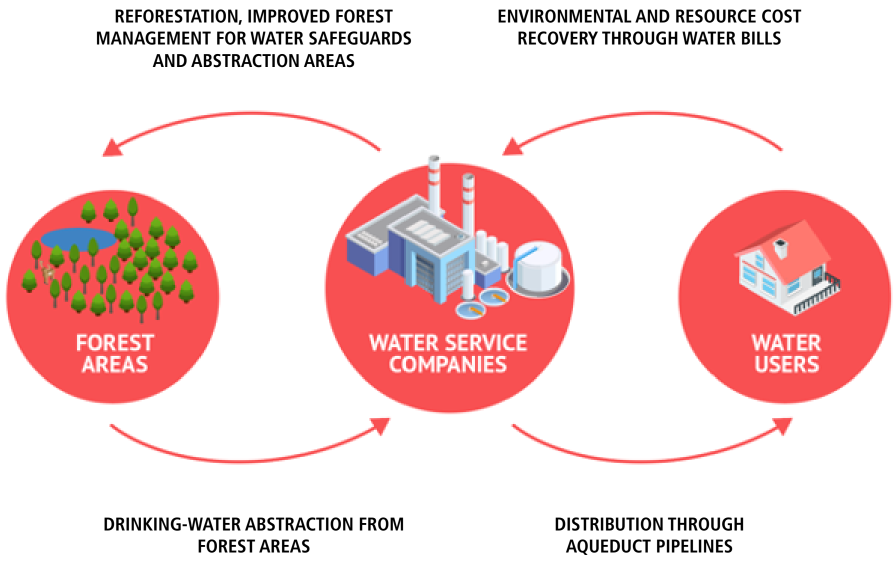
BOX 4.6

Viet Nam's payment scheme for watershed ecosystem services

Viet Nam was the first country in Asia to implement a national payment scheme for watershed ecosystem services, which the Government of Viet Nam views as a major breakthrough for the forest sector. Implemented in 2011, the Payments for Forest Environmental Services (PFES) scheme, which is regulated by Decree 99, contributed about 22 percent of total forest-sector investments in 2015. Payments are being channelled through water and electricity bills as a result of Decree 147/ND-CP in 2016, which amended and supplemented articles of a previous decree establishing the PFES. Accordingly, from 1 January 2017, the unit price of electricity increased from VND 20 to VND 36 per kWh for hydroelectric plants for commercial electricity and from VND 40 to VND 52 per m^3 for clean-water-supply plants. These price adjustments increased PFES revenues to about USD 86.7 million per year, with further potential increases for the forest sector. PFES provides funding for forest protection contracts, staff time, operational costs and capacity development for forestry activities; income for forest management boards, protected areas, national parks and state forest enterprises; and support for community development programmes.

Despite the success of the scheme in raising funds for forest management, there are still doubts about its efficiency and equity. A key finding of one study of the scheme is that, "No matter how the payment distribution mechanism is designed and selected, it has to be conducted in a participatory manner where stakeholders are properly consulted and their voices are well-considered and taken into account in the final decision" (Pham *et al.*, 2018).

FIGURE 4.3
The basic concept for fee-based payment schemes for water services



BOX 4.7
South Africa's Working for Water programme

South Africa's Working for Water (WfW) programme, which was launched in 1995, is administered by the national Department of Environmental Affairs. The programme has enabled the clearing of more than 1 million ha of invasive alien plants in mountain catchments, restored natural fire regimes and hydrologic functioning, and provided jobs and training for about 20 000 people from among the most marginalized sectors of South African society. Through their water fees, water utilities and municipalities contract WfW to restore catchments that affect their water supplies.

The success of the programme is due to a combination of clear hydrologic benefits and social co-benefits (Turpie, Marais and Blignaut, 2008; DEA, 2020). Although the WfW programme has been successful, payment schemes for watershed services often fail to improve water services in Africa because the need to focus on poverty reduction increases transaction costs. Such schemes also tend to rely on general public tax revenues for financing rather than direct payments by private beneficiaries (Ferraro, 2009).

Multiple-benefit partnerships. This model (depicted in Figure 4.4) has various names in the literature, including watershed partnerships, catchment partnerships, co-investments and collective action funds. Its main characteristic is that it is based on a participatory and collaborative local–national governance system in which public regulators, local authorities, private companies, non-governmental organizations and professional associations act together – often organized under an umbrella organization, partnership or cross-cutting institution – to improve watershed management. The model has the following key strengths:

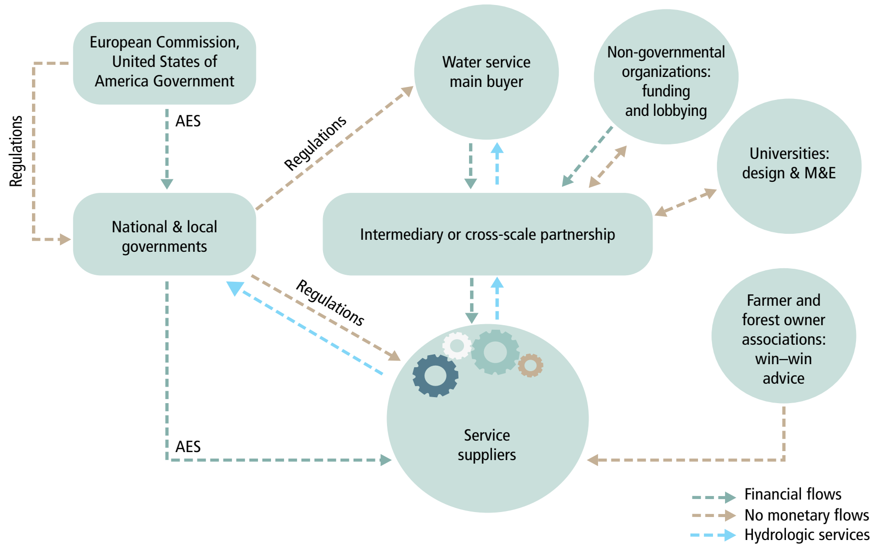
- **Multilateral agreements.** Contracts are signed by more than one organization and therefore differ from a market orientation and simple buyer–provider relationship. Rather, multi-actor contracts establish a common vision and agreement for the management of a watershed or forest.
- **Multiple sources of funding.** Various funding sources are used through the different development phases of the partnership, and matched funding ensures greater stability and complementarity among sources. Grants are used in the startup phase, payments

from beneficiaries in the implementation phase and private–public investments for scaling up.

- **Co-benefits.** Even though multiple-benefit partnerships have the main aim of ensuring adequate water quality and quantity, they often also provide biodiversity, carbon and socio-economic benefits. This constitutes the main means by which multi-actor participation and scheme acceptability are obtained.

Networks and collaborative approaches at the local level are a common characteristic in successful case studies, in which regulators, private companies, local authorities and technical and civil-society organizations share their expertise – through matched funding – to deliver high-level watershed schemes (UNECE and FAO, 2018).

FIGURE 4.4
Schematization of a partnership model



Notes: AES = agri-environmental schemes; M&E = monitoring and evaluation.
Source: Leonardi (2015).

Investing in forests as natural infrastructure

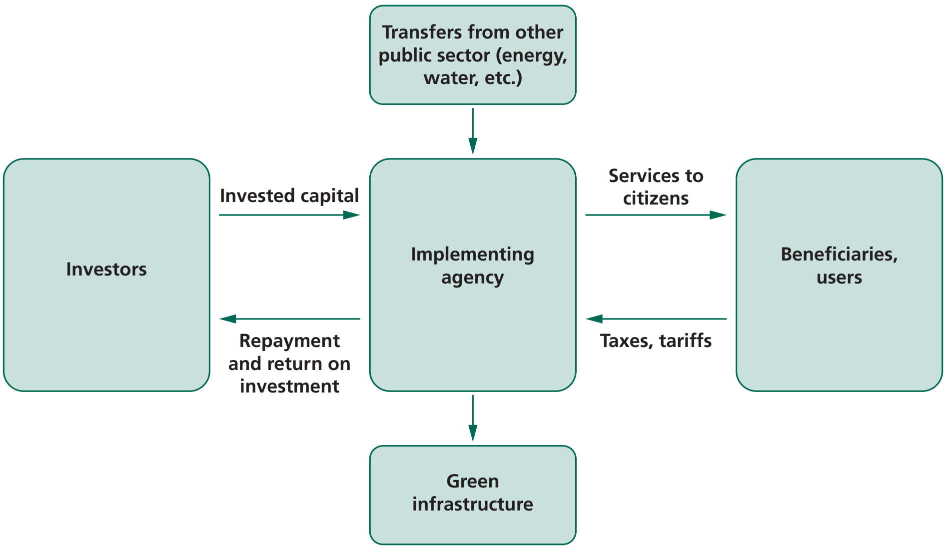
Global demand for infrastructure is growing, but governments often struggle to finance it; many governments are also failing to ensure the delivery of high social and environmental standards. Therefore, governments, the private sector and development agencies are increasingly providing concessional loans and guaranteed funds to couple grey-infrastructure development projects with green infrastructure in ways that support broader environmental and social goals while easing financing challenges. The World Bank, for example, financed 81 projects with nature-based approaches between 2012 and 2017; most of this green–grey infrastructure involves forests with the aim, for example, of mitigating dam sedimentation, absorbing urban stormwater and stabilizing coastlines (Browder *et al*., 2019).

These programmes run under investment logics, meaning that they are expected to provide a financial return. Compared with typical PES schemes, green infrastructure investment projects work in partnership with trust funds, guaranteed funds, banks and other financial institutions to provide the liquidity needed for forest-related investments (Figure 4.5). Compared with typical forestry businesses, where revenue is generated by timber sales, forest infrastructure projects provide savings by reducing operational costs, such as in dam maintenance and floor repair; this is the key factor in establishing investment deals (European Investment Bank, 2019). This model is useful when:

- the actors involved have cash-flow problems, with reduced liquidity;
- the project can demonstrate significant cost savings from reduced operational costs; and
- impact investors or guaranteed funds exist that can ensure below-market interest rates.

In the United States of America, private investors can purchase "forest resilience bonds" to fund forest–water management that reduces operational costs and increases natural capital (Box 4.8).

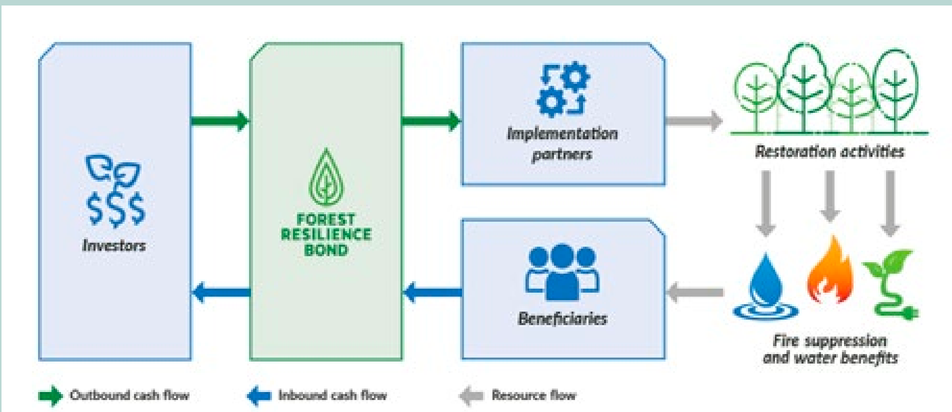
FIGURE 4.5
Forest infrastructure investment model



BOX 4.8
Forest resilience bonds in the United States of America

The Forest Resilience Bond is a public–private partnership to enable the financing of forest restoration in the western United States of America using private capital. Under the scheme, investors provide upfront capital, in collaboration with public and private beneficiaries (such as water utilities and the United States Forest Service), and make contracted payments based on the water-related and other benefits provided by the restoration (Figure 4.6). The investment opportunity (from a financial, technical and operational point of view) is packed into bonds – a widely used financial instrument – to facilitate the involvement of investors.

FIGURE 4.6
A schematic depiction of cash and resource flows under forest resilience bonds



Source: www.forestresiliencebond.com

Main phases in the development of water-related payment schemes

Globally, investments in forest-related watershed services are gaining importance as a tool for achieving forest–water policy aims (Bennett, 2016), but their design and governance are complex. Key challenges include:

- the complexity of choosing appropriate governance designs (Engel, 2016);
- legal and governance barriers (Hawkins, 2011);
- transaction costs for setting up and piloting schemes (Viani, Bracale and Taffarello, 2019);
- the additionality and permanence of interventions (Ezzine-De-Blas *et al.*, 2016);
- leakage effects and fairness (Lopa *et al.*, 2012); and
- monitoring and determining the effectiveness of forest management practices in improving water indicators.

Before embarking on a payment scheme, therefore, careful consideration should be given to whether it is the most appropriate policy option. Engel (2016) provided a useful guide for evaluating the appropriateness of PWS in a given context and selecting the appropriate design features, depending on the objectives.

The design of a PWS scheme involves the following ten operational steps:

1. **Identify and define a water quality/quantity issue and its related "forest solution".** Establish a clear link between the forests (biophysical structure), their primary environmental functions (e.g. phytodepuration and water retention) and the ecosystem services they provide (e.g. water quality and flood protection) (Brauman *et al.*, 2007). Awareness-raising is usually required before beginning the development of a PWS scheme because it is essential that key stakeholders recognize the problem and the potential of the PWS scheme to address it.
2. **Identify local actors.** All stakeholders linked to the water services need to be ascertained. These may include: downstream water users and others likely to be affected by the loss of an existing water service; landowners and land managers providing the water services (or those responsible for the source of diffuse pollutants); local authorities and regulators; and trusted intermediaries.
3. **Assess the feasibility of a PES scheme.** Are there willing buyers or payers for the water-related forest ecosystem services? Are those actors who benefit from forests or are affected by forest degradation willing to cooperate and pay for improved upstream land-use practices? Is the relevant government willing to revise or establish new regulations and encourage private actors to engage in collaborative and participatory resource management?
4. **Conduct a cost–benefit analysis.** It is important to assess whether the scheme will be able to achieve its goals given the likely budget and willingness of beneficiaries to pay. It is also important to understand the timeframe and geographical scale within which the goals can be achieved. Incentives and rewards designed to improve forest management can only be set when the economic value of the benefits to be derived from such improvement is clear and understood by stakeholders and beneficiaries.
5. **Explore potential win–wins.** Consider whether delivering the identified water services will also deliver other ecosystem services, such as carbon sequestration, recreation and biodiversity and, if so, whether markets exist for these. Where willing buyers exist, assess the scope to develop an integrated scheme and revise the cost–benefit assessment accordingly.
6. **Define roles and responsibilities.** Assuming there is local support for developing a PWS scheme, define the roles and responsibilities of actors, set boundaries, and agree on measures, associated costs, payments and timelines.
7. **Resolve any legal issues.** Consider the legal, fiscal and regulatory issues that may arise for the various actors, especially those making or receiving payments,

such as implications for taxes, property rights and pollution control.

8. **Set technical specifications.** Develop and agree on the technical specifications for the design and management of the forest measure(s) to be implemented (as identified in previous steps). These need to ensure the effectiveness of the forest interventions, including in terms of additionality and avoided leakage.
9. **Formalize payment contracts.** Draw up and finalize formal contracts between buyers and sellers, covering, among other things, technical specifications for the measures to be implemented, timelines for delivery, baseline water conditions, success criteria, monitoring needs, staged payments and scheduled reviews.
10. **Monitor, evaluate and review.** Monitoring can take many forms that vary greatly in cost. It should encompass biophysical aspects to verify whether the forest measures are providing clear water-related benefits; social and economic aspects to check how payments are affecting local communities and other stakeholders; and governance and design aspects to assess their effectiveness and the need or otherwise for modifications.

The complexity of developing a PWS scheme means that it requires the strong, ongoing commitment of all actors.

What can governments do to facilitate the emergence, consolidation and maturity of payment schemes for water services?

Governments are crucial for ensuring the success and longevity of PWS schemes; ways in which they can support such schemes are described below.

Develop national guidelines, toolkits and best practices. In many countries, local professionals and practitioners struggle to find adequate information on PWS schemes in their own languages and suited to their local contexts. Governments can assist by creating clear guidance documents to provide a basis for developing PWS schemes at the national and subnational levels. For example, the Government of the United Kingdom of Great Britain and Northern Ireland published a national guide on PES, including an annex of best practices (DEFRA, 2013), which formed the basis for developing PES nationally and also elsewhere in Europe. In Latin America, national governments (supported by international organizations) have cooperated to develop the Latin America Water Funds Alliance, a website dedicated to the setting up of "water funds" in the region (see Table 4.6, which also provides information on a toolbox created by The Nature Conservancy and a database of case studies maintained by Forest Trends).

TABLE 4.6
Toolboxes and databases on payment schemes for watershed services

Owner	Type	Source
Nature Conservancy	Toolbox with a database of case studies, training and a dedicated online network	www.waterfundstoolbox.org
Forest Trends	Online database with case studies	www.forest-trends.org/about-our-project-data
Latin America Water Funds Alliance	Toolbox with a database of case studies, training and a dedicated online network for Latin America	www.fondosdeagua.org

Establish legal frameworks that allow/oblige water services. Domestic and industrial water uses (e.g. irrigation, hydroelectricity generation and drinking water) should include green taxes/charges in water/energy bills to reinvest in forest watershed protection. Worldwide, PWS schemes have emerged when a solid legal framework has been provided through government action. Most frequently, these frameworks have been included in comprehensive water laws and therefore provide a holistic approach to watershed management. Table 4.7 shows examples of legislation that created fees for watershed services to help pay for forest watershed management.

TABLE 4.7
Examples of legislation that includes water fees for forest watershed management

Place	Legislation	Articles detailing water-related fees
European Union	Directive 2000/60/EC establishing a framework in the field of water policy (EC, 2000)	"Article 9. Recovery of costs for water services. 1. Member States shall take account of the principle of recovery of the costs of water services, including environmental and resource costs, and considering the economic analysis … and in accordance with the polluter pays principle in particular"
Colombia	Decree 1900/2006 and further modification (MADS, 2006)	"Article 1. Any project that involves the use of water taken directly from natural sources and that is subject to obtaining an environmental licence, must allocate 1% of the total investment for the recovery, conservation, preservation and monitoring of the water basin that feeds the respective water source"
Peru	Law 28823 – Creation of the National Water Fund FONAGUA (Government of Peru, 2006)	"Article 1. Creates the Water Fund FONAGUA with the aim to promote sustainable and integrated watershed management. Art. 3 establishes that FONAGUA's economic resources are made up of: a) 2% of the component 'Water board association' referred to in article 8 of the Regulation of Rates and Fees for the Use of Water approved by Supreme Decree No. 003-90-AG; b) 3% charge on water fees for non-agrarian use"
Costa Rica	Decree 32868 (Government of Costa Rica, 1997)	"The charge for water use must be used as an economic instrument for the regulation of the use and administration of water that allows water availability for reliable supply in human consumption and social development. Economic growth of the country and also the generation of economic resources to finance long-term sustainable water resource management in Costa Rica"
Viet Nam	Decree 147/2016/ND-CP amending 99/2010/ND-CP – Policy Payment of Forest Environment Service Charge (Government of Viet Nam, 2016)	"Beneficiaries of forest environment services shall pay service charges to service providers. 1. For hydropower generation establishments: The rate of forest environmental service charge payable by hydropower: generation establishments is VND 36 per kWh of commercial electricity. The electricity amount used to calculate the payable charge amount is that sold by a hydropower generation establishment to electricity buyers under electricity trading contracts; 2. For clean water production and supply establishments: the rate of forest environment service charge payable by clean water production and supply establishments is VND 52 per cubic metre of commercial water. The water volume used to calculate the payable charge amount is that sold by a clean water production and supply establishment to consumers"

Note: Some of the text in this table comprises unofficial translations of the original.

Establish a small funding programme for pilot activities. The startup phase of a PWS scheme (or any PES scheme) is likely to require considerable time and resources. Startup costs are generally considered higher than general transaction and operational costs and may influence directly the efficacy of a scheme (Wunder, 2007). It may be possible to at least partly cover startup costs with the help of international funds in the form of grants for feasibility studies, environmental monitoring and participatory activities. International non-governmental organizations such as WWF, The Nature Conservancy and Forest Trends have specific support programmes that may provide technical assistance and startup funding.

In some cases, governments have created funded programmes designed to systematically support national learning processes and capacity development. The Government of the United Kingdom of Great Britain and Northern Ireland, for example, commissioned three rounds of PES research pilots between 2012 and 2015 to test the practical application of the concept in new contexts. All 16 pilots, which addressed a range of habitats, services and spatial scales, were commissioned after competitive bidding processes, with each receiving grants of about USD 30 000 to fund feasibility studies and startup costs. Catchment-based projects showed the most potential, for example by delivering cost-effective water-quality improvements. The pilot testing was a valuable learning experience for stakeholders and governments in developing feasible PES concepts; it highlighted the important role of governments in developing metrics and frameworks that provide assurance and confidence for investment (DEFRA, 2016).

Create a national public–private investment fund. PWS schemes are relatively new tools, and their returns on investment may not always be clear or predictable. Therefore, forest–water infrastructure projects may find it difficult to attract co-investors or donors who lack understanding of the risks involved. The European Union's fund for the environment and climate action has created a guarantee fund to incentivize the European Investment Bank to engage in green infrastructure and sustainable forest management projects. This will help the bank bear the risk of highly innovative projects and provide below-market interest rates for projects in the range of USD 10 million–20 million (European Investment Bank, 2019); the bank has also created the Natural Capital Financing Facility to support such investments (Box 4.9).

Create a link with social protection and livelihood programmes. In developing countries especially, schemes have emerged that aim to couple environmental protection goals with social inclusion and livelihood improvements in marginalized rural and forested areas.

BOX 4.9

The European Investment Bank's Natural Capital Financing Facility

Interest is growing in the conservation sector in innovative forms of blended finance – that is, financing mechanisms that involve the integration of funds of different sources and character, often combining public and private investments. The Natural Capital Financing Facility (NCFF), which has been put in place by the European Commission's Directorate General of Environment and the European Investment Bank, is dedicated to supporting innovative natural-capital conservation projects and the application of nature-based solutions. The NCFF is looking for new EU-based projects to finance green infrastructure, payments for ecosystem services, funds for environmental compensation and biodiversity-friendly business activities. Existing forest and green infrastructure projects include one to increase stormwater absorption in Athens, Greece, and another to convert monocultural plantations to multifunctional forests in Ireland. The facility has two components:

1. A technical assistance service that offers non-repayable financing for preparation, implementation, monitoring and evaluation (up to EUR 1 million).
2. A flexible financial service that provides loans or investments in the form of debt or equity (USD 2 million to USD 15 million) for a maximum of 75 percent of the cost of the project. Using this tool has several advantages, including:
   - increasing the number of loans available at rates lower than market rates;
   - decreasing investment risk due to the European Commission's guarantee fund; and
   - integrating financial support with external assistance through non-repayable financing.

The technical assistance service is considered a preparatory tool parallel to the investment phase. The fund has been piloted since 2017 and will welcome projects until the end of 2021.

Source: www.eib.org/en/products/blending/ncff/index.htm

In the Kulekhani watershed in Nepal, for example, a revenue-sharing mechanism aims to avoid dam sedimentation in a hydroelectricity scheme and provide additional annual funds for the community to, for example, supply households with electricity, construct new roads and support children's education. The scheme has been questioned, however, for failing to meet its environmental goals and for local political issues (Khatri, 2012).

In the Serchio Valley in Tuscany, Italy, a forest watershed monitoring scheme was set up to involve forest owners in the cleaning of waterways and the restoration of riparian vegetation as means for mitigating flooding and slope erosion. The scheme has been successful thanks to its clear co-benefits – it provides forest owners with an additional source of income and a cost-effective alternative to centralized water-authority interventions. A payment-for-results mechanism helps maintain the performance and commitment of forest owners and ensures adequate monitoring and the successful achievement of environmental objectives.

The provision of social co-benefits is a key feature of success for PWS schemes, but these should not distract attention from the primary goal, which is to improve the provision of water services through forest management. Strong political involvement and dependency should be avoided, and payments by results and effective monitoring systems should be put in place to ensure the achievement of project ecosystem goals.

MANAGING TRADE-OFFS AND DECISION-SUPPORT SYSTEMS

Forests face conflicting demands for the ecosystem services they supply. Most ecosystem services are interdependent, and their relationships may be non-linear (Heal *et al*., 2001); therefore, understanding their interactions may be challenging (Tallis *et al*., 2008). Nevertheless, an understanding of the linkages between ecosystem services and their management is needed for effective decision-making.

Various terms exist for the relationships among ecosystem services, such as associations and bundles (Mouchet *et al*., 2014), but, in most cases, these relationships are framed as trade-offs and synergies (Raudsepp-Hearne, Peterson and Bennet, 2010). In this sense, "trade-off" means that an increase in one ecosystem service results in a decrease in one or more other ecosystem service (e.g. increasing carbon stock in a forest may lead to a decrease in water yield). "Synergy" refers to situations in which management to increase the provision of one ecosystem service also increases the provision of one or more others (e.g. riverine vegetation, if properly managed and conserved, can both increase water quality and improve habitat quality for aquatic and amphibian species).

Trade-offs and synergies in ecosystem services emerge from the biophysical properties of ecosystems and their associated constraints, but they are also linked to socio-economic dimensions. Stakeholders may differ in their needs or preferences for ecosystem services due to differing contexts, cultures or scales. Moreover, external policy, institutional, cultural and economic factors may influence the efficient management of ecosystem services by impeding or enabling trade-offs and synergies (Cavender-Bares *et al*., 2015).

Trade-offs and synergies can arise at different dimensions and scales (Rodríguez *et al*., 2006), as follows:

- spatial scale – when spatial lags may be identified between ecosystem-service supply and demand (e.g. the effects of interactions among ecosystem services are perceived locally or at more-distant locations);
- temporal scale – when the effects of interactions among ecosystem services differ over time, and temporal lags may be identified; and
- reversibility – that is, the likelihood that an ecosystem service may return to its original state after perturbation.

Trade-offs and synergies may also be observed in different states (e.g. higher or lower supply) of the same ecosystem service as a result of external independent drivers (Bennett, Peterson and Gordon, 2009).

The Economics of Ecosystems and Biodiversity assessment (TEEB, 2010) proposed a classification using terminology similar to that suggested by Rodríguez *et al.* (2006) for the Millennium Ecosystem Assessment but framed in terms of economic benefits and costs. It implies trade-offs between beneficiaries, where beneficiaries can be either “losers” or “winners” depending on who bears the cost or benefits of the ecosystem service (Mouchet *et al*., 2014). For example, upstream farmers can increase agricultural output and therefore their revenue by increasing chemical inputs (e.g. fertilizers), but this may generate costs for downstream communities and reduce their access to clean water.

Trade-offs are inherent in the supply of water services, which might vary in terms of quantity, quality, location and timing (Brauman *et al*., 2007). Therefore, a robust understanding of relationships among ecosystem services is needed to optimize land-use decisions and synergies and to avoid unwanted trade-offs, unexpected changes in the supply of ecosystem services and missed opportunities to support synergistic interactions and win–win management solutions. Such an understanding needs to be embedded in a supportive framework that integrates policies and initiatives in line with evolving social demand for forest-based ecosystem services.

There is generally poor recognition in policies and among policymakers that trees and forests play a role in water recycling; trees and forests, therefore, are often seen as end users rather than as part of a greater system that redistributes water (Springgay, 2015). Optimizing trade-offs between water use, water yield and forest-related ecosystem services requires strengthening the interface between the scientific community, knowledge-holders and policymakers, thereby developing capacity for and strengthening the use of science and knowledge in policymaking on forest–water interactions.

Most studies of the forest–water nexus and interactions among ecosystem services have investigated water yield and quality at different scales, considering both trade-offs and synergies among land uses and ecosystem services, with particular reference to timber yield and carbon sequestration and, to a lesser extent, biodiversity conservation. Water services, however, go far beyond water yield; they include aspects such as soil retention, land surface cooling, soil salinity management, physical barrier and riparian protection, freshwater biodiversity benefits, infiltration and groundwater recharge, and contributions to precipitation patterns. Many of these are little discussed in the scientific literature on trade-offs and synergies (Malmer *et al*., 2010; Creed *et al*., 2016; 2019).

Balancing ecosystem services, human well-being, livelihoods and poverty alleviation

Ecosystem and land management strategies imply making choices, not only among the various land uses and ecosystem services but also among groups in society (e.g. upstream and downstream communities, current and future generations, local resource users and the global community) (Vira *et al*., 2012; Lehmann *et al*., 2014). Land-use and management choices can exacerbate trade-offs by altering socio-environmental interactions, affecting local resource users and increasing the vulnerability of certain groups or community members (Kerr *et al*., 2007; Goldman-Benner *et al*., 2012). This may particularly be the case with the allocation of the benefits and costs of ecosystem services, especially if governance processes are poorly conceived (Lehmann, Martin and Fisher, 2018). Any strategy for ecosystem management implies opportunity costs (Tallis *et al*., 2008), and stakeholders within the system are differentially exposed to these (Vira *et al*., 2012). Similarly, individuals and groups may perceive the benefits of ecosystem services differently because of differences in their access, knowledge, norms and values and the surrounding as well as individual contexts (Daw *et al*., 2011; McDermott, Mahanty and Schrekenberg, 2013). Ronnback, Crona and Ingwall (2007) reported that although coastal villagers in Kenya make use of a broad range of ecosystem services provided by mangrove forests, individuals perceive these services differently depending on their home village, gender and livelihood.

Management choices that change the delivery of ecosystem services can affect people differently, generate trade-offs among people, and ultimately create winners and losers. For example, the creation of a protected area for mangrove forests may have negative effects on fishers and woodfuel collectors, who may lose (part of) their livelihoods but increase revenue and employment opportunities in the tourism sector (Daw *et al*., 2011).

There are knowledge gaps on how the well-being of particular groups of people is affected by trade-offs among ecosystem services. The costs and benefits of ecosystem services are often considered in terms of their total social value – that is, aggregated at a regional or higher level – without considering how different groups may share the costs and benefits (Kovács *et al*., 2015; Robinson, Zheng and Peng, 2019). Most attempts to assess and quantify ecosystem services do not disaggregate beneficiaries or differentiate between stakeholder groups at different scales (Lau *et al*., 2018). Aggregated values provide important information for understanding policy options and assessing biophysical trade-offs (Zheng *et al*., 2016), but they may be inappropriate for designing targeted PWS schemes and identifying where trade-offs occur (Robinson, Zheng and Peng, 2019). The direct relationship between ecosystem services and human well-being can be better measured at the local scale, such as a community or household, and this may enable improvements in efficiency and the incorporation of multiple dimensions of social equity into policies on ecosystem services (McDermott, Mahanty and Schrekenberg, 2013; Pascual *et al*., 2014).

When designing and implementing mechanisms for valuing ecosystem services, including in PWS schemes, trade-offs need to be identified carefully in order to ensure both natural-resource protection and livelihood security. Market-based instruments such as PWS might constitute new strategies for exploiting synergies among ecosystem services but are unlikely to eliminate the trade-offs that characterize many resource-use decisions (Redford and Adams, 2009). Economic analysis (Carpenter *et al*., 2009) and multicriteria decision analysis (Vogdrup-Schmidt *et al*., 2017) can help in dealing with trade-offs, but an overreliance on technical approaches may neglect the political dimension of negotiating and integrating different visions (Friend and Blake, 2009). This suggests that social-equity considerations should be integrated into the management of ecosystem services – although there is a risk that such considerations could be obscured by the focus on economic efficiency that characterizes some PES schemes (Pascual *et al*., 2014).

The design of PWS schemes requires the disaggregation of ecosystem services and their values, as well as negotiation with multiple stakeholders with differing and sometimes conflicting positions (Hope *et al*., 2007). Decision-support approaches and decision-making tools can help in building and negotiating effective agreements and mechanisms.

The decision-making process

Forest owners, users and managers should consider the trade-offs and synergies that arise from specific management decisions (e.g. policies, plans and investments). The decision-making process will need to be adapted depending on the number of stakeholders involved, differences in their goals, interests and perceptions, their desired level of participation (see Germain, Floyd and Stehman, 2001), and the models and methods adopted for valuation scenarios. Once the decision hierarchies have been defined and a role assigned to each ecosystem service under consideration, it is necessary to address trade-offs and synergies in valuation and decision-making.

Including the valuation of ecosystem services in decision-making

TEEB (2010) proposed a stepwise approach cited by Masiero *et al.* (2019) for the valuation of ecosystem services and their inclusion in decision-making. The three main steps are described below.

1. **Obtain the information needed to identify and assess each ecosystem service.** Consider, and take steps to involve, the full range of stakeholders influencing or benefiting from the affected ecosystem service.
2. **Define and implement appropriate valuation methods to make the economic value of each ecosystem service explicit.** This step also involves analysing the linkages over space and time that affect when and where the costs and benefits of particular uses of biodiversity and ecosystems are realized (e.g. local to global, current versus future use, upstream versus downstream and urban versus rural) to help frame the distributive impacts of decisions. Valuation is best used for assessing the consequences of changes in the provision of ecosystem services arising from different management options (scenarios) rather than attempting to estimate the total value of ecosystems (TEEB, 2010). Scenarios might consider mutually exclusive alternative solutions as well as possible future developments deriving from a given solution as a consequence of different internal and external factors and drivers. Several approaches can be adopted for building scenarios and analysing ecosystem services, some of which can be implemented together in complementary ways, rather than as stand-alone approaches. These include:
   - Participatory techniques – Lynam *et al.* (2007) provided a review of tools for incorporating community knowledge, preferences and values in natural resource management.
   - Expert opinion – professionals with expertise in the economic effects of ecosystem services provide inputs and outline the expected impacts of policy changes (e.g. via focus groups or using the Delphi method – see Mukherjee *et al*., 2015) (Masiero *et al*., 2019).
   - Analysis of similar cases – especially when gathering primary site-specific data is costly, a popular alternative method is to conduct a benefit transfer involving the application of economic value estimates in one location at a similar site elsewhere (Plummer, 2009).
   - Modelling – this might involve the use of dedicated tools for modelling ecosystem services and software to support and improve decision-making and planning (see below).
   - Mixed approaches – a combination of two or more of these approaches is used (Masiero *et al*., 2019).
3. **Capture the value of ecosystem services.** Capturing the value of ecosystem services and seeking ways to overcome their undervaluation can be done using technically and economically sound and informed policy instruments. Such instruments may include changes in subsidies and fiscal incentives; charging fees for access and use; PES; targeting biodiversity in poverty reduction and climate adaptation/mitigation strategies; creating and strengthening property rights and liabilities; and voluntary ecolabelling and certification. The choice of tools will depend on context and should take into account the cost of implementation.

Below, we provide a more operational approach to the use of decision-support systems for managing trade-offs and synergies in forest–water management.

Decision-support systems for forest–water management. Although many models and a range of software exist to support decisions in forestry, most are tools for the valuation of biophysical ecosystem services that simulate various scenarios and provide quantitative outputs. Thus, such decision-support systems[13] should be used in conjunction with other techniques, such as participatory approaches, to ensure robust, comprehensive decisions. This also includes the consideration of socio-economic factors, which are sometimes difficult to estimate. In connection with this, a useful reference is a guide for decision-makers by Ranganathan *et al.* (2008).

The scientific literature refers to many software packages and tools for the valuation of ecosystem services, from general to specific. We reviewed 108 forest management decision-support systems to identify those specifically addressing forest management goals related to water services. Twelve systems (about 11 percent of the total) had management goals related to water (mainly water quality and groundwater recharge). Four of those with water-related objectives enable analysis at a regional or national level – that is, at a spatial scale suitable for supporting decisions at the scale of river basins (most systems operated at a local or landscape level). Table 4.8 describes these four tools, which are deemed suitable for the management of trade-offs and synergies for water services.

TABLE 4.8
Forest management decision-support systems potentially suitable for addressing trade-offs relevant to water services

Decision-support system	Description	Management goals for water services
InVEST (Sharp, Douglass and Wolny, 2016)	A tool for exploring how changes in ecosystems are likely to lead to changes in benefits that flow to people. It enables decision-makers to assess quantified trade-offs associated with alternative management choices and to identify areas where investment in natural capital can enhance human development and conservation	Water quality Hydroelectricity
Ecosystem Management Decision Support (Reynolds, 2006)	An application framework for knowledge-based ecological assessments at any geographic scale. The system integrates state-of-the-art geographic information systems with knowledge-based reasoning and decision-modelling technologies to provide decision support for a substantial portion of the adaptive management process of ecosystem management	Watershed restoration
NED-2 (Twery *et al.*, 2005)	A Windows-based system designed to improve project-level planning and decision-making by providing natural-resource managers with useful, scientifically sound information. Resources addressed are visual quality, ecology, forest health, timber, water and wildlife. The NED-2 system is adaptable to small private holdings, large public properties and cooperative management across multiple ownerships. NED-2 implements a goal-driven decision process that ensures that all relevant goals are considered; the character and current condition of forestland are known; alternatives for managing the land are designed and tested; the future forest under each alternative is simulated; and the alternative selected achieves the owner's goals	Groundwater recharge Water quality
Pimp your Landscape (Fürst *et al.*, 2010)	A platform to support planners by simulating land-use scenarios and evaluating the benefits and risks for regionally important ecosystem services. The platform also supports the integration of information on environmental and landscape conditions into impact assessments and the integration of the impacts of planning measures on ecosystem services. It is a modified two-dimensional cellular automaton with geographic information system features	Water quality

Other decision-making approaches. Various free GIS applications can also be used to support decision-making. For example, Brancalion *et al.* (2019) and Strassburg *et al.* (2019) both present multicriteria spatial restoration prioritization frameworks in which scenarios can be simulated by weighing each factor under observation.

[13] For the purposes of this publication, decision-support systems are "computer based systems that represent and process knowledge in ways that allow the user to make decisions that are more productive, agile, innovative and reputable" (Burstein and Holsapple, 2008).

COMMUNICATING AND BRANDING FORESTS FOR WATER PROJECTS AND INITIATIVES

Communication in forest-related activities is not a major research topic nor one of the main skills among forest–water practitioners (IUCN, 2010). A search for the term “forest communication” in Scopus (the main global scientific literature database) obtained only ten records; a search for “forest marketing” produced five records; and no records were produced in a search for “forest branding”. No paper identified in these searches addresses forests or sustainable forest management from a marketing or communication perspective. The number of scientific papers on the topic is not the only possible indicator of communication efforts; nevertheless, the lack of academic attention does suggest that forest communication has not been a high priority in the sector. Given the importance of forests in the provision of a wide range of ecosystem goods and services, including water services, it is essential to address this communication gap to influence community knowledge on, and attitudes towards, forest–water management. Here, we present an approach for communicating and branding forest–water management to enhance community engagement, policy commitment and willingness to invest.

“Marketing” addresses the values that a project brings to target beneficiaries (e.g. the environmental changes to be delivered); “communication” is the means (i.e. content and channels) by which such values and changes are delivered. A company, project or programme can create a “brand” through marketing and communication – that is, the way in which stakeholders perceive the initiative and support, engage in and ultimately pay for it (Box 4.10). In this section, we use the term “communication” to encompass the concepts of marketing, communication and branding.

BOX 4.10

Marketing, communication and branding

According to the American Marketing Society, marketing “is the activity, set of institutions, and processes for creating, communicating, delivering, and exchanging offerings that have value for customers, clients, partners, and society at large”; another definition is “the act of making change happen” (Godin, 2018).

Communication is the act of conveying meanings from one entity or group to another through the use of mutually understood signs, symbols and semiotic rules.

A brand is a set of expectations, values, principles, memories, stories and relationships that, taken together, account for a consumer’s decision to choose one product or service over another. If the consumer (whether a business, buyer, voter or donor) doesn’t pay a premium, make a selection or spread the word, then no brand value exists for that consumer. A brand’s value is merely the sum total of how much extra people will pay for, or how often they will choose, the expectations, memories, stories and relationships of one brand over the alternatives (Godin, 2018). Branding is most commonly carried out by individual companies, but it can also be part of broader environmental communication campaigns that promote specific behaviour changes.

There is value in employing a communication strategy as a means to increase the effectiveness of forest–water initiatives. Regrettably, the forest sector has generally failed to adopt the following basic rules of communication:

- **Negative versus love messages.** The dominating communication approach in conservation and forestry has been to use negative messages such as those around deforestation and forest fires (IUCN, 2010). Alternatively, it is possible to produce communication material that highlights the importance of forest management in the provision of clean freshwater.

- **Technical versus simple wording.** People tend to trust conservation scientists and technicians, but they often do not understand what such experts are saying (Thompson *et al.*, 2016) and there is a need, therefore, to simplify the messages. For example, tree–water linkages could be used to increase awareness of the relationship between forests and water. Trees are simple natural objects that most people understand well; "forests" and "forest management", on the other hand, are concepts that many people have difficulty grasping.
- **Public versus specific target.** It may be beneficial to know the motivations of private forest owners in adjusting their behaviours to conform to forest policy objectives (Boon, Meilby and Thorsen, 2004). A forest–water project may have different communication messages for upstream and downstream users and for other key players. Such projects are usually complex, and communication needs to be targeted carefully to reach the right audiences with the right messages.
- **Add action.** A communication message should end with a "call to action" – that is, something the target audience can do to help address the identified challenge. What change are we hoping for? How do we help the target audience make a change?

Communicating a forest–water project will assist in (Konijnendijk *et al.*, 2005):

- gaining political and public support and funding;
- strengthening the morale and internal organization of institutions and partnerships involved in the initiative by providing a broader vision and mission;
- engaging more beneficiaries and buyers and thereby spreading the word; and
- building trust and relationships with new users, including ethnic minorities, women and youth.

Building a communication strategy

The aim of communication is to provide a venture – such as a forest–water project, programme or initiative – with a recognizable identity, differentiating it from the others and building public support by creating a community of "followers".

Converting a forest–water venture into a brand requires a strategy designed to translate the venture's environmental goals into a specific identity and a set of marketing and communication activities that must be integrated into the operations of the venture. Thus, communication shouldn't be addressed at the end of a project preparation phase – it should be taken into account throughout all phases of the venture.

Although there is a general lack of marketing, communication and branding research directly relevant to forest–water ventures, examples exist. Also, lessons can be learnt from "territorial branding", a practice in destination-marketing projects in which public and private organizations come together to create a brand to promote a nature-tourism site.

Based on an analysis of communication strategies for existing forest–water projects and nature tourism, we propose a nine-step process for designing a communication strategy (Figure 4.7). All these steps, described in detail below, can be addressed while developing or improving a business or project and answering the "why, what, where, who and when" of the strategy.

FIGURE 4.7
Components of a forest–water communication strategy

1. **Background analysis.** This step is carried out to analyse the environment of the venture to understand better where to position the brand. It is likely to be useful to gather information from similar projects and to involve key staff and communication, marketing and branding experts to brainstorm ideas for developing the strategy.
2. **Strategy objectives.** This step should clarify the questions "Why?" and "What change do we want to make happen?". The analysis should start with discussing the key objectives of the venture and gaining understanding of the desired behaviour change. Possible specific questions might be, "Do we want citizens to pay for a green water bill?"; "Do we want forest managers to improve their management to ensure high water quality?"; and "Do we want investors to finance our green infrastructure project?" List objectives in order of importance. The Great Green Wall for the Sahara and the Sahel Initiative provides a good example of clearly stated qualitative and quantitative goals.[14]
3. **Target audiences.** Analyse the potential stakeholders involved in the forest–water venture as customers or beneficiaries and categorize them in terms of scale, influence and interest. Who are the key partners? What people, and groups of people, will be affected by or will benefit from the venture? Which actors could influence these stakeholders (e.g. influencers, media and policymakers)? Group stakeholders into audience categories and list them in order of importance (Raum, 2018).
4. **Value proposition and claims.** A value proposition is a promise of the value to be delivered, communicated and acknowledged by the venture. It is also a belief held by customers or beneficiaries about how the value (benefit) will be delivered, experienced and acquired. Identifying the value proposition is the first step towards developing an effective claim by which to communicate the venture to key audiences. Questions to pose include, "What are the main gains the key audiences will get from the venture?" (e.g. improved water quality); and "What pains (or problems) will the venture solve for them?" (e.g. increased water bills or the risk of forest fire). The following are examples of claims that communicate key benefits for audiences:
   - The Mersey Forest in the United Kingdom of Great Britain and Northern Ireland – which combines the name of the project, "The Mersey Forest", with

[14] More information on the initiative is available at www.greatgreenwall.org/about-great-green-wall

the catchy tagline, "more from trees", to get "The Mersey Forest – more from trees".[15]

- The Coca Cola Foundation's "Replenish Africa Initiative (RAIN)".[16]
- In reaffirming its brand defined by a natural setting (mountains), Evian's tagline is – "Evian – water the way nature intended".[17]

5. **Key messages.** Key messages should target specific audiences relevant to the forest–water venture, and they should have the following characteristics: clarity, consistency, repetition, tone, appeal, credibility, public need, and language of communication. There should not be too many key messages, which should be simple and easy to understand and help motivate the audience. List the key messages and provide a content description for each, with references and key facts and figures.
6. **Visual identity.** A visual identity can be formed by various components (Figure 4.8) to attract the viewer's attention and communicate – even before words – the venture's intrinsic values. A strong visual identity is one that:
   - Is easily recognizable – a single design direction and visual identity makes it easier for stakeholders and audiences to recognize a venture's products.
   - Builds trust and confidence – when materials are neatly designed and organized, there is greater trust that things are working well. An inherent messiness, on the other hand, starting from the design and use of branding, risks confusing audiences (who do not know what to expect) and reducing their trust in the knowledge materials.
   - Stands out from the crowd.
7. **Channels and tools.** This step answers the questions of "Where?" and "How?" The proper communication of information to the full range of stakeholders and others is crucial for the venture's design and implementation and requires choosing appropriate communication means and channels. These might include field trips, seminars, events, television, media, film, posters and flyers, online outreach with websites and newsletters, social media, and information workshops (offline and online) (Box 4.11). Communication channels should be selected and planned for different audiences, bearing in mind constraints related to funds, time and human resources. Sometimes, conservation actions might be adapted to act as communication channels themselves – for example, tree-planting to protect a water resource could be carried out by organizing community tree-planting days for families, which also serve as opportunities for communication about the venture.
8. **Action plan and budget.** A clear set of work packages and activities should be planned and implemented (Figure 4.9). The action plan should specify the human and financial resources required for implementation.
9. **Monitoring and evaluation.** The monitoring and evaluation plan should answer the following key question: What are the objectives of the evaluation? (They should be tied to the objectives of the communication strategy and the broader goals of the venture.) Monitoring and evaluation should track progress in the implementation of key communication activities, including indicators of the impacts of communication.

[15] More information is available at www.merseyforest.org.uk
[16] More information is available at https://replenishafrica.com
[17] More information is available at www.evian.com/en_us/sustainable-bottled-water/water-sustainability

FIGURE 4.8
Visual identity components

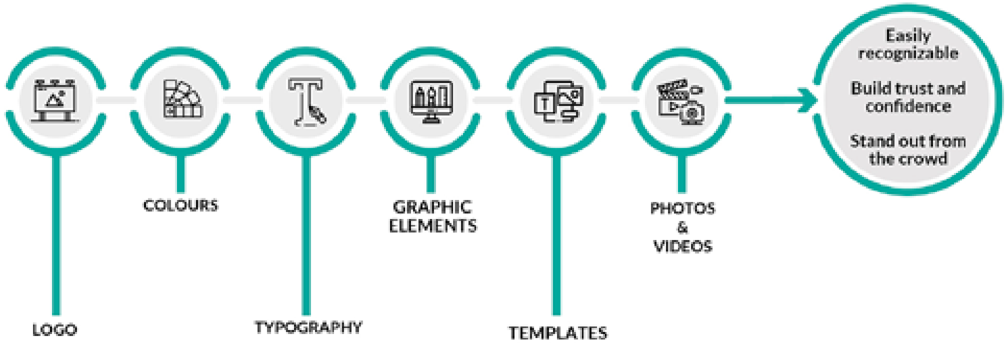


FIGURE 4.9
Components of a communication action plan

Note: WP = work package.

BOX 4.11

Examples of water-related communication messages and tools

The regional agency for agricultural and forestry services (ERSAF) in Lombardy, Italy, is promoting its commitment to sustainable forest management and forest–water-source protection by banning plastic bottles at all recreational sites (e.g. mountain huts and shelters) and serving only tap water using labelled jugs indicating the forest source. ERSAF's forest management is certified by the Forest Stewardship Council (FSC), and its contribution to water services is certified according to the FSC procedure on ecosystem services. More information: www.ersaf.lombardia.it/it/b/460/imbroccalacquadibosco

CamminaForeste 2017, CC by ERSAF Lombardia

Drinking-water protection areas can be branded using public signage on the importance of these forest areas. Protecting source water from contamination helps reduce treatment costs and may avoid or defer the need for complex treatment. The delimitation and communication of forest–water protection areas can form part of national policies on the protection of water resources. More information: https://commons.wikimedia.org/w/index.php?curid=76951800

Conservation Keith – Own work, CC BY-SA 4.0

Demonstration sites or recreational areas can be used as controlled spaces in which the public interacts with specific features of a venture. This is the case for Bosco Limite in Italy, where a "forest infiltration area" has been opened to the public to provide an outdoor "showroom" for recreational and educational purposes. Thousands of people who have visited the small site now have greater understanding of the concept of forest infiltration areas. A massive communication strategy has been deployed to increase exposure, including a website, a tree-adoption campaign, branded signage, social networking and other communication materials.
More information:
www.wownature.eu/areewow/bosco-limite

Forest infiltration area in Veneto Region, Italy

The aim of the "Forests to Faucets" campaign of the United States Forest Service is to communicate to communities the importance of forest areas for drinking water; it includes the interesting use of web maps. More information: www.fs.fed.us/ecosystemservices/FS_Efforts/forests2faucets.shtml

Seattle's faucets_CC by LukeMcGuff

Table 4.9 presents a list of existing international and regional networks and toolboxes that provide useful tools for communicating and promoting forest–water projects. Communication has a cost, but it is essential for building successful projects. Therefore, forest–water ventures should include adequate planning, budgets and staff for effective communication.

TABLE 4.9
Forest–water-related communication networks and toolboxes

	Description	Source
United Nations Economic Commission for Europe (UNECE) Communication and Outreach Hub and Forest Communicators' Network	Produces news releases, hosts the Forest Information Billboard, shares presentations and reports and is involved in a range of forest-related events. The UNECE/FAO Team of Specialists on Forest Communication – the Forest Communicators' Network – is the major platform for forest communication experts to exchange and find common strategies	www.unece.org/forests/information/fcn.html
Forest Pedagogics	Provides a Europe-wide forum for information and communication about forest pedagogics, presenting data, activities, materials and networks for foresters, teachers and other actors in forest-related education for sustainable development	http://forestpedagogics.eu/portal
FAO Forest Communication Toolbox	Comprises photos, videos, infographics, social media cards, PowerPoint presentations and key facts and messages, by topic, including watershed management and sustainable forest management. Since 2011, FAO has supported the development of forest communicators' networks in five regions – Africa, Asia-Pacific, Europe and Central Asia, Latin America, and the Mediterranean and the Near East	www.fao.org/forestry/communication-toolkit
The Nature Conservancy's Water Funds Toolkit	Contains presentations, templates, examples and guidance documents for the development of consistent messaging and materials on water security, source-water protection, and water funds	https://waterfundstoolbox.org/component/communication

5 Key ecosystems for forest–water management

Water is an integral component of all forest ecosystems, but the relationship is especially pronounced in some ecosystems. This chapter brings together the various concepts explored in chapters 3 and 4 by describing four forest ecosystem types – mangrove, pleatland, tropical montane cloud, and dryland – in which management for water services is particularly important and which are especially vulnerable to climate change, deforestation, land degradation and land-use change. Although these ecosystem types use and provide water in unique ways, they all serve to maintain forest and natural resource sustainability. Moreover, each is threatened by climate change, variability and associated disturbances (e.g. shifts in weather patterns, sea-level rise, drought and wildfire).

MANGROVE FORESTS

Key points

- There are approximately 13.8 million ha of mangrove forests worldwide, with the bulk in 15 countries.
- Mangroves provide many essential ecosystem services and play important roles in climate change mitigation and adaptation.
- An estimated 30–35 percent of mangroves have been lost since the 1980s. About one-quarter of remaining mangroves are considered to be moderately to severely degraded.
- Forest width is the most important factor in determining the mitigation potential of mangrove forests against tsunamis and storm surges.
- Integrating mangroves in disaster risk reduction strategies and coastal management planning can help reduce the risk of coastal disasters.

Mangrove forests occur commonly along coasts, rivers and estuaries in the tropics and subtropics, with the largest areas at latitudes between 5° North and 5° South. In 2020, 113 countries reported approximately 14.8 million ha of mangroves worldwide (FAO, 2020a), distributed mainly in 15 countries (Giri *et al*., 2011). These highly specialized forested wetland systems (distinguished by the functionality of the plant species they contain – Box 5.1) occupy intertidal zones and are adapted to regular water inundation in a range of salinities (e.g. freshwater to hypersaline) (Tomlinson, 1986).

BOX 5.1
Defining mangroves

The term mangrove is a descriptor of function, not phylogenetic relationship, with nearly 75 mangrove species found in 20 families that include small shrubs, palms and trees (Duke, 1992). Several morphological and physiological adaptations enable mangrove trees to survive the harsh conditions of coastal and estuarine life. Their highly vascularized root systems exclude salt from the soil water they use and pump oxygen into anoxic sediments. Pneumatophores and knee roots project upwards from sediments, and prop roots and buttresses extend radially from trunks to provide stability in unconsolidated sediments and areas of high tidal action. In some species, roots and leaves are also able to extrude salt to maintain the balance of cellular fluids.

Mangroves provide many essential ecosystem services to human communities living in and near them. For example, many species of fish and invertebrates that either live within mangrove systems or access them during flood tides are important sources of protein for humans, other wildlife and livestock (Primavera *et al*., 2004; Nagelkerken *et al*., 2008; MacKenzie and Cormier, 2012; Analuddin *et al*., 2019). Many species of mangrove trees are harvested for their insect- and rot-resistant wood, which is used for building, handicrafts and woodfuel; mangrove palms are important sources of roofing thatch (Dahdouh-Guebas *et al*., 2000; Primavera *et al*., 2004; Naylor and Drew, 1998).

Mangroves play important roles in climate-change mitigation and adaptation. Their high rates of primary productivity can remove large amounts of carbon dioxide from the atmosphere (Alongi, 2012). The majority (up to 90 percent) of this carbon is stored in mangrove soils under waterlogged and anaerobic conditions that reduces microbial respiration (Donato *et al.*, 2011; Murdiyarso *et al*., 2015). If undisturbed, mangroves can act as carbon sinks for several millennia (Atwood *et al*., 2017). Belowground root growth is an important mechanism for maintaining the elevation of mangrove forest floors relative to sea-level rise (Krauss and Allen, 2003), thus providing mangrove forests with capacity to adapt to climate change, although the rate of sea-level rise could outstrip this capacity.

Because mangroves are located at the terrestrial–ocean interface, they may be considered as coastal guardians that protect inland areas from storms and nearshore areas from sediments and pollution. Mangrove trees and aboveground root structures (e.g. prop roots and pneumatophores) can significantly reduce the velocity of water moving through them (Furukawa and Wolanski, 1996), thus mitigating the wave energy generated by storm events. During inundation from flood tides, decreased water velocity reduces the transport of sediments, which are then more likely to be deposited and trapped on the forest floor (Furukawa and Wolanski, 1996; MacKenzie *et al*., 2016). This also increases the residence time of water in mangrove forests and, to an extent, enables nutrients and heavy metals to be taken up by plants or incorporated into sediments and increases the quality of water that eventually flows out to adjacent nearshore seagrass beds, coral reefs and water bodies (Clough, Boto and Attiwill, 1983; Schaffelke, Mellors and Duke, 2005). In addition to the protection they provide to adjacent nearshore ecosystems, mangrove forests help shield forests directly inland such as coastal strand forests and peat swamps by attenuating wave energy and minimizing salt spray.

Threats to mangrove forest–water relationships

Despite the benefits they provide, an estimated 30–35 percent of mangrove forests have been lost worldwide since the 1980s (Alongi, 2002; FAO, 2007), although some regions have lost much more than that (FAO, 2007). An estimated one-quarter of remaining mangroves are moderately to severely degraded and under threat of conversion to

Mangroves in Guna Yala, Panama

agriculture, aquaculture and other development and of overexploitation for charcoal and timber (Giri *et al.*, 2008). Rising sea levels due to global warming have been identified as one of the greatest future threats to mangroves and the ecosystem services they provide (Gilman *et al.*, 2008).

Management of mangrove forests for water services

Key parameters that determine the magnitude and effectiveness of the protection offered by mangroves against coastal disasters include forest width, tree density, age, tree diameter, tree height and species composition (Box 5.2 and Box 5.3). These parameters can be manipulated through forest management to produce the required level of mitigation against potential disasters; they are, however, co-dependent, interlinked and influenced by other physical and geographical features, such as ground elevation, nutrient and freshwater input, exposure to the sea, and undersea topography (Forbes and Broadhead, 2007).

The following practical guidance, based on current understanding of how and when mangroves can help reduce the risks posed by coastal disasters, is provided to assist forest managers in optimizing the role of mangroves in coastal protection strategies.

- **Maintain wide mangrove belts.** Ideally, mangrove belts should be hundreds or thousands of metres wide to reduce the impact of winds and high waves during storms and tsunamis. If maintaining or restoring mangrove belts of this width is not possible, narrower strips and even isolated patches can offer certain degrees of protection and serve as sources of propagules for natural mangrove expansion and replanting. The reduction of coastal flooding by mangrove belts can minimize salt water intrusion to freshwater and potentially maintain freshwater resources for drinking and/or agriculture.
- **Prevent mangrove conversion.** Existing mangrove forests should be maintained by enabling local communities to use them sustainably, thereby incentivizing responsible mangrove management. Mangrove-friendly aquaculture and community-based forest and fisheries management can be effective in minimizing the degradation and conversion of mangroves.

- **Conserve healthy mangrove forests.** Forest quality influences the degree of protection that mangroves provide**.** It is important, therefore, to encourage and maintain structurally and biologically diverse mature mangroves over large areas by (for example) minimizing pollution, waste dumping, drainage (from upstream and local areas) and unsustainable use. In addition, evapotranspiration from intact mangrove forests is also an important water source for upwind and upland landscapes, including forests and/or streams.

BOX 5.2

Factors in the mitigation effects of mangroves

Forest width is the most important factor in determining the mitigation potential of mangrove forests against tsunamis and storm-generated waves. Waves lose energy and height as they move through mangrove vegetation. Estimates vary on how different mangrove-forest widths reduce wave energy and height. Spalding *et al.* (2014) reported that the height of waves caused by major storm events is reduced by 13–66 percent across a 100-m width of mangrove vegetation. Hiraishi and Harada (2003) used analytical models to demonstrate that 30 trees per 100 m^2 in a 100-m wide greenbelt could potentially reduce maximum tsunami flow pressure by more than 90 percent. Modelling by Yanagisawa *et al.* (2010) indicated that a 500-m-wide belt of mangrove forest could potentially reduce the hydrodynamic force of a tsunami by 70 percent for waves less than 3 m in height. In summary, although the wider the better, mangrove belts of 100–500 m are likely to offer substantial protection against tsunamis and high waves caused by storms. For waves less than 6–8 m in height, widths as small as 50–100 m may provide substantial mitigation (Forbes and Broadhead, 2007).

The density or permeability of mangroves is another important factor in their ability to reflect and absorb wave energy. Both vertical (i.e. how biomass is distributed vertically) and horizontal density are important. In general, mitigation potential increases with increasing vertical and horizontal density (Forbes and Broadhead, 2007). Density is easier to manipulate in planted compared with natural mangrove forests, but natural forests have other advantages. The high density of stilt roots (e.g. of *Rhizophora* species) in mature, healthy mangrove forests, which are usually uneven-aged and multistoried, offer considerable protection against coastal disasters. A clear relationship was observed in many countries in the 2004 Indian Ocean tsunami between the coverage of dense, intact mangroves and reduced damage to coastal infrastructure (Forbes and Broadhead, 2007). Increases in age, diameter and height generally enhance the mitigation effects of coastal forests, including mangroves (Harada and Kawata, 2005; Tanaka *et al.*, 2007; Forbes and Broadhead, 2007). As mangrove trees grow older, the density, height and thickness of their stilt roots and the height of their canopies increase, reducing porosity and increasing the reflection of incident waves, resulting in increasing hydraulic resistance with age (Mazda *et al.*, 1997). The risk of overtopping by high waves decreases with increasing tree height; the top height of mature mangrove stands can reach more than 30 m, which is beyond the height of storm surges and even large tsunamis.

Species composition also has implications for the mitigation potential of mangroves because it determines forest structure. In natural mangrove forest, different mangrove species generally dominate in different zones depending on the responses of individual species to variations in tidal inundation, salinity and other edaphic gradients. Combinations of different mangrove species growing in their natural habitats maximize mitigation potential by offering different types of resistance and increasing structural heterogeneity. If planting is required in the restoration of mangrove forests, consideration should be given to proper site–species matching to ensure long-term sustainability.

- **Restore mangroves.** Mangrove restoration is highly desirable in areas that were previously covered by mangrove forests and where the original cause of mangrove loss has ceased (e.g. where aquaculture ponds have been abandoned and where mangroves were destroyed by disasters). In many cases, mangroves will recover naturally where ecological and socio-economic conditions are suitable; mangroves restored through natural regeneration generally survive and function better than planted mangrove systems. If replanting is necessary, appropriate site–species matching will increase the chances of success, along with the involvement of local communities in planning and management.
- **Integrate mangroves in disaster-risk-reduction strategies and coastal management planning.** This should be done based on local-level assessments of the role of mangroves in coastal defence and risk mitigation and the full value and costs of mangrove conservation.
- **Adopt hybrid approaches to disaster risk reduction.** It is unlikely to be feasible to establish and maintain an unbroken mangrove bio-shield of sufficient width along an entire coast, and hybrid approaches that combine green and grey infrastructure should be considered (Spurrier *et al*., 2019). Given the low cost of establishing and maintaining mangroves relative to hard structures, however, and their potential to provide additional economic and environmental benefits, mangroves should be preferred wherever possible (Forbes and Broadhead, 2007).

Mangrove research needs and knowledge gaps

The ability of mangroves to protect inland forests from wave and wind action is well documented, but mangroves may also influence water cycles at the local, regional and global levels. Evapotranspiration from intact coastal forests can be an important source of precipitation. Freshwater flowing into coastal waters is used by mangroves and returned to that atmosphere through evapotranspiration (MacKenzie and Kryss, 2013), which in turn contributes to precipitation upwind and inland, providing water to landscapes, including forests and streams. While studies on evapotranspiration are generally lacking from mangrove forests, Lagomasino *et al.* (2015) has suggested that mangrove evapotranspiration rates could produce an equivalent amount of water as annual rainfall in certain years, although this likely varies across forest structure, tidal regimes and salinities (Barr, DeLong and Fuentes, 2014; Krauss *et al*., 2015). Studies combining remote sensing and field measurements are needed to fully understand the spatial variability in mangrove evapotranspiration as well as their contributions to the regional water balance.

Mangroves such as these that line the Pukusruk River on Kosrae Island in the Federated States of Micronesia protect upland forests from storms, store large amounts of carbon, and provide valuable habitat for many species of native fish, shrimps and crabs

BOX 5.3

The protective role of coastal vegetation

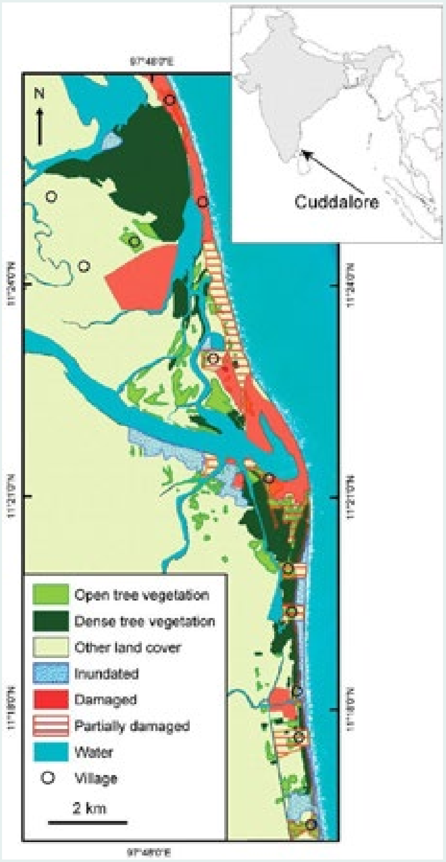
On 26 December 2004, a megathrust earthquake off the west coast of northern Sumatra, Indonesia, generated an unprecedented series of massive tsunami waves that ranged in height, depending on location, from less than 1 m to nearly 20 m (Danielsen *et al.*, 2005; Goff *et al.*, 2006; Satake *et al.*, 2006; Tsuji *et al.*, 2006). The waves flooded areas up to 2 km inland, destroying parts of the coasts of 14 Indian Ocean nations and killing more than 280 000 people (Danielsen *et al.*, 2005; Lay *et al.*, 2005; Jankaew *et al.*, 2008).

A study in India by Danielsen *et al.* (2005) documented the effective physical barrier provided by intact mangroves against powerful tsunami waves. The coastline of the Cuddalore District of Tamil Nadu, India, is relatively straight and comprises vegetated and non-vegetated areas. The vegetated areas include mangrove forests and plantations of *Casuarina equisetifolia* plantations. Mangrove forests were dominated by trees 3–8 m tall and 4.5–16.5 cm in diameter at breast height; the main tree species were *Rhizophora apiculata* and *Avicennia marina* at stem densities ranging from 1 400 to 2 600 trees per ha. The *C. equisetifolia* plantations were established along the coast as 200-m-wide shelterbelts after a cyclone in 1969; the trees were 18–23 m tall and 9.8–18.8 cm in diameter at breast height, and the stem density was 1 900–2 200 trees per ha. Pre- and post-tsunami assessments of 1 000-m-wide strips of coastline using LANDSAT images revealed that the tsunami, which had a maximum runup height of 4.5 m, completely destroyed most of one village located at a river mouth that lacked protective vegetation, as well as two villages to the north located in front of a dense mangrove forest (Figure 5.1). In contrast, three villages, also to the north but located behind mangrove forest, suffered no destruction, even though the tsunami damaged areas to the north and south of the villages that were unprotected by vegetation. The study also found that dense tree vegetation was associated with undamaged areas and disassociated with damaged areas.

Although the study by Danielsen *et al.* (2005) underscores the importance of maintaining intact mangroves to protect inland areas from tsunamis and other storms, the differences in damaged versus undamaged areas reported here could have been due to differences in wave energy along the coastline. Although bathymetric charts suggest that the slope was similar along the coast, undetected topographic features could have influenced wave energy along the coast. These results also only pertain to a small study area more than 1 500 km from the source of the tsunami.

FIGURE 5.1

Pre-tsunami vegetation cover and post-tsunami damage in Cuddalore District, Tamil Nadu, India



Note: Dark-green shaded areas represent mangroves and shelterbelt plantations of *Casuarina equisetifolia* that were planted after a cyclone in 1969. Open vegetation comprised all other woody vegetation, including degraded mangroves and gaps in the plantations

PEATLAND FORESTS

Key points

- Forests growing on peat soils play a crucial role in water regulation (flood and drought mitigation) and the maintenance of water quality at the catchment level.
- Unlike other forest types, there is a synergistic relationship between the water and carbon services provided by peatland forests.
- Peatlands are the world's most carbon-dense terrestrial ecosystems; their conservation is one of the most cost-effective ways to decrease greenhouse-gas emissions.
- It is estimated that one-quarter of the world's peatland forests disappeared in the period 1990–2008.
- Peatland drainage dramatically increases the risk of fire.
- Effective peatland ecosystem restoration would help ensure the delivery of water-filtering and regulating services and also provide sustainable livelihood options in wet peatlands while reducing forest and peat fires and land degradation and loss.

Peatland forests (defined in Box 5.4) are distinguished from other peat ecosystems by trees, which comprise the main biomass-forming flora, resulting in woody peat. They occur worldwide and total area is generally in decline despite ongoing mapping efforts identifying new peatland areas yearly, especially in the tropics (Joosten, 2010; FAO, 2014; Dargie *et al.*, 2017). It is estimated that peatland forests have declined from over 93 million ha in 1990 to less than 70 million ha in 2008 (Joosten, 2010). Boreal and temperate areas host the majority of the world's peatlands where they have formed under climatic regimes with high precipitation and low temperatures (FAO, 2014; Dargie *et al.* 2017). In the tropics, peatland forests commonly occur as peat swamps – rain-fed ecosystems in which the partly decomposed organic matter from dense rainforest vegetation accumulates in peat.

Over several millennia, peat increment results in the formation of peat massifs between rivers. The tree diversity of boreal and temperate peatland forests is low, generally dominated by the Pine family (Pinaceae) acoompanied by Picea, Pinus, and Larix (Bourgeau-Chavez *et al.*, 2018). In contrast, tropical peatland forests are often extremely high in biodiversity. At least 200 tree and palm species occupy tropical peat swamps in Indonesia alone (Bourgeau-Chavez *et al.* 2018). An Indonesian study of 26 (2000 m^2) plots contained 82 tree species with a diameter above 5 cm. (Lampela *et al.*, 2017; Astiani *et al.*, 2021.).

BOX 5.4

What is a peatland forest?

Peatland forests are a type of wetland recognized in the Convention on Wetlands of International Importance Especially as Waterfowl Habitat (also known as the Ramsar Convention). More than 280 peatland forests covering over 19.8 million ha have been designated worldwide as Ramsar sites or "wetlands of international importance" (Ramsar Convention Secretariat, undated). There is no universal definition of peatland forests; this report draws on the definition of forest used in the Global Forest Resources Assessment (FAO, 2020a) and the Intergovernmental Panel on Climate Change (IPCC)'s approach for peatlands. The IPCC's wetland supplement (IPCC, 2014b) includes peatlands in (land with) organic soils. Organic soils, also called histosols or peat, are identified based on three criteria related to the thickness of the organic horizon (at least 10–20 cm); organic carbon content (at least 12–20 percent by weight); and saturation with water (FAO, 2020b). Therefore, peatland forests can roughly be defined as wetland forests growing naturally on organic soils.

Peatlands provide various water services. Peat forms from the accumulation of partially decomposed plant matter over thousands of years under conditions of waterlogging in oxygen-deficient conditions. In peatland forests, trees and other vegetation such as mosses are essential for water regulation (flood and drought mitigation) and the maintenance of water quality at the catchment level. Drinking and irrigation water is often extracted from peatland forests.

Peatland forests have an air-conditioning effect due to evapotranspiration in the landscape. The relatively high evapotranspiration from trees and wet areas reduces surface temperatures and mitigates temperature extremes, thus minimizing nutrient and water losses (Hesslerová *et al.*, 2019). Evapotranspiration and therefore vegetation are considered effective climate regulators, not only locally but also globally (Hesslerová *et al.*, 2019).

In addition to their water services, peatlands are the world's most carbon-dense terrestrial ecosystems and are crucial, therefore, for climate-change mitigation (FAO, 2020c). Unlike other forest types, peatland forest management does not require trade-offs between water and carbon. Because most of the carbon in peatland forests is stored in the organic soil, which requires water for its formation and conservation, there is a synergistic relationship between the two.

Many peatland forests are biodiversity hotspots that support a wide diversity of habitats for rare flora and fauna species. This facilitates the provision of diverse products – food, biofuel and fibre – that support the livelihoods of many local communities (Wichtmann, Schröder and Joosten, 2016).

Threats to peatland forest–water relationships

Despite their importance and the large array of services that peatland forests provide, many of these ecosystems have become degraded and are under increasing threat from agricultural crops such as palm oil and cacao (FAO, 2020c; Miettinen and Liew, 2010). Current estimates suggest that 11–15 percent of peatlands on Earth have been drained and another 5–10 percent are degraded due to other changes such as the removal or alteration of vegetation (FAO, 2020c). Moreover, farming and plantations on dried peatlands are unsustainable. A combination of progressive soil degradation, decreasing productivity and the increasing cost of drainage has resulted in the abandonment of many peatlands, in which state they are especially prone to fire (FAO, 2014).

There has been widespread conversion of tropical peatland forests in Southeast Asia; this could also occur in other regions (e.g. the Amazon and Congo basins) unless lessons are learnt and solutions found (Murdiyarso, Lilleskov and Kolka, 2019). Other threats to peatland forests include forest plantations (for pulpwood or timber), mining, oil extraction and peat extraction.

Peatland drainage greatly increases the risk of fire. Fires in the peat soil layer, which are very difficult to detect and extinguish, may last for months, even during extensive rains (Joosten and Clarke, 2002). In the past, peat fires have been associated with large numbers of premature human deaths from respiratory illnesses, as well as large economic losses due mainly to air pollution in the form of haze (Koplitz *et al.*, 2016). Disturbances can also affect the hydrologic balance of peatland forests and cause severe erosion in mountain regions as well as soil compaction and land subsidence.

Land subsidence due to the draining of peatland forests can occur especially quickly in the tropics – as much as 1.5 m in the first five years after draining and 3–5 cm per year in subsequent years, depending on peat oxidation and watertable depth (Hooijer *et al.*, 2012). The lowering of peatland surface levels, combined with rising sea levels, increases the risk of flooding (and subsequent saltwater intrusion) in coastal areas and of major losses of productive land. The draining of peatland forests also increases the overall risk of drought and flooding as peat soils subside due to compaction and drainage, which changes the ability of the soil to contain water (FAO, 2020c; Taufik *et al.* 2020;

Ikkala *et al.* 2021). If the peat surface is strongly degraded, it can become hydrophobic, thereby reducing soil infiltration and increasing run-off, which prevents groundwater recharge (Wösten *et al.*, 2008) and makes it impossible to restore the peatland. Peatland drainage also leads to growing water pollution associated with increased concentrations of nitrates and dissolved organic carbon (Abrams *et al.*, 2016). It is clear, therefore, that the sustainable management of peatland forests is essential for their water regulation functions and their role in reducing the vulnerability of local communities to the increased likelihood of extreme climate events and hydrologic impacts.

Management of peatland forests for water services

Elements for the sustainable management of peatland forests for their water services are discussed below.

Recognize, define and quantify the problem. The first requirement for improving the management of peatland forests for water is to recognize and define the problem. Baseline information can be acquired through mapping and monitoring using both remote sensing and ground-level data (see FAO, 2020c).

Stakeholders should also be identified and their capacity developed to assess and monitor the status of peatland forests, identify management objectives and develop tangible and sustainable management plans, regulations and policies. Empowering local women and men to become stewards of their environment and its resources will enable them to obtain financial and social benefits from the sustainable use of peatland forests. Bottom-up solutions implemented jointly with local communities and other stakeholders, as well as scientists, are more successful than centralized top-down decision-making approaches (Wösten, Rieley and Page, 2008).

Protect peatland forests. Protection is the easiest and most cost-effective means for increasing the resilience of local communities in the face of increasing climate variability, which threatens to undermine water availability, food production and livelihoods. Protecting peatland ecosystems from deforestation and drainage will have tangible benefits for the environment and societies (FAO, 2014; World Bank, 2016); failure to do so will require ecosystem restoration, which is expensive and may be unable to restore full ecosystem functionality. For example, it has been estimated that USD 4.6 billion is required to fully restore 2 million ha of peatland forests in Indonesia (Hansson and Dargusch, 2018).

Use holistic landscape approaches. Peatland forests should be managed holistically at the landscape scale. Peatland forests should be considered at the scale of individual forests (i.e. the peat dome) and that of the watershed. Changes in water quantity and quality in peatland forests may affect their health and functionality. This is important because drainage or other disturbances to the hydrology of a peatland complex (or part thereof) will lower the watertable in the entire peatland area. Landscape approaches need to be applied in any intervention to conserve and restore peatlands to optimize efficiency in decision-making and avoid or minimize leakage.

Shift to wet management systems. An increase in awareness of, and investment in, the sustainable use of peatland forests is required to shift the peatland drainage paradigm towards recognition that low-impact, mixed-livelihood activities such as ecotourism, fisheries, agriculture and forestry are possible in wet peatland landscapes.

"Paludiculture" is the production of biomass on wet and rewetted peatlands in conditions that maintain and facilitate peat accumulation and ensure the provision of peatland ecosystem services. In many places, however, peatland forests must be put to productive use if they are to be conserved, and paludiculture is much more preferable than drainage (Wichtmann, Schröder and Joosten, 2016) (Box 5.5). Income from paludiculture can be generated through the use of biomass as well as through payments for ecosystem services such as those related to water, carbon (e.g. REDD+) and biodiversity (Wichtmann, Schröder and Joosten, 2016; Wösten, Rieley and Page,

BOX 5.5

Potential for sustainable livelihoods in tropical peat swamp forests

In Indonesia, restoration efforts started at an unprecedented scale after the peat and forest fires of 2015–2016, with a national ambition to restore 2 million ha by 2030. Technical improvements are needed for these efforts to be sustainable, but they offer promise (Giesen and Sari, 2018). One aspect that could make a considerable difference is the inclusion of local awareness and engagement in peatland restoration activities. Smallholders (who represent about half the peatland area converted to palm oil and acacia pulpwood) particularly depend on the income derived from these drained peatland forests, and local participation is key to ensuring the success of rewetting.

Paludiculture, including sustainable low-impact fisheries, is currently the only approach for balancing the productive use of peatlands and the provision of ecosystem services by providing livelihood options that do not require draining and which discourage the use of fire. An assessment of Southeast Asian swamps identified 534 peat-swamp plant species with known uses (e.g. timber, medicines and food), and 81 non-timber forest products had "major economic use" (FAO, 2014). Species such as candlenut (*Aleurites moluccanus*), illipe nut (*Shorea* spp.) and swamp jelutung (*Dyera polyphylla*) have the potential to provide alternative products and offset some of the environmental pressures associated with oil-palm and acacia cultivation (FAO, 2014).

Peat swamp fisheries have considerable potential for the production of both food and ornamental fish species. To facilitate the fish catch, fishers use artificial ponds called *beje*, which use the overflow of rivers during the rainy season to trap 5–12 fish species (FAO, 2014). The non-intensive raising of small livestock in rewetted peatlands is another option because a range of peatland plant species provide palatable and relatively nutritious fodder (Giesen and Sari, 2018). Beekeeping also shows promise, in combination with tree species such as *Melaleuca cajuputi*. However, despite the potential, the expansion of paludiculture is hindered in Indonesia by a lack of knowledge and market opportunities. The lack of a supportive regulatory framework is another obstacle (Giesen and Sari, 2018).

Peatland forest landscape in Katingan, Indonesia

2008). Sustainable value-chain development remains a key challenge for paludiculture approaches, particularly on badly degraded peatlands distant from large communities.

Paludicultural practices must be well adapted to site-specific conditions such as peatland type, soil conditions, nutrient availability, the natural high acidity of peatlands, and hydrology (Wichtmann, Schröder and Joosten, 2016). In general, intercropping, perennial species and mixed agroforestry approaches are likely to be more beneficial for peat hydrology than monocultures.

Apply adaptive management. If the rewetting of drained peatland forests is infeasible, adaptive management practices should be applied that avoid or minimize deep drainage, soil tillage and the use of fertilizers. Peatland forest management should aim to maintain continuous forest cover and employ selective harvesting (rather than clearfelling). If the land is put to agricultural use, permanent crops should be preferred (FAO, 2014). The key to the sustainable management of peatlands is simple: the closer the watertable is to the surface, the larger the benefits for peatlands and the communities around them.

Restore peatland forests. Peatlands are formed and maintained through the interaction between three elements: water, plant matter that creates peat, and soil. Changes in any of these can cause major alterations in the entire peatland ecosystem. Vast areas of peatland forests have already been drained, deforested or otherwise degraded; these activities must be stopped and reversed through peatland restoration and rewetting (Box 5.6; Box 5.7). Rewetting may not always be feasible; moreover, the longer restoration activities are delayed after drainage, the harder and more time-consuming it will be to bring back ecosystem functions to close to pre-drainage levels.

Vegetation helps maintain water in peatlands. Even in waterlogged anoxic conditions, peat slowly diminishes if there is no replacement with new peat-forming vegetation. If a peatland area to be restored was originally forested, native peatland forest species should be enabled to return; if, however, no individuals of the original species are growing in the area to act as seed sources, replanting may be needed (FAO, 2020c).

BOX 5.6

Rewetting peatlands is essential for their restoration

The first step in peatland restoration is rewetting (i.e. raising the watertable) and then maintaining the watertable at levels as close to the surface as possible throughout the year. Changes in water content (particularly groundwater levels and soil moisture) and disturbances (e.g. deforestation, fire and new canals, logging tracks and roads) should be monitored at the landscape scale over time to assess success (which is necessary for planning further restoration) and plan interventions designed to reduce fire risk and prevent other unsustainable human activities (FAO, 2020c; Wösten, Rieley and Page, 2008).

Peatland rewetting is done by blocking the drainage system, either with built structures or by infilling (Andersen *et al.*, 2017; Strack, 2008). The various peatland restoration methods all have pros and cons related to costs, the machinery required, durability and effectiveness. Local transport needs should also be considered because, in some cases, canal drainage systems facilitate the movement of boats and thereby economic activities. The most widely used means for blocking drains is compacted peat because of its persistence in peak flows and the wide availability and sustainability of the raw material (FAO, 2020c).

The restoration of peatland forests commenced in boreal regions several decades ago, but the restoration of tropical peatland forests is still nascent, and there are unanswered questions about (among other things) the feasibility of rewetting large areas.

Revegetation programmes involving communities should prioritize peatland forest species that can provide economic benefits (Giesen and Sari, 2018). In any case, dryland species, and species associated with high rates of evapotranspiration, should be avoided to prevent the drying out of peatlands. Revegetation activities should be appropriate for the level of degradation; for example, the more an area has been affected by fire and the further ecologically it is from a natural peatland forest, the more likely it will require replanting to complement natural regeneration. Given the fast rate of peat oxidation, compaction and subsidence, rewetting should be initiated rapidly. The longer the restoration process takes, the more expensive it will be to restore the ecosystem. There is a diminishing window of opportunity for reaping the full benefits of peatland restoration.

BOX 5.7

Enabling holistic peatland restoration in the boreal zone

Most of the drainage in Fennoscandia, the Russian Federation and the Baltic states has occurred in naturally tree-covered mires, and these types of ecosystems are now scarce (Joosten and Clarke, 2002). In Finland, most of the conversion of these peatlands was for forestry (Similä, Aapala and Penttinen, 2014), where draining started in the 1930s and protection interventions only began in the 1960s. However, if the drainage system is not maintained, wetter conditions can restart peat formation proceses, known as repaludification (Joosten & Clarke, 2002).

Various projects have focused on the restoration of drained peatlands in Finland, including the LIFE projects of the European Union (as part of the Natura 2000 network), which began in the early 2000s. Finnish peatland restoration know-how has accumulated over more than 25 years (Similä *et al.*, 2014). About 20 000 ha of drained peatlands were restored as natural areas in 1989–2013 by blocking and damming ditches and felling and removing trees grown after draining (Similä, Aapala and Penttinen, 2014). Peatland forest patches were restored to increase the amount of deadwood, which favours biodiversity. The overall goal is to restore the natural hydrology of spruce mires and other forest-covered mires, reduce fragmentation and increase biodiversity. A national monitoring network for restored peatlands was developed as part of the Boreal Peatland LIFE project in 2010–2014, including the impacts of peatland restoration on hydrology (e.g. watertable level and quality) and biodiversity (vegetation, birds, butterflies and dragonflies) (Similä *et al.*, 2014). An extensive awareness-raising campaign shared information on peatlands – their flora, fauna, protection and restoration – with people both online and on-site (e.g. information boards and mire exhibitions, with duckboards enabling easy and safe access for visitors).

A rare orchid, Cypripedium calceolus, *in a rich fen in Finnish Lapland, Oulanka National Park, Finland*

Improve the enabling environment for peatland ecosystems. To enable successful peatland forest restoration, context-specific management measures (such as species suitability) may need to be tried and tested in a given project area, value chains and markets may need to be developed, and regulatory frameworks (including land tenure) may need to be adapted. Regulations and laws at the national and subnational levels may be needed to disincentivize unsustainable uses such as deforestation and peatland drainage (e.g. Silvius and Suryadiputra, 2002).

Integrating peatland ecosystems in existing institutional, policy and legislative frameworks, and harmonizing them, is a key challenge for many countries. Improving land and water management in peatland forests over the long term often requires subsidies for restoration and viable, drainage-free livelihoods. The integration of peatlands into national land-use plans and systems for monitoring land-use change will support the process but requires technical collaboration and resources. In countries where large areas of peatlands have been drained, the inclusion of peatlands in strategies to reduce the risks associated with, for example, fire, flooding, erosion and drought would allow better-targeted decision-making, taking into consideration the unique characteristics of these ecosystems and the local context.

Peatland forest research needs and knowledge gaps

Peatland forests cannot be protected or sustainably managed over the long term without proper mapping; therefore, the mapping of the extent, peat depth and status of peatland forests, including areas with shallow peat, should be a priority (FAO, 2020c). The definition of peatland varies between countries, and such definitions should be based on sufficient scientific evidence to ensure that no large areas of peat are omitted. In areas of degraded peatlands, drainage prevention techniques should be explored and improved to restore such systems to as close as possible to pristine. This is imperative in areas facing strong pressure from agriculture and other competing land uses to limit any further damage to these delicate hydrologic systems and to foster recovery and ecosystem integrity.

More research and development is needed in a wide range of areas, such as appropriate timber-extraction techniques as part of the sustainable use of peatland forests and alternative wet crops and products to replace crops grown on drained and deforested peatlands.

TROPICAL MONTANE CLOUD FORESTS

Key points

- Tropical montane cloud forests (TMCFs) are among the most valuable terrestrial ecosystems for their role in the hydrologic cycle because they influence the amount of available water and regulate surface and groundwater flows in watersheds while maintaining high water quality.
- The high water yield of TMCFs arises from their location in areas with high rainfall, additional inputs of cloud-water capture by canopies, and low evaporative losses.
- TMCFs are rare; area estimates range from 1 percent to 14 percent of tropical forests globally. Approximately 55 percent of the original area of TMCFs has been lost.
- The conservation of remnant mature TMCF forests needs strengthening and their conversion to agricultural land uses avoided.
- Low-intensity selective logging in secondary TMCFs conforming with low-impact logging guidelines is strongly recommended to mitigate the deleterious effects of logging on soils, water yields and biomass.
- In restoring TMCFs, efforts should be made to plant mixtures of native water-use-efficient species.
- Payment schemes for the water services of TMCFs could help compensate landowners, maintain forest cover and counteract deforestation and water scarcity.
- Research is needed to better understand the hydrologic impacts of climate change on TMCFs.

The generally high water yields in tropical montane cloud forests (TMCFs) (Box 5.8) arise because of their location in areas with high rainfall, additional inputs of cloud-water capture by canopies ("fog stripping"), and low evaporative losses (Hamilton, Juvik and Scatena, 1995; Bruijnzeel, 2001). Catchment water yields typically increase from lower to upper montane forests, reflecting concurrent increases in incident precipitation and decreases in evaporative losses at higher elevations (Bruijnzeel, 2005). The presence of clouds not only increases water inputs from fog capture but also reduces losses via evaporation because of the lower radiation and higher atmospheric humidity they generate (Bruijnzeel, Mulligan and Scatena, 2011).

Cloud interception is highly seasonal in many regions and becomes a more crucial component of total water budgets during dry seasons and therefore in sustaining flows in those dry periods. In comparison with montane forests unaffected by fog or low clouds, waterflows from TMCFs tend to be more stable during extended periods of low rainfall (Bruijnzeel, 2001).

BOX 5.8

What are tropical montane cloud forests?

Tropical montane cloud forests receive frequent moisture inputs from fog and mist. There are multiple classifications of such forests, but the broadly adopted definition is "forests that are frequently covered in cloud or mist" (Hamilton, Juvik and Scatena, 1995), highlighting the importance of clouds for these ecosystems. Tropical montane cloud forests are among the most valuable terrestrial ecosystems for their role in the hydrologic cycle because they influence the amount of water available and regulate surface and groundwater flows in watersheds while maintaining high water quality.

TMCFs are important in the protection of soils because they are often found on steep slopes, which tend to be highly susceptible to erosion and mass movement if forests are removed (Bruijnzeel, 2004). TMCFs are also a priority hotspot for biodiversity conservation because of their high species richness and endemism (Hamilton, Juvik and Scatena, 1995; Beck *et al*., 2008; Bendix *et al*., 2013), especially for epiphytes (Gentry and Dodson, 1987) and insects (Brehm *et al*., 2005).

Threats to tropical montane cloud forest–water relationships

Because TMCFs develop in particular climatic and topographic conditions, their spatial distribution is naturally fragmented and restricted in extent. They are relatively rare: estimates of the area of TMCFs range from 1 percent to 14 percent of the total area of tropical forests worldwide (Bruijnzeel, Mulligan and Scatena, 2011; Mulligan, 2011). Of all mapped TMCFs, 43 percent are in Asia, 41 percent are in the Americas and 16 percent are in Africa. (Mulligan, 2011; Hamilton, Juvik and Scatena, 1995).

There is a lack of up-to-date data on change in the area of TMCFs; it has been estimated that 55 percent of original cover had been lost by 2000 (Mulligan, 2011). An annual deforestation rate of 1.55 percent has been estimated for tropical montane forests (including TMCFs) in Latin America (Armenteras *et al*., 2017). Conversion to agriculture and cattle grazing are the main drivers of deforestation in TMCFs (Scatena *et al*., 2011; Aide, Ruiz-Jaen and Grau, 2010; Armenteras *et al*., 2017). Large areas of pasture created on land formerly occupied by TMCFs have been abandoned worldwide, however, giving rise to secondary forests (Scatena *et al*., 2011; Mulligan 2011). Overharvesting and invasive grasses and ferns such as bracken are also significant threats (Aide, Ruiz-Jaen and Grau, 2010). Unplanned selective logging usually involves the exploitation of high-value timber species (e.g. in the families Juglandaceae, Lauraceae and Podocarpaceae), causing forest degradation and thereby increasing the probability of conversion to agricultural land uses.

Another significant threat to TMCFs is climate change: because of their restrictive climatic requirements and fragmented distribution, TMCFs are highly vulnerable to increased temperatures and alterations in patterns of precipitation and cloud distribution (Feeley *et al*., 2013; Lutz, Powell and Silman, 2013). Alterations in the altitude at which cloud condensation occurs and increased evapotranspiration – both possible due to global warming – would reduce the area of montane land directly exposed to clouds (Still, Foster and Schneider, 1999; Bruijnzeel, 2004). Recent projections indicate that cloud immersion could shrink or dry out 57–80 percent of neotropical TMCFs (Helmer *et al*., 2019). This would make TMCFs more susceptible to fire, disease and invasive species, reducing ecosystem resilience. The impacts of such alterations on TMCF water cycles are likely to be considerable, causing reductions in water availability in lower parts of watersheds; these impacts are little studied, however.

The conversion of TMCFs to annual crops and pasture causes an increase in the volume of surface runoff because soil compaction reduces infiltration capacity (Bruijnzeel, 2004). Although forest transpiration is significantly reduced, causing an overall increase in the streamflow (Bruijnzeel, 2005), such extra soil water does not compensate for the loss of soil infiltration capacity; runoff peaks increase during wet seasons and streamflows decline in dry seasons (Bruijnzeel, 1989; 2004). Forest clearing also reduces tree and epiphyte interception of rain and fog water (Bruijnzeel, 2004). The replacement of mature TMCFs with pastures has decreased water input in the Venezuelan Andes and eastern central Mexico (Ataroff and Rada, 2000; Holwerda *et al*., 2010; Muñoz-Villers and López-Blanco, 2008).

The abandonment of agriculture and livestock grazing in former TMCFs enables the development of secondary TMCFs, but these younger forests capture less water from rainfall and fog than mature forests (8 percent versus 17 percent in Mexico; Holwerda *et al*., 2010). Nevertheless, water yields are higher in secondary TMCFs, likely due to the higher canopy storage capacity of mature TMCFs. This, in turn, is because leaf area per unit ground surface area and epiphyte biomass are higher in mature TMCFs, contributing to the capture and storage of water (Holwerda *et al*., 2010; Köhler *et al*., 2011). Epiphytes are abundant in the canopies of TMCFs, possess a high capacity for water storage, and can release stored water slowly (Veneklaas *et al*., 1990). Despite their considerable water-storage capacity, however, the contribution of non-vascular epiphytes to overall rainfall interception is relatively low (6 percent; Hölscher *et al*., 2004). Leaf and epiphyte surface area reductions in secondary TMCFs decrease canopy water retention and evaporation, thereby increasing throughfall and stemflow inputs to soil (Nadkarni *et al*., 2004; Ponette-González, Weathers and Curran, 2010). Overall, however, Muñoz-Villers *et al*. (2012) found very similar hydrologic behaviour between a 20-year-old secondary TMCF and a mature TMCF in Mexico, showing the value of natural regeneration in the recovery of hydrologic functioning in TMCFs.

Soil erosion is a potentially significant impact of any type of forest operation in the humid tropics (Bruijnzeel, 1992). The resultant input of sediments into rivers reduces water quality and channel capacity, the latter of which can increase the risk of flooding (Chappell *et al*., 2005).

Management of cloud forests for water services

Given their essential role in the hydrologic cycle and as reservoirs of biodiversity, the management of TMCFs should aim to integrate multiple ecosystem services, including those related to water, soil and biodiversity. Management objectives may vary widely, from conservation to timber production, depending on the socio-economic and biogeographic context.

Ideally, all old-growth TMCFs would be protected because of their valuable ecosystem functions. This is only likely to occur, however, when pressure from other land uses is low or the enforcement of conservation measures is high, which is not the case in many areas. Unplanned selective logging is common among communities in or near TMCFs (Hölscher *et al*., 2010; Toledo-Aceves *et al*., 2011), but the impacts of this exploitation on water services have not been evaluated systematically. The use of TMCFs for commercial timber production is rare, no doubt because of the low commercial timber volumes and grow rates; moreover, the steep slopes of most TMCFs make timber extraction complicated and costly.

Low-impact logging should be applied in any harvesting operations in TMCFs, adapting its key elements of pre-logging planning; the maintenance of vegetated stream buffer zones; the timing of operations to avoid very wet periods and minimize soil compaction; and post-harvesting measures such as soil bunding and the installation of cross-drains on skid trails (Cassells and Bruijnzeel, 2005). Directional felling is also an important measure to minimize the risk to workers and damage to harvested and potential crop trees.

The minimization of disturbances in forests on very steep slopes is crucial. Means for reducing the impact of log extraction in TMCFs by reducing the need for skid trails (which can have substantial impacts on hydrology and increase erosion) include using horses for skidding; mobile sawmills or chainsaw frames to mill logs on site; and cable yarding (Günter *et al*., 2008).

Given the protective functions of TMCFs for soils and their role in the hydrologic cycle, permanent forest cover and forest structure should be maintained wherever possible (Aus der Beek and Sáenz, 1992). Polycyclic selection systems will best enable this, and clearfelling should be avoided in TMCFs. Ensuring the financial competitiveness of selective timber harvesting compared with other land uses may require a PWS scheme (Günter, 2011; Knoke *et al*., 2014).

PWS schemes have been popular in TMCFs as a means to compensate landowners and thereby reduce deforestation and water scarcity. For PWS to be effective, however, the financial benefits must be comparable with the opportunity costs associated with not converting to pasture or other land-use activities; Box 5.9 presents a case study in Mexico, and there have also been promising experiences in Bolivia (Plurinational State of), China, Colombia, Costa Rica, Ecuador, the Dominican Republic and Viet Nam; there are few in Africa, however (Asquith, Vargas and Wunder, 2008; Bösch, Elsasser and Wunder, 2019).

BOX 5.9

A payment scheme for ecosystem services provided by cloud forests in Mexico

Programmes making payments for water services in Mexico began in central Veracruz, where a combination of high deforestation rates, associated losses of water services such as the regulation of water quality and flood–drought cycles, and climate change made the sustainable management of water and forest resources a top priority for decision-makers. Table 5.1 summarizes the region's two main programmes, one in the Gavilanes watershed (providing 90 percent of the water supply for the city of Coatepec) and the other in the Pixquiax watershed (providing 40 percent of the water supply for the city of Xalapa), both co-financed by the National Forest Commission, local water operators and municipal governments.

TABLE 5.1

The strengths and weaknesses of two payment schemes for water services in Veracruz, Mexico

	Fidecoagua (Coatepec)	**Acuerdos por Nuestra Agua (Xalapa)**
Strengths	First payment scheme for ecosystem services in Mexico (2002) Stable financing, with a fee of MXN 2 included in water bills Novel "Adopt a Hectare" programme to conserve shade-coffee farms The target watershed is entirely within one municipality	Started by Sendas, a non-governmental organization, in 2005 Novel combination of cash payments and technical assistance to promote sustainable alternatives (Nava-López *et al.*, 2018) Science-based approach used to concentrate payments in hydrologic priority areas Long-term monitoring using citizen science Significantly stronger alliances and social network with greater impacts on landowner well-being (Torres-Pérez, 2018)
	Both programmes monitor deforestation, which has declined significantly in areas where the schemes operate (by 5.5 percent compared with areas where they don't operate), with no detectable leakage (von Thaden *et al.*, 2019)	
Weaknesses	Government-run, with limited growth and creativity Few efforts to partner with other sectors	Lack of legal framework, making local-government funding and support unstable Operation across multiple municipalities is politically difficult
	Both programmes focus more on water providers than on downstream users, whereas the latter could help ensure long-term political support Both programmes have poor additionality, with only 38.5 percent of payments made in areas of high deforestation risk (von Thaden *et al.*, 2019)	

Source: Compiled by Robert H. Manson.

In degraded TMCFs, efforts may be needed to restore structure and function. Passive restoration – that is, restoration involving no active intervention (although it requires the diminution or exclusion of the factors that caused the degradation, such as cattle grazing) – natural processes will determine forest structure and function. This type of restoration requires less investment than active restoration, but its effectiveness depends on the intensity and type of the previous land use and the quality of the surrounding landscape. For example, the natural regeneration of tropical montane forest on abandoned pastures can be limited by low seed arrivals and the absence of seed dispersers (Aide, Ruiz-Jaen and Grau, 2010); competition with pioneer species (e.g. grasses and ferns; Aide, Ruiz-Jaen and Grau, 2010); and unfavourable microhabitats (e.g. due to high solar radiation, soil compaction, erosion and infertility; Holl, 1999). Moreover, whereas relatively high tree diversity might be achieved through passive restoration in TMCF landscapes (Muñiz-Castro, Williams-Linera and Benayas, 2006; Trujillo-Miranda *et al*., 2018), varying rates of recovery of other important taxa have been observed for epiphytes and insects (Köhler *et al*., 2011; Adams and Fiedler, 2015). Additionally, the slow recovery of provisioning services achieved through passive restoration increases the risk of conversion to agricultural land. Lower rates of vegetation recovery have been reported with increasing distance from mature TMCFs (Muñiz-Castro, Williams-Linera and Benayas, 2006; Trujillo-Miranda *et al*., 2018). Active restoration should be encouraged, therefore, in landscapes with few, small or degraded remnants of TMCFs or with high forest-conversion pressure.

Various active restoration strategies can be pursued depending on management goals and the economic, social and environmental context. The most common approach is to establish plantations in deforested areas, which, by increasing forest cover, may improve infiltration and runoff and help reduce erosion, sedimentation and downstream flooding. Efforts should be made to establish mixes of native species to restore some of the original tree diversity and thereby increase forest resilience and to encourage the growth of native plant species in plantation understoreys (Aide, Ruiz-Jaen and Grau, 2010; Liu *et al*., 2018; Trujillo-Miranda *et al*., 2018). In landscapes with few or distant patches of existing TMCFs, practices such as the installation of bird perches, direct seed-sowing and soil translocation may help in establishing nuclei of native vegetation (Boanoares and de Azevedo, 2014).

The design of TMCF restoration should take into account the potential for altered biophysical conditions associated with climate change and, where necessary, accommodate the potential future redistribution of tree species to higher altitudes and latitudes. Shifts in the distribution of TMCF tree species towards higher elevations – and increased mortality at lower elevations – in response to increased temperatures have been observed in Colombia, Costa Rica and Peru (Feeley *et al*., 2011; 2013; Duque, Stevenson and Feeley, 2015). The assisted migration of plant species via enrichment plantings using shade-tolerant TMCF tree species at sites above the reported limit of their present distribution has shown early promise as a climate-change mitigation strategy (García-Hernández *et al*., 2019).

Cloud forest research needs and knowledge gaps

Given the large diversity of TMCF types and the lack of data in most countries, it is essential to monitor changes in TMCF cover and to analyse drivers to improve understanding of the causes of TMCF loss and ways to reduce this.

Assessments are needed of the relationships between change in TMCF cover and water services under various climatic regimes at the watershed scale. A priority should be the identification of causes of decreases in dry-season flows and the development of approaches for restoring hydrologic function. More investigation is also needed on the effects of changes in TMCF water cycles on erosion and landslides. Increased knowledge on this aspect would support the development of integrated water resource management plans at the watershed scale.

There is a lack of knowledge on the relationship between biodiversity and water cycles in TMCFs. Managing for water services should not mean trade-offs with biodiversity conservation. Indeed, high biodiversity in TMCFs could increase ecosystem resilience in the face of altered patterns of precipitation and cloud formation. TMCF tree species differ in their tolerance and responses to environmental change; although some species might be more vulnerable, others could be more successful as conditions change (Feeley *et al*., 2011, 2013; Toledo-Aceves *et al.*, 2019). Studies of long-term catchment-scale hydrologic changes associated with selective logging in TMCFs under various felling intensities, forest ages and structures would generate valuable information for regional water resource planning and TMCF conservation.

Additional research is needed to better understand the hydrologic impacts of climate change on TMCFs: areas for further study include quantifying associated changes in fog-stripping, forest water use and, ultimately, streamflow. More knowledge is also needed on how changes in the composition of tree communities as a result of increasing temperatures might affect water yield. Such knowledge is essential for the design of effective climate-change mitigation measures. Adaptive management requires flexibility, with the knowledge generated from monitoring and evaluation used to modify management practices to ensure the ongoing optimal provision of water services.

Cloud forest in Veracruz, Mexico

DRYLAND FORESTS

Key points

- Drylands support the livelihoods of millions of people globally.
- Dryland forests and trees survive and grow on limited water resources, but they also influence various components of the water cycle and water availability.
- Management strategies for dryland forests such as canopy opening, pruning and species selection might help combat local water scarcity by increasing soil and groundwater recharge.
- Given the complexity of multi-objective management and the intrinsic variability of dryland forests and other dryland systems with trees, more effort is needed to quantify and value the goods and ecosystem services produced in these systems and the management options available.
- The reuse of wastewater can help in maintaining dryland ecosystem services in the face of water scarcity.

Drylands are biomes characterized by water scarcity; they can be defined as land where the ratio of mean annual precipitation to total annual potential evapotranspiration (known as the aridity index) is below 0.65. The fragile balance between water input and consumption means that drylands face a wide range of threats and challenges, including low productivity, water stress, climate variability and change, a high risk of natural disasters and hazards, marginality and remoteness, migration, and population pressure (Schwilch, Liniger and Hurni, 2014). Drylands host around 2 billion people and cover 41 percent of the world's land surface (Millennium Ecosystem Assessment, 2005c); thus, a significant proportion of the global population relies directly on dryland ecosystem services for their livelihoods and income. Seventy-two percent of drylands are in low- and middle-income countries (Millennium Ecosystem Assessment, 2005c), which, added to their often low productivity, may explain the relatively low attention they attract.

Forests (as defined in FAO, 2020a) cover 17.7 percent (1 079 million ha) of the total dryland area globally, making such forests (Box 5.10) similar in extent to tropical moist forests (1 156 million ha) (Bastin *et al.*, 2017).

BOX 5.10

What are dryland forests?

Most dryland forests and other wooded lands are in the semiarid and dry subhumid zones (Bastin *et al.*, 2017; FAO, 2019). Dryland trees and shrubs have developed effective functional adaptations to cope with the combination of high temperatures and water scarcity, all of which have clear benefits for plant functioning and survival but come with costs related to water use, carbon gain and leaf cooling (Peguero-Pina *et al.*, 2020). Such adaptations include modifications to leaf angle, size and shape and transpiration rates designed to (for example) reduce the absorbed light energy, enhance the ability for heat dissipation, and reduce plant water consumption (Peguero-Pina *et al.*, 2020).

Woody vegetation and forests in drylands generate diverse ecosystem services to support the livelihoods of many people, such as the provision of food, fodder, fuel, fibre and genetic resources; water purification; regulation of the hydrologic cycle; flood mitigation; erosion minimization; the maintenance of soil fertility; the provision of habitats for fauna and flora; and contributions to cultural identity and diversity, cultural landscapes and heritage values (Shvidenko *et al.*, 2005; Jindal, Swallow and Kerr, 2008; Chidumayo and Gumbo, 2010; Asbjornsen *et al.*, 2014; Sinare and Gordon, 2015).

In addition to the 1 079 million ha of forests in drylands, trees occur in another 583 million ha of other wooded lands (defined as a tree canopy cover of 5–10 percent), as well as outside forests and other wooded lands in croplands, urban areas and other lands (FAO, 2019). Almost 30 percent of the area under croplands in drylands is estimated to have at least some tree cover (FAO, 2019). When forests, other wooded lands and trees outside forests are taken into account, trees are present on 2 billion ha of drylands, which is 32 percent of the total dryland area. Africa has the largest area of drylands (32 percent of the world total), followed by Asia, North America, Oceania, South America and Europe (FAO, 2019).

Threats to dryland forest–water relationships

Dryland forests and trees face climatic constraints both due to heat, and especially due to changes in water availability (IPCC 2021) that increase the importance of soil processes and properties in the regulation and magnitude of water-related issues, especially those concerned with water storage (e.g. soil depth, infiltrability, deep water storage and erosion). Dryland forests and trees face several abiotic and biotic threats, such as wildfire, insect pests and severe drought (Petrie *et al*., 2017), that may reduce their capacity to persist in their current geographic ranges and to colonize new habitats (Bell, Bradford and Lauenroth, 2014; Rehfeldt *et al*., 2014). The persistence of dryland forests and trees in the twenty-first century, therefore, will depend increasingly on tree regeneration – which, however, was only episodic in the twentieth century and limited to infrequent periods of favourable climatic and environmental conditions (Savage, Brown and Feddema, 1996; Mast *et al*., 1999; Brown and Wu, 2005).

According to climate-change projections, the occurrence of favourable climatic conditions for dryland trees and forests will be even less frequent in the future, thus diminishing regeneration potential and increasing the threat at the ecosystem scale. Thus, although climate-change projections indicate an expansion of dryland biomes of 11–23 percent by the end of the century (Feng and Fu, 2013; Huang *et al*., 2017), the climate-driven niche space for dryland forests and trees will likely decline, and climate change will alter the geographic ranges of tree species (Coops, Waring and Law, 2005; van Mantgem *et al*., 2009; Williams *et al*., 2013), giving rise to more arid dryland ecosystems This will significantly alter the local hydrology by decreasing soil infiltration capability at the same time that overland runoff and soil water evaporation increase (D'Odorico *et al.*, 2006). Thus, practices such as farmer-managed natural regeneration and assisted natural regeneration are key to ensuring the persistence of trees and forests in drylands and their water relationships.

Management of dryland forests and trees for water services

Understanding water as a limiting factor, and its many interlinkages with soil, vegetation and climate, is essential for ensuring the provision of goods and ecosystem services from dryland forests and trees. Water availability affects not just the production of particular goods and services but also their long-term sustainability; water, therefore, is the key element enabling and maintaining the provisioning of other goods and ecosystem services. Thus, water is the most crucial resource for the socio-ecological resilience of dryland forests and must constitute a quantitative basis of any management approach (Falkenmark, Wang-Erlandsson and Rockström, 2019).

The importance of water in dryland forests and other ecosystems with trees requires ecohydrologic forest management that determines the trade-offs between water and vegetation (del Campo *et al*., 2019a). Among other things, this means modifying forest and tree cover and species composition according to the local balance between water availability and consumption. In this sense, strategies such as canopy opening, pruning and species selection can be effective in combating water scarcity (by increasing soil and groundwater recharge) while also increasing climate change resilience and adaptation. Ecohydrologic forest management in drylands can perform a dual service – that is, increase the resilience of forests and trees in the face of drought and other water-related disturbances and enhance water security for people. The optimum management intensities and strategies (e.g. involving thinning, pruning and planting) are likely to vary with ecosystem characteristics, even within the same catchment or region (del Campo *et al*., 2019a).

Canopy manipulation. Del Campo *et al*. (2019b) reported a significant increase in tree water availability in a low-biomass holm oak forest after removing 33 percent of the standing biomass; no increase in soil erosion was observed for five years following this thinning. A significant decrease in climate vulnerability was reported in a marginal Aleppo pine plantation after thinning, which also significantly decreased the fire risk while increasing the water budget (García-Prats *et al*., 2016). Thus, managing forest biomass to shape tree–soil–water relationships can increase local water availability and consequently tree growth and vigour (by reducing tree competition) and forest resilience and reduce fire risk.

Agroforestry systems. Trees consume considerably more water than shorter vegetation (Zhang, Dawes and Walker, 2001), and increases in tree cover have often been discouraged in drylands because of perceived negative impacts on local water availability (Jackson *et al*., 2005). But this views trees as mere water consumers, ignoring the many other mechanisms by which trees modify water availability. A more nuanced view of the impact of trees on water availability recognizes several opportunities to increase water security in drylands by increasing tree cover (Sheil and Bargués Tobella, 2020). One of these arises from the benefits of moderate tree cover on groundwater recharge (Box 5.11; Ilstedt *et al*., 2016). Landscapes with open tree cover – for example where trees are integrated on farms in agroforestry systems – can improve local water availability compared with similar landscapes without trees where soil degradation has reduced infiltration. Thus, promoting and maintaining agroforestry and encouraging active ecohydrologic management that takes into consideration appropriate tree species, optimum tree cover based on local environmental contexts, and grazing control has the potential to improve water security in drylands.

Reusing water. The reuse of treated wastewater in drylands is a necessary strategy in the face of water scarcity. Using treated wastewater from agroforestry systems and domestic sewage to irrigate forests or feed natural or constructed wetlands can enhance dryland ecosystem services, including by improving water quality. The sustainable irrigation of dryland forests with treated wastewater is a promising strategy for increasing forest resilience, decreasing fire risk (both rate of spread and intensity), and increasing the provision of forest goods and ecosystem services (e.g. through cooling effects, carbon sequestration and biomass production). Creating wetlands to emulate the functions of natural wetlands for human needs (Haberl *et al*., 2003) implies building ecosystems capable of recycling nutrients, purifying water, attenuating floods, maintaining streamflows, recharging groundwater and enhancing the livelihoods of local people by providing (for example) fish, drinking water, fodder, biofuel and ecosystem services. The design of constructed wetlands depends on climatic conditions and the volume of treated wastewater. Unless properly treated, sewage can constitute a major environmental and health hazard in drylands; constructed wetlands as the major component of sewage treatment systems can help solve a pollution problem while producing good-quality water (Tencer *et al*., 2009). Constructed wetlands can also be integrated into agricultural and fish production systems, where products are usable or can be recycled for optimal efficiency; communities can realize economic and ecological benefits from such systems (Avellán and Gremillion, 2019).

The establishment of forest plantations is another common strategy in drylands, providing a means for soil conservation while producing products such as fodder, fruit and woodfuel (Jama, Elias and Mogotsi, 2006). Forest plantations can reduce erosion,

BOX 5.11

Agroforestry systems – the importance of tree density

Recent studies in the agroforestry parklands of Saponé in central Burkina Faso show that trees can play a significant role in improving the recharge of deep soil water by enhancing soil infiltration capacity and preferential flow (Bargués-Tobella *et al.*, 2014; Bargués-Tobella *et al.*, 2020; Ilstedt *et al.*, 2016). Ilstedt *et al.* (2016) showed that the amount of soil water drainage collected at 1.5 m depth peaked in areas at the canopy edges of trees and decreased both towards tree stems and towards the centre of adjacent open areas. Findings from a modelling exercise based on these results and additional data on tree water use indicated that, in this system, groundwater recharge is maximized by intermediate tree cover (Ilstedt *et al.*, 2016). On sites where tree density is less than this optimum tree cover, increasing tree cover will increase groundwater recharge because the positive effects on soil hydraulic properties of the additional trees exceed their additional water use (evapotranspiration losses). Above the optimum, increased evapotranspiration losses will outweigh the positive impacts of trees, leading to reduced groundwater recharge. The optimum tree density for groundwater recharge varies depending on local conditions, but it is likely that moderate tree cover will improve local water availability over wide areas of drylands. These findings suggest a more nuanced view of the role of tree cover in water availability in drylands. Moreover, they highlight the enormous potential for improving water availability through the management of tree-canopy cover and additional practices such as species selection, pruning and grazing control. At the catchment scale, Suprayogo *et al.* (2020) found that, in an agroforestry watershed in Rejoso, East Java Province, Indonesia, thresholds of infiltration "friendliness" exist between systems that are mostly "agro" and those that are mostly "forest", but higher tree-cover systems are desirable.

mitigate dust storms and lessen the silting of streams. In some circumstances, they may also reduce streamflow due to higher water use, with potentially severe consequences for water management, water security and the overall ecosystem (Mátyás, Sun and Zhang, 2013).

Boswellia socotrana *in a dryland forest, Socotra, Yemen*

In some cases, such as that described by Lima *et al.* (1990), production-oriented plantations in drylands have significantly decreased water resource availability. In establishing and managing forest plantations in drylands, the primary objective should be to provide water services; wood production, biodiversity conservation and carbon sequestration may occur as co-benefits, but forest plantations are most likely to be sustainable when designed to reduce soil erosion, regulate water fluxes and protect reservoirs and other infrastructure from siltation (del Campo *et al*., 2020).

Dryland forest research needs and knowledge gaps

Ensuring the sustainability of ecosystem services and improving people's livelihoods in drylands is crucial for achieving the Sustainable Development Goals, with many indicators of human well-being and development lower in drylands than in other regions (Millennium Ecosystem Assessment, 2005c). The most critical trade-offs in dryland forests are all related to the allocation of water (Birch *et al*., 2010). Understanding the role of water in dryland forests is essential, therefore, and more research is needed on how different management practices in forests, woodlands and other systems with trees modify tree–water relationships and overall water availability. For example, identifying tree species that best promote soil hydrologic functioning and use relatively low quantities of water would help guide ecohydrologic tree and forest management in drylands, as would determining the level of tree cover that maximizes groundwater recharge. A science-based, water-centred approach to silviculture is urgently needed.

Acknowledging that dryland forests provide a wide range of ecosystem services is insufficient to encourage their conservation and sustainable management. It is essential to quantify the value of these ecosystem services and to communicate the findings. The few existing studies conducted in dryland forests have mostly been short-term and small-scale (Wangai, Burkhard and Müller, 2016). The economic valuation of water services in dryland forests would help in the development of payment schemes to compensate landholders for providing these services and internalize the positive externalities offered (Salzman *et al*., 2018).

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FAO FORESTRY PAPER

A guide to forest–water management

Water security looms as a major planetary challenge. Many people worldwide already lack adequate access to clean water, and pressure on water resources is increasing as populations grow, ecosystems are degraded and the climate changes.

Forests and trees are integral to the global water cycle and therefore vital for water security; they regulate water quantity, quality and timing and protect against erosion, flooding and avalanches. Forested watersheds provide 75 percent of our freshwater, delivering drinking water to more than half the world's population.

The purpose of *A Guide to Forest–Water Management* is to improve the global information base on the protective functions of forests for soil and water. It reviews emerging techniques and methodologies, provides guidance and recommendations on how to manage forests for their water services, and offers insights into the business and economic cases for this. The guide pays special attention to four ecosystems that are crucial for forest–water management – mangroves, peatland forests, tropical montane cloud forests and dryland forests.

A Guide to Forest–Water Management finds that both natural and planted forests offer cost-effective solutions to water management while providing considerable co-benefits, such as the production of wood and non-wood goods, climate change mitigation, biodiversity conservation and cultural services. The task of ensuring global water security is formidable, but this report provides essential guidance for water-centred forestry as a means of increasing the resilience of our precious water resources.

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