Forest and Water on a Changing Planet: Vulnerability, Adaptation and Governance Opportunities

A Global Assessment Report

Editors: Irena F. Creed and Meine van Noordwijk

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CPF
Collaborative Partnership on Forests

IUFRO
Interconnecting Forests, Science and People



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Preface

Since its establishment in the year 2007, the Global Forest Expert Panels (GFEP) initiative of the Collaborative Partnership on Forests (CPF) has been effectively linking scientific knowledge with political decision-making on forests. GFEP responds directly to key forest-related policy questions by consolidating available scientific knowledge and expertise on these questions at a global level. It provides decision-makers with the most relevant, objective and accurate information, and thus makes an essential contribution to increasing the quality and effectiveness of international forest governance.

This report entitled "Forest and Water on a Changing Planet: Vulnerability, Adaptation and Governance Opportunities" presents the outcomes of the sixth global scientific assessment undertaken in the framework of GFEP. All assessment reports are prepared by internationally recognised scientists from a variety of biophysical and social science disciplines. The publications are presented to stakeholders across relevant international policy fora. In this way, GFEP supports a more coherent policy dialogue about the role of forests in addressing the broader environmental, social and economic challenges reflected in the United Nations Sustainable Development Goals (SDGs).

The current report reflects the importance of integrated action towards ensuring access to water for all and sustaining life on land. The provision of clean water is the most basic ecosystem service necessary for life on earth. Yet, growing demand for water caused by an increasing human population, combined with adverse effects of climate change, are creating unprecedented challenges for sustainable development.

Forests influence water resources in multiple ways, and at multiple levels. Whereas the interplay between forests and climate is regularly considered in decision-making, that between water and forests remains under-represented. Today, the fact that the world has mobilised around the seventeen SDGs, all of which have a connection to water, provides a crucial argument for paying more attention to the forest-water link.

While the international community agreed the SDG framework based on moral principles, science is essential for developing the policies and practices required for achieving the related targets. Scientific reports like the one in hand are important tools for supporting policymakers and stakeholders in their ambition to ensure sustainable development and to advance the implementation of the United Nations 2030 Agenda.

I would like to thank the Co-Chairs of the Global Forest Expert Panel on Forests and Water, Irena F. Creed and Meine van Noordwijk, GFEP Coordinator Christoph Wildburger, GFEP Editor Stephanie Mansourian, and GFEP Project Manager Andre Purret for their excellent work in guiding the assessment process and in leading the development of this publication. It is my sincere hope that those with a responsibility for implementing the SDGs at all levels will find this report, and its accompanying policy brief, a useful source of information and inspiration.

Alexander Buck

Alexander Buck
IUFRO Executive Director

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Co-Chair

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Stephanie Mansourian
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Contents

Acronyms, Units and Symbols

AWY	Annual Water Yield
CDP	Carbon Disclosure Project
COP	Conference of the Parties
CPF	Collaborative Partnership on Forests
CSR	Corporate Social Responsibility
DR	Democratic Republic
DWS	Department of Water and Sanitation
ECA	Equivalent Clear-cut Area
ENMOD	Environmental Modification Convention
ENSO	El-Niño Southern Oscillation
EPA	Environmental Protection Agency
ES	Ecosystem Service
ET	Evapotranspiration
FAO	Food and Agriculture Organization of the United Nations
FLR	Forest Landscape Restoration
FSA	Forestry South Africa
FSC	Forest Stewardship Council
GDP	Gross Domestic Product
GFEP	Global Forest Expert Panels
GPFLR	Global Partnership on Forest Landscape Restoration
HDO	heavy water
HIV/AIDS	Human Immunodeficiency Virus/Acquired Immune Deficiency Syndrome
HKH	Hindu Kush – Himalaya
HRU	Hydrological Response Unit
ICIMOD	International Centre for Integrated Mountain Development
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
ITCZ	Intertropical Convergence Zone
LAAC	Licence Assessment Advisory Committee
LAI	Leaf Area Index
LULUCF	Land Use, Land Use Change and Forestry
MBI	Market-Based Instruments
MCPFE	Ministerial Conference on the Protection of Forests in Europe
MDBA	Murray-Darling Basin Authority
MEA	Millennium Ecosystem Assessment
MLG	Multi-Level Governance
MPB	Mountain Pine Beetle
NAO	Northern Atlantic Oscillation
NASA	National Aeronautics and Space Administration
NBI	Nile Basin Initiative
NDC	Nationally Determined Contribution
NDVI	Normalized Difference Vegetation Index
NELSAP	Nile Equatorial Lakes Subsidiary Action Programme
NEA	National Ecosystem Assessment
NTFP	Non-Timber Forest Product
NGO	Non-Governmental Organisation
NRC	National Research Council
NWA	National Water Act
ODA	Overseas Development Agency
PDO	Pacific Decadal Oscillation
PDR	People's Democratic Republic

PES	Payment for Ecosystem Service
PET	Potential Evapotranspiration
PNV	Potential Natural Vegetation
PROFOR	Program on Forests
PVC	Polyvinyl Chloride
PWE	Paired Watershed Experiment
PWS	Payment for Water Service
RECOFTC	The Center for People and Forests
REDD+	Reducing Emission from Deforestation and forest Degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries
RRI	Rights and Resources Initiative
RSPO	Roundtable on Sustainable Palm Oil
SDG	Sustainable Development Goal
SDL	Sustainable Diversion Limit
SE	Southeast
SES	Social-Ecological System
SFRA	Stream-Flow Reduction Activity
SMART	Specific, Measurable, Achievable, Realistic, and Timely
TEEB	The Economics of Ecosystem Services and Biodiversity
TEK	Traditional Ecological Knowledge
THED	Theory of Himalayan Environmental Degradation
TPW	Total Precipitable Water
UK	United Kingdom of Great Britain
UN	United Nations
UNCCD	United Nations Convention to Combat Desertification
UNESCO	United Nations Educational, Scientific and Cultural Organization
UNFCCC	United Nations Framework Convention on Climate Change
UNFF	United Nations Forum on Forests
UNFI	United Nations Forest Instrument
UNGA	UN General Assembly
UNSPF	UN Strategic Plan for Forests
US	United States of America
USD	US Dollar
VOC	Volatile Organic Compound
WARF	West African Rainforest
WEF	World Economic Forum
WFD	Water Framework Directive
WfW	Working for Water
WMD	Watershed Management Division
WUE	Water Use Efficiency

Units and symbols

The International System of Units (SI) is used in the publication.

MW	Megawatts
ha	hectare (100 ha = 1 km^2)
Mha	Millions of hectares
yr	year
C	carbon
CO_2	carbon dioxide
CO	carbon monoxide
NO_x	nitrogen oxides

Chapter 1
Forests, Trees and Water on a Changing Planet: A Contemporary Scientific Perspective

Coordinating lead authors: Irena F. Creed and Meine van Noordwijk

CONTENTS

1.1 Introduction

More than seven billion humans share the planet with approximately three trillion trees (Crowther et al., 2015), 46% less trees than at the start of human civilisation. Approximately 1.36 trillion of these trees exist in tropical and subtropical regions, 0.84 trillion in temperate regions and 0.84 trillion in the boreal region; overall nearly one-third are outside forests (Crowther et al., 2015). There is a wide variation in the ratio of trees to humans and whether or not this matters can be answered in many ways. Even so, we know that the majority of the four billion people facing severe water scarcity (Oki and Kanae, 2006; Rockström et al., 2014; Mekonnen and Hoekstra, 2016) live in areas where forests and trees outside forest are currently scarce.

Perhaps because the co-occurrence of forest and water is so common, water is rarely considered to be a priority in forest management. Forests and trees are important modulators of water flows (Vörösmarty et al., 2000; Bruijnzeel, 2004; Bonell and Bruijnzeel, 2005; Calder et al., 2007), with water flows being among the most prominent determinants of human health and wellbeing (Sullivan, 2002; Falkenmark and Rockström, 2004; Kummu et al., 2010; Rockström et al., 2014). However, as the rate of climate change and the uncertainty of climatic variability continue to increase (Thornton et al., 2014), the relationship between forests and water flow will also change (Caldwell et al., 2012). Would it help to plant more trees? Would this make water scarcity worse? Does it matter what type of trees? Does it matter where and how they are integrated into the landscapes? Are floods and droughts linked?

To respond to these concerns, this Global Forest Expert Panels (GFEP) assessment focuses on three key questions:

1) *"Do forests matter?":* To what degree, where and for whom, is the ongoing change in forests and trees outside forests increasing (or decreasing) human vulnerability by exacerbating (or alleviating) the negative effects of climate variability and change on water resources?
2) *"Who is responsible and what should be done?":* What can national and international governance systems and co-investment in global commitments do in response to changes in water security?
3) *"How can progress be made and measured?":* How can the UN SDG framework of Agenda 2030 be used to increase the coherence and coordination of national responses in relation to forests and water across sectors and from local to national and international scales?

The scientific evidence on these questions has not yet been systematically assessed, but partial answers exist for many parts of the world. The world's primary bodies dealing with global climate change (IPCC[1] and UNFCCC[2]) have viewed the role of forests and trees exclusively as carbon sinks and stores. In contrast, water and the role of forests and trees as modulators of the hydrological cycle have not received the explicit attention needed (Díaz et al., 2015; Maier and Feest, 2016; Pascual et al., 2017). The GFEP on Forest and Water recognised that the answers to the three questions would depend on the region of focus and require a timeframe and resources beyond those available at the time. In this GFEP assessment report (hereon the 'report') we identify globally relevant information on forest-water interactions and showcase implications for international policymakers. At the sub-national scale, there is significant variability in the values, priorities and attitudes of local people, associated with changes in the quantity and quality (type) of forests and local drivers of change. The combined effects of climate change, reduced forest functions, and increased demand for water for human health and well-being deserve more explicit attention by our governance systems at, at least, four scales: the local, the landscape, the national and the global (including transboundary) scale.

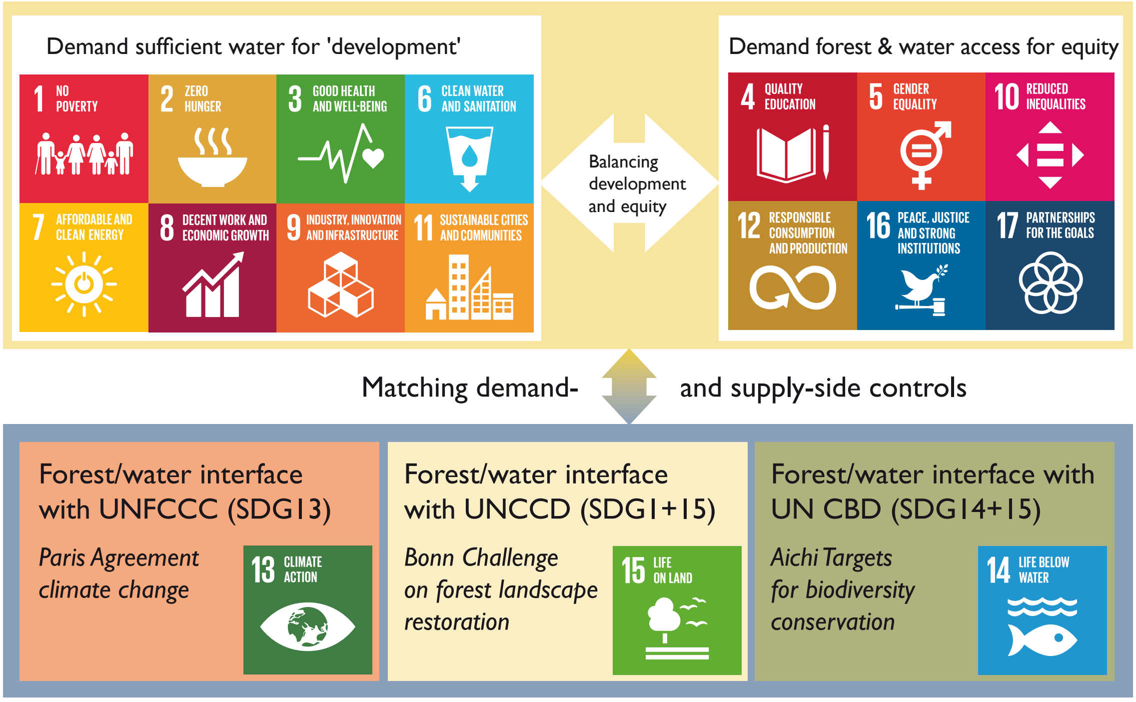
1.2 Policy Context

The primary global policy context for this assessment is shaped by the 17 SDGs defined in the Agenda 2030 by the UN in 2015[3]. The SDGs can be split into three groups (Griggs et al., 2013; Figure 1.1):

- **Eight SDGs require an increased supply of safe, secure and reliable water.** SDG 1, 2, 3, 6, 7, 8, 9 and 11 imply an increased demand for water of the right quality and temporal availability, for use in agricultural or industrial production, in support of (hydro)energy, urban systems, sanitation and health services. Goals for the water-energy-food-income nexus and the general requirement of more water for development create challenging contexts which require making trade-offs where water supply is limited, especially when urban and industrial water needs are added to this list of demands.
- **Six SDGs address social justice and equity, and their attainment will reduce unjust and inequitable access to forests and water.** SDG 4, 5, 10, 12, 16 and 17 deal with changes in human and social capital (education, gender, reduced inequality, responsible consumption and production, strong institutions and international cooperation), and their attainment will reduce inequity in access to forests and water, through education, gender equality, conflict management and changes in institutions.
- **Three SDGs build and maintain an ecological infrastructure in support of the other 14 SDGs by adapting to climate change and securing the integrity of the terrestrial and aquatic parts of the planetary system.** The three remaining SDGs deal with climate change (13), integrity of aquatic (14) and terrestrial

[1] Intergovernmental Panel on Climate Change: http://www.ipcc.ch/
[2] United Nations Framework Convention on Climate Change: https://unfccc.int/
[3] https://sustainabledevelopment.un.org/?menu=1300

Sustainable Development Goals (SDGs) in relation to forest/water relations Figure 1.1



Source: Authors' own elaboration

(15) parts of the planetary system and try to maintain an ecological infrastructure conducive to goals of the first and second group.

The challenge of water security in the face of climate change and increased demands has been recognised at high policy levels (Pittock, 2011; Hussey and Pittock, 2012; Benson et al., 2015; Pahl-Wostl, 2015; Smajgl et al., 2016) but will not be adequately addressed if each of the SDGs (and their associated targets) are seen as independent ambitions (Figure 1.1). Rather, the overall philosophy of the UN SDGs calls for a synergistic approach and integration. Water-relevant targets have been framed for all SDGs (Table 1.1).

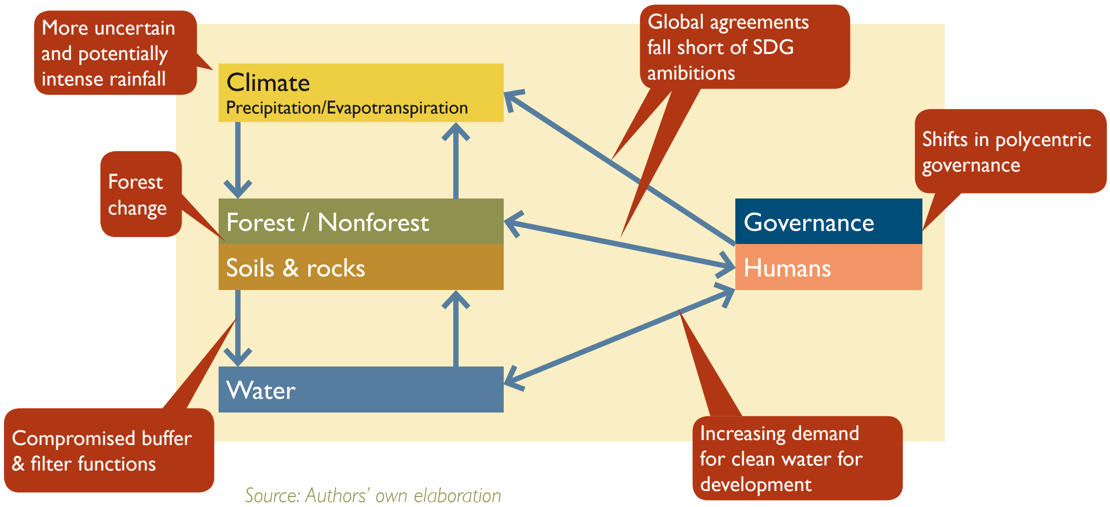
1.3 The Climate-Forest-Water-People System

The four core elements of the system of focus for this assessment are climate, forest, water and people (Figure 1.2).

1) *Climate.* Climate zones are characterised by differences in precipitation and temperature, which are primary determinants of water and energy limitations to evaporation.
2) *Forests.* Biomes vary among these climate zones (Holdridge, 1967). The anthropogenically induced diversity of ***forests and trees*** within each biome can be described as a 'forest transition' (Dewi et al., 2017) – i.e., old-growth (in some rare cases, pristine) forests, secondary forests, agroforests, plantations, agriculture with sparse tree cover and (peri)urban forests. Associated with this forest transition is a range of terms for changes in quantitative and qualitative tree cover, including deforestation, forest degradation, reforestation, afforestation and agroforestation. Defining an operational forest is a non-trivial issue in this context (Chazdon et al., 2016), and here we take an inclusive approach to all tree cover, including trees outside forest (de Foresta et al., 2015), domestic forests (Michon et al., 2007), trees on farms (Zomer et al., 2016) and trees in urban environments (Dwyer et al., 1991; Nowak et al., 2001; Hegetschweiler et al., 2017).
3) *Water.* Various parts of the global hydrological cycle have been studied as '*blue water'* (in streams, rivers, lakes or groundwater stocks and available for a range of human uses) and *'green water'* (held in the soil and vegetation and available for use by plants and/or slow release to 'blue water' forms) (Falkenmark and Rockström, 2006). A further colour of water closes the hydrological cycle: '*rainbow water'* which is atmospheric moisture, as a potential source of rainfall (van Noordwijk et al., 2014), also known as 'invisible' water (Keys et al., 2016) or 'rivers in the sky' (Arraut et al., 2012; Witze, 2015). In colder climates some precipitation is in the form of snowfall and seasonal temperature matters for its phase change to blue or green water. In cloud forests, rainbow water can be captured by vegetation as 'horizontal' precipitation; and, in response to temperature fluctuations, condensation of dew on plant surfaces can similarly make water available without measurable rainfall.

Core relations between subsystems as focus for this report and the global changes that are affecting the way forests function in relation to water supply

Figure 1.2



Source: Authors' own elaboration

Sample of the specific targets within the SDG framework that are relevant to this GFEP report (UNGA, 2015)

Table 1.1

Target 4.7	By 2030, ensure that all learners acquire the knowledge and skills needed to promote sustainable development, including, among others, through education for sustainable development and sustainable lifestyles, human rights, gender equality, promotion of a culture of peace and nonviolence, global citizenship and appreciation of cultural diversity and of culture's contribution to sustainable development.
Target 6.6	By 2020, protect and restore water-related ecosystems, including mountains, forests, wetlands, rivers, aquifers and lakes.
Target 8.4	Improve progressively, through 2030, global resource efficiency in consumption and production and endeavour to decouple economic growth from environmental degradation, in accordance with the 10-year framework of programmes on sustainable consumption and production, with developed countries taking the lead.
Target 10.2	By 2030, empower and promote the social, economic and political inclusion of all, irrespective of age, sex, disability, race, ethnicity, origin, religion or economic or other status.
Target 12.8	By 2030, ensure that people everywhere have the relevant information and awareness for sustainable development and lifestyles in harmony with nature.
Target 13.1	Strengthen resilience and adaptive capacity to climate-related hazards and natural disasters in all countries.
Target 13.2	Integrate climate change measures into national policies, strategies and planning.
Target 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.
Target 15.2	By 2020, promote the implementation of sustainable management of all types of forests, halt deforestation, restore degraded forests and substantially increase afforestation and reforestation globally.
Target 15.5	Take urgent and significant action to reduce the degradation of natural habitats, halt the loss of biodiversity and, by 2020, protect and prevent the extinction of threatened species.
Target 15.9	By 2020, integrate ecosystem and biodiversity values into national and local planning, development processes, poverty reduction strategies and accounts.
Target 16.6	Develop effective, accountable and transparent institutions at all levels.
Target 16.7	Ensure responsive, inclusive, participatory and representative decision making at all levels.
Target 17.14	Enhance policy coherence for sustainable development.
Target 17.15	Respect each country's policy space and leadership to establish and implement policies for poverty eradication and sustainable development.

Existing meteorological precipitation data, therefore, only represent part of these inputs of water to vegetation. There is an increase in uncertainty and related challenges for policy and management decisions from *'blue'* to *'green'* to *'rainbow'* water.

4) *People.* People depend on water for a multitude of functions – e.g., drinking water, sanitation, irrigation, transportation, hydropower generation and industrial cooling and processes. Dependency on surface-, ground- or piped water from non-local sources determines substantial variation in water security and vulnerability to climatic variability among social strata. Vulnerability is also associated with gendered differentiation of roles and rights in relation to access to water. Governance in this context, represents the set of formal and informal institutions and behaviours (actors, actions and rules) through which people act to alter forests and water. Many forms of governance are possible, at many scales.

In our GFEP report, the climate-forest-water-people system and all of the interactions this entails are considered. Specifically, climate is a cross-cuttting theme and 'Water for forests', 'Forests for water', 'Water/forests for people' and 'People for forests/water' are considered.

1.4 Risks to the Climate-Forest-Water-People System

Climate change is not just an issue of increasing temperature, but a symptom of more encompassing changes to the global energy balance and water cycle. Human populations and societies have risen and fallen, large areas of forest have been cut down and regrown, and the climate has varied before (Williams, 2003). The current 'Anthropocene' era, however, is the first geological period globally dominated by a single species (Crutzen, 2006; Waters et al., 2016). Climate, water availability, forest conditions, water management and societal expectations are changing very rapidly (Milly et al., 2008). We are in a 'new normal' of ongoing change (Rosegrant et al., 2012; Angeler and Allen, 2016). Today's decisions must anticipate changes that will occur during the lifetime of trees that start to grow now.

The concept of "Planetary Boundaries: Exploring the Safe Operating Space for Humanity" (Rockström et al., 2009a, b) puts a spotlight on the unsustainability of current development trajectories and ambitions with the idea of a 'safe space'. The basic premise is that this safe space is bounded in at least nine dimensions by limits to human resource appropriation and disturbance of nutrient and water cycles. Transgression of any of the nine boundaries will be "deleterious or even catastrophic due to the risk of crossing thresholds that will trigger non-linear, abrupt environmental change within continental- to planetary-scale systems" (Rockström et al., 2009a). Positive feedback and accelerated change may lead to abrupt shifts to alternate configurations, radically different from the current situation, for example in atmospheric or ocean circulation or terrestrial climates. Rockström et al. (2009a) suggested that three of these nine boundaries have already been exceeded, and that for all others the current trajectory is heading for the boundary, rather than away from it. Despite debate (e.g., Montoya et al., 2018), the concept of planetary boundaries to human resource appropriation is a key feature of contemporary discourse on environmental policy.

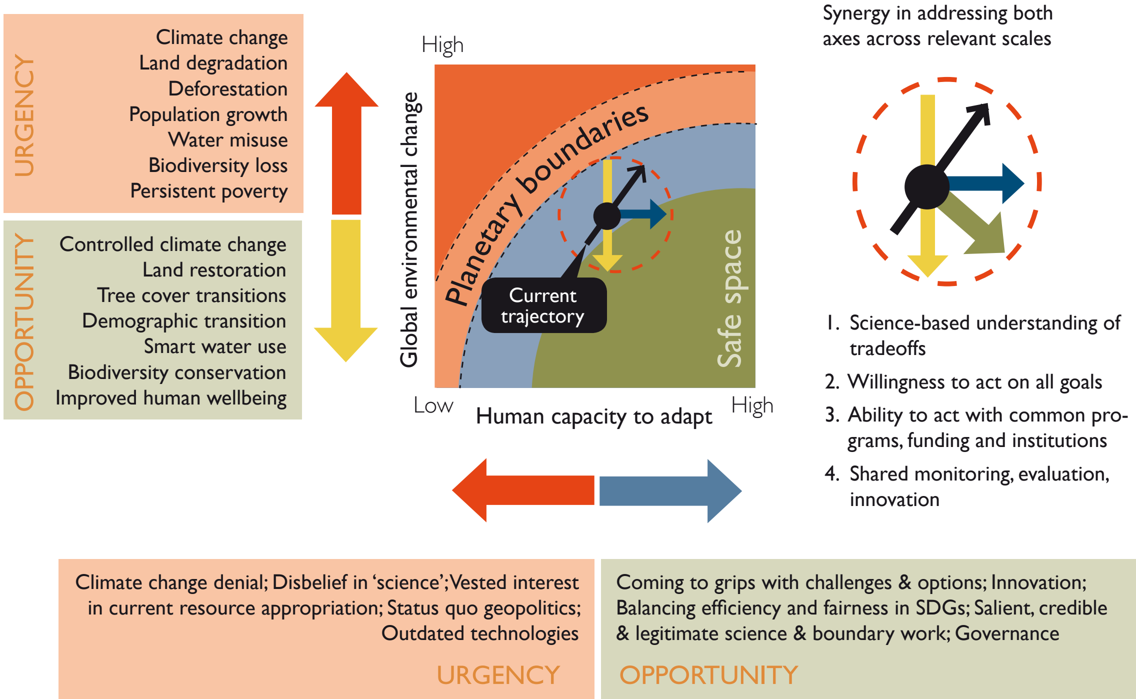
An extension of the concept of planetary boundaries is to shift the focus from just the Earth system to the role of humans in this system (Figure 1.3). Both human appropriation of global resources and human capacity to adapt define the safe space. If human capacity to adapt is low (for example by remaining in denial phase for issues such as global climate change or by systematically discrediting results obtained through scientific analysis) maintenance of the current resource appropriation trajectory makes collapse more likely.

We adapt the extended concept of planetary boundaries to deal with renewable resources such as forests and water. Seen from this perspective, two equally important shifts are (on the ecological Y axis) a rapid halt to, and reversal from, the current tendency towards increased human appropriation of global resources (including forest and water) and (on the social X-axis) an increased human capacity to adapt. Under this perspective, issues of forest and water cannot be singled out for separate action. Steps in the desired direction may need a combination of: 1) science-based understanding of tradeoffs, 2) willingness to act on all goals, to maximise the platform for positive change, 3) the ability to act with common programmes, funding and institutions, and 4) shared monitoring, evaluation and innovation, to ensure effective learning loops.

A systems approach supports the consideration of interacting scales (global to local, and back) (Rockström et al., 2014), captures interdisciplinary aspects (MEA, 2005; Díaz et al., 2015; Pascual et al., 2017; Ellison et al., 2017) and considers multiple interacting knowledge systems between policy arenas, local stakeholders and various types of science (Leimona et al., 2015; Clark et al., 2016; Creed et al., 2016; van Noordwijk, 2017). The risk management standard (ISO 31000) of the International Organization for Standardization (ISO) is a globally-accepted system that provides an opportunity to manage risk in a structured manner within the scope of a given policy objective. Within the ISO 31000 standard, the ISO 31010 Bowtie Risk Management Assessment Tool (IEC/ISO, 2009) has been used to evaluate the overall performance of a system of management measures that was put in place to reduce risk and achieve policy objectives. Governments around the world are starting to use the ISO 31000 and ISO 31010 tools to improve ecosystem management (e.g., Creed et al., 2016; Kishchuk et al., 2018) and to assess governments' ability to achieve the SDGs. We apply this framework to identify, analyse, evaluate and treat the risk of not meeting the SDGs by mismanagement of the forest-water relationship.

Figure 1.3

Two main axes that determine current trajectory for humankind interacting with planetary boundaries: global ecological change on the Y-axis, and human capacity to adapt on the X-axis. The current trajectory is towards increased human appropriation of global resources (including forest and water)



Source: Authors' own elaboration

1.5 Structure of the Report: Considering Risk in a Systematic Way

The structure of this GFEP report is inspired by the Bowtie Risk Management Assessment tool (Figure 1.4). We linked drivers of forest and land use change to pressures on ecosystem structure and changes in ecosystem functions. These pressures affect ecosystem services and their delivery to people, leading to a range of prevention controls to reduce pressures caused by drivers or mitigation controls to reduce impacts or to enable adaptation, at local, landscape, national and international scales.

Furthermore, this GFEP report zooms in from global-to-local scales to diagnose current risks, and then zooms out in considering options to adapt to global change, or deal with its consequences. Specifically, the structure of the report is as follows:

Chapter 2 reviews the science underpinning seven of the 10 system delineations (Figure 1.5) that represent 'building blocks' for the current report; it clarifies the interactions between climate, forests and water regimes at the landscape scale, focussing on the current situation (*status quo*) as a basis for the system response to ongoing change. It also introduces the social and governance dimensions of dynamic social-ecological systems;

Chapter 3 describes the determinants of change in the forest-water relationship, and global drivers of change that affect climate, forest, water and people at the landscape scale. This chapter highlights the relevance of time and space when considering the role of drivers on the social-ecological system;

Chapter 4 synthesises understanding of the hydrological effects of the changes described in Chapter 3; hydrological regimes in forests and land with partial tree cover are shaped by interactions and feedbacks between climate and vegetation with implications for local and global hydrology;

Chapter 5 presents future scenarios of forest-water ecosystem services that relate the rate of global change to the capacity of people and their governance systems to adapt;

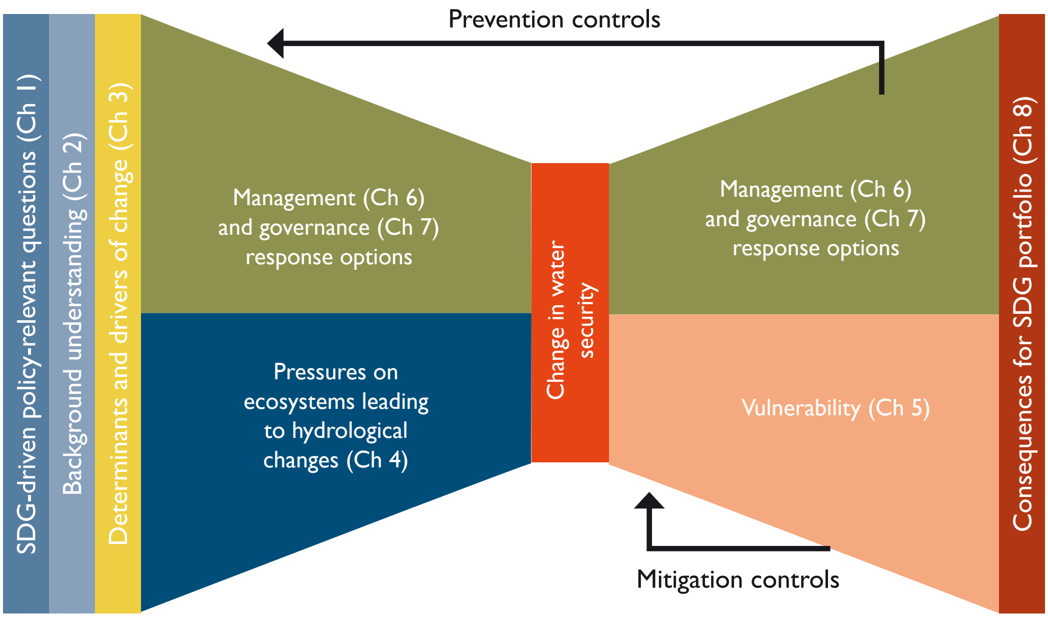
Chapter 6 presents management options to address stresses on the forest-water-climate system at the catchment scale;

Chapter 7 considers options for policy and governance responses at the landscape, national and international scales; and

Chapter 8 provides the main conclusions, summarises outstanding research gaps and highlights points of relevance for policy dialogue;

The ISO 31010 Bowtie Risk Management Tool inspired the structure of this GFEP report

Figure 1.4



Source: Authors' own elaboration

Box 1.1

Shoulders on which we stand...

A selection of titles indicates the range of textbooks, reviews and expert syntheses that form a backdrop to current thinking. Their authors are the shoulders on which we stand, allowing us (as Newton) to see further.

These include (in chronological order):

Principles of forest hydrology (Hewlett, 1982)

Forests, climate, and hydrology: regional impacts (Reynolds and Thompson, 1988)

Climate, water and agriculture in the tropics (Jackson, 1989)

Elements of physical hydrology (Hornberger et al., 1998)

The blue revolution: land use and integrated water resources management (Calder, 1999)

Forests and water, international expert meeting on forests and water, 20-22 Nov., 2002 Shiga, Japan (International Forestry Cooperation Office of Japan, 2002)

The cost of free water: The global problem of water misallocation and the case of South Africa (Bate and Tren, 2002)

World water and food to 2025: dealing with scarcity (Rosegrant et al., 2002)

Deforesting the earth: from prehistory to global crisis (Williams, 2003)

Forests, water and people in the humid tropics: past, present and future hydrological research for integrated land and water management (Bonell and Bruijnzeel, 2005)

Forests and Floods: Drowning in Fiction or Thriving on Facts? (FAO-CIFOR, 2005)

Forest hydrology: an introduction to water and forests (Chang, 2006)

Towards a new understanding of forests and water (Calder et al., 2007)

Hydrologic effects of a changing forest landscape (National Research Council, 2008)

Floods, famines, and emperors: El Niño and the fate of civilizations (Fagan, 2009)

Sustainability science for watershed landscapes (Roumasset et al., 2010)

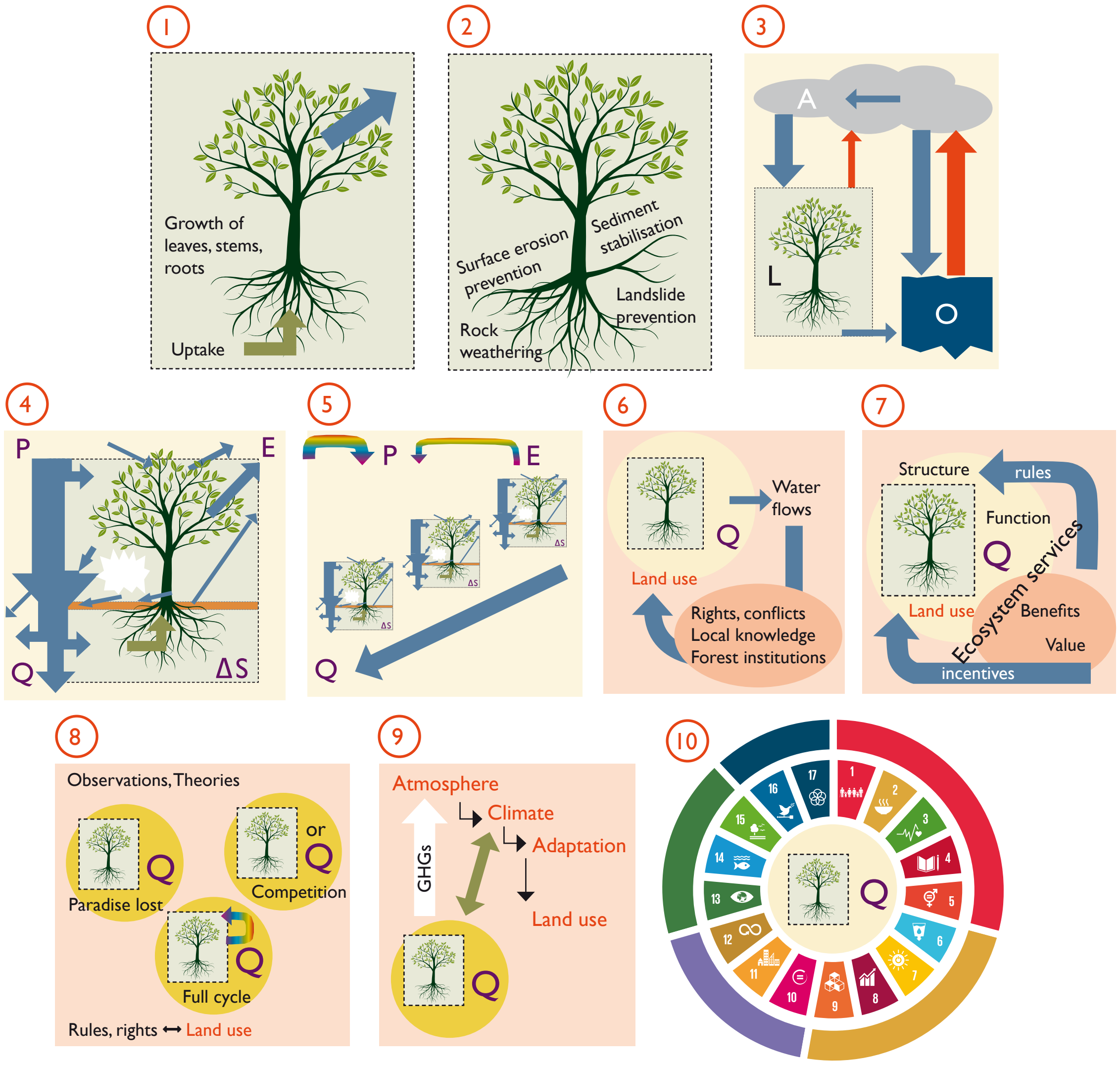
Hydrology and the Management of Watersheds (Brooks et al., 2013)

1.6 Strong Foundations and Emerging Perspectives

This report is by no means the first time the relationships between forest and water are reviewed (Box 1.1). While acknowledging solid foundations and excellent previous reviews, we found that our questions on the way forest-water relations interact with the SDG portfolio as a whole have hardly been asked, let alone answered. Yet, our literature review showed significant progress in the past decade for many 'subsystems' (Figure 1.5), that have a much narrower delineation.

Probably the largest progress in the past decade is the acknowledgement of the feedback loops between the four elements of the system and the full hydrological cycle. The hydrological system has been described as a cycle for hundreds of years. Yet, most of hydrology as a science has been based on a flow perspective, where incoming precipitation is the starting point and its subsequent use is the primary concern for practitioners as well as science.

Ten subsystems of the climate-forest-water-people system studied in the scientific literature Figure 1.5



1. Trees and water (structure and function of leaves, stems and roots), *part of*
2. Forests, soils and climate (sponge effects), *part of*
3. Atmosphere, oceans and terrestrial vegetation (global water fluxes), *part of*
4. Precipitation, evapotranspiration and discharge (water balance and buffering), *part of*
5. Dynamic landscape mosaics (streamflow), *part of*
6. Land and water use rights, local knowledge and forest institutions (landscapes), *part of*
7. Social-hydrological systems (ecosystem services as valued human benefits).

In addition, there are systems that build on the first seven nested systems that explore governance of society, including:

8. Contested and evolving forest-water paradigms in public discourse, legislation and underpinning existing policies;
9. Climate change policy in its relation to forest and water interaction; *and*
10. SDG coherence in an interlinked, multiscale and polycentric governance perspective.

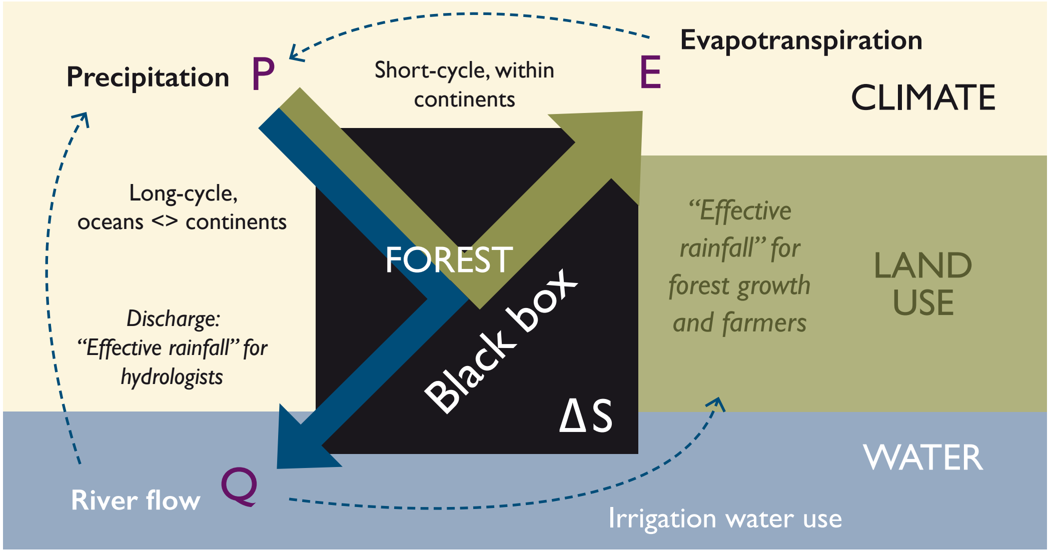
Source: Authors' own elaboration

It has taken some time before managers of a national economy realised that they were not just dealing with stocks and flows, but with a cycle, where cause-effect relations represent feedback loops. Similarly, understanding of the full hydrological perspective has been slow to emerge across scales and spheres of influence (Figure 1.6). Cyclical relations in a climate interacting with oceans and vegetated land masses are only partially addressed in current greenhouse gas and carbon dominated climate discourse (see Chapter 2).

This important change in our understanding has implications for forest-related policies which should consider not only carbon-related forest ecosystem services but also water-related ones. Major policy instruments such as REDD+ (reducing emissions from deforestation and forest degradation plus sustainable forest

Hydrological cycle and the way precipitation (P) is partitioned over evapotranspiration (E) and river flow (Q) at time scales in which the change in soil water storage (ΔS) is considered to be negligible

Figure 1.6



Source: Authors' own elaboration

management and restoration) have failed to deliver on the expectations raised (Minang and van Noordwijk, 2013; Matthews et al., 2014; Matthews and van Noordwijk, 2014), especially from a local perspective (Bayrak and Marafa, 2016; Sanders et al., 2017), where issues of water are more relevant than the rather abstract concept of carbon accounting. While some authors remain optimistic on REDD+ (Brockhaus et al., 2017), there certainly are important lessons on institutional development (Minang et al., 2014) that can be used in a new round of policies that look at the climate-forest-water-people interactions in a more holistic way, as this GFEP assessment report shows. Water may be the key to unlocking policies that flow readily from local to global scales.

1.7 No Simple Rules to Guide Policy: Perspectives Addressed in this Assessment

The questions that this report sets out to address are only partially addressed by current forest hydrology as a relatively well-defined discipline (Hewlett, 1982; Chang, 2006; Brooks et al., 2013). To operate effectively at the science-policy interface, an assessment such as this must relate to multiple knowledge systems (Jeanes et al., 2006; Rahayu et al., 2013; Leimona et al., 2015; van Noordwijk, 2017) compared to those that have historically shaped laws and institutions plus those influencing today's decisions. Simplifying a richer and more complex reality, we identify three perspectives concerning the forest-water relationship: 'no forest-no water', 'more forest-less water', and 'it depends' (Figure 1.7), with a swinging back and forth among these three perspectives.

Perspective 1: No Forest – No Water/More Forest – More Water

The first perspective is that all aspects of forests are positive for any issue related to water, and that any problem of flooding, droughts, landslides or pollution is the direct consequence of deforestation or forest degradation, with restoration and reforestation as logical, universal solutions; in slogan format: *No Forest, No Water*.

Perspective 2: More Forest – Less Water

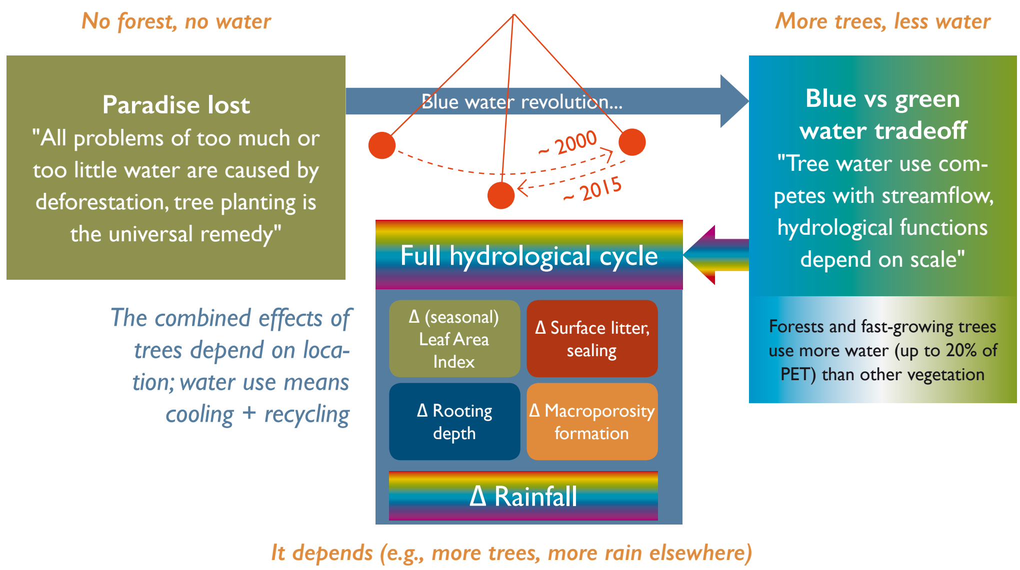
The second perspective is that trees use more water than other vegetation, that the evidence for linking deforestation to floods in anything but a small catchment is weak, and that there is a near-universal loss of 'blue water' when there are more trees using 'green water'. Climate change is the primary culprit of floods. Large-scale reforestation does not increase (but rather decreases) total water yield, but also (in many cases, at least) dry-season streamflow; in slogan format: *More Trees, Less Water.*

Perspective 3: It Depends

A 'full hydrological cycle' perspective of forests and water demands a more nuanced, spatially explicit position that, depending on the context, changes in tree cover can be related to a range of quantifiable functions and their trade-offs. This 'it depends' rule suggests that a "right tree at the right place for a clear function" concept should replace blanket reforestation targets. Furthermore, it combines the two apparently conflicting perspectives above, and focuses on identifying particular types of benefits for particular groups.

Cartoon of a pendulum swinging between three public perspectives of the key forest-water relations

Figure 1.7



Source: Authors' own elaboration

It is from the 'it depends' perspective that the GFEP assessment report builds a scientific foundation (in Chapters 3, 4 and 5) for policy and management (Chapters 6 and 7).

1.8 Scope and Objectives of the Global Forest Expert Panel on Forests and Water

The Collaborative Partnership on Forests (CPF)[4] established a GFEP on Forests and Water through its Global Forest Expert Panels initiative. Like previous Global Forest Expert Panels, the aim of this Panel is to provide policy-relevant scientific information to intergovernmental processes and institutions related to forests and trees, thereby supporting more informed decision making by policymakers, investors, donors and other stakeholders, and contributing to the achievement of international forest-related commitments and internationally-agreed development goals.

The GFEP on Forests and Water[5] was tasked to "carry out a comprehensive global assessment of available scientific information about the interactions between forests and water, and to prepare a report to inform relevant international policy processes and the discussions on the 2030 Agenda for Sustainable Development and related Sustainable Development Goals". The scientists on the panel defined a more detailed outline and reviewed recent literature on the specific questions that emerged. The report has been peer reviewed anonymously.

The ***scope*** of the review has focused on issues of flow regime as influenced by changes in forests and tree cover, specifically water quantity and flow regularity, with a focus on surface water and atmospheric moisture flows. Other parts of the forest-water nexus are discussed but without the depth that we had hoped for, as many of the issues were site or location specific and generalisations were weak (e.g., as for water quality issues). Groundwater dynamics, the relation between tree cover and dryland salinity, and consequences for land subsidence of groundwater extractions (as they plague a metropole like Jakarta, for example) were deemed beyond the scope of this report.

The ***objectives*** for this review are to provide an independent expert evaluation of the science-based evidence and/or major gaps of:

1) The functions that forests provide in influencing the relationship between climate and the timely availability of good-quality water to match human needs;
2) The risks that these functions are compromised by changes to forest conditions; and
3) The need for further policies and management strategies to reduce risks and deal with its consequences.

[4] More info about CPF and its members: http://www.cpfweb.org/73947/en/
[5] More info on the GFEP on Forests and Water: https://www.iufro.org/science/gfep/forests-and-water-panel/

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Chapter 2
Climate-Forest-Water-People Relations: Seven System Delineations

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CONTENTS

2.1 Introduction

In this chapter, we review current scientific understanding and hypotheses at seven system delineations that build up from the level of a 'tree' interacting with water, to that of a social-ecological system at the scale of landscapes. A system delineation separates internal entities that interact dynamically from external entities that may have a one-way influence but are not significantly influenced by feedback from within the system boundaries. Each system level has its characteristic outcomes or results. The seven (nested) system delineations (Figure 2.1) are:

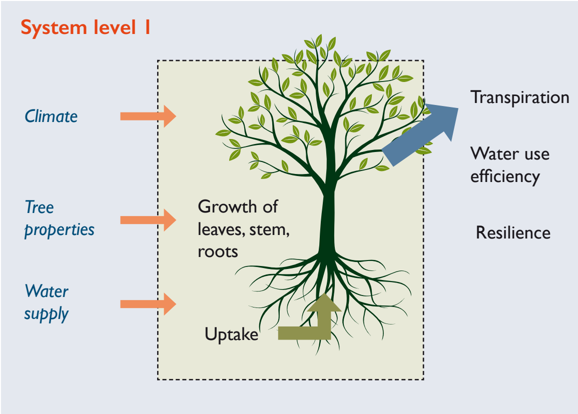
1. Trees and water: Structure and function of leaves, stem and roots, *which are part of:*
2. Forests, soil and climate: Sponge effects; *part of:*
3. Atmosphere, oceans and terrestrial vegetation: Global water fluxes; *part of:*
4. Precipitation, evapotranspiration and discharge: Water balance and buffering; *part of:*
5. Dynamic landscape mosaics: Streamflow; *part of:*
6. Land and water use rights, local knowledge and forest institutions: Landscapes; *part of:*
7. Social-hydrological systems: Ecosystem services as valued human benefits.

Elsewhere in this report, three additional system concepts are used that build on system delineation 7 (and include it as a subsystem) and explore governance of a society dealing with issues of coherence between the sustainable development goals:

8. Contested and evolving forest-water paradigms in public discourse, legislation and underpinning existing policies (as covered in Chapter 1);
9. Climate change policy in its relation to forest and water interactions (as covered in Chapter 7); and
10. SDG coherence in an interlinked, multiscale and polycentric governance perspective (as covered in Chapter 7).

2.2 Forests, Soils and Water

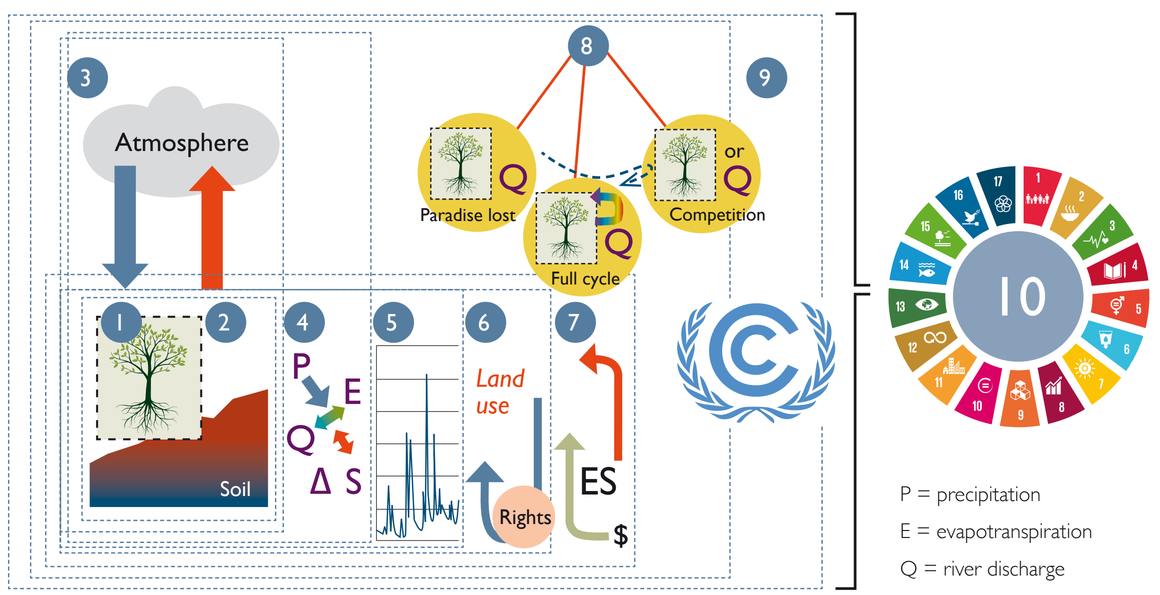
2.2.1 Trees and Water: Structure and Function of Leaves, Stems and Roots



Whole-plant physiology

Ecophysiology at the whole-plant level as a field of scientific study has a long history and rich toolbox of methods (Reynolds and Thornley, 1982; Kramer and Boyer, 1995; Lambers et al., 2008). Interactions between trees

Nesting of ten systems describing the relation between forest, water and Sustainable Development Goals (SDGs) Figure 2.1



Source: Authors' own elaboration

and water are shaped by their leaves, stems and roots. The leaves and their stomata lose water in the process of transpiration, cooling leaves while allowing carbon dioxide (CO_2) to be captured in photosynthesis.

Green leaves are essential for photosynthesis, but without stems, the leaves would stay close to the ground and be shaded by others. Trees, found in over 100 of the 620 plant families (often alongside other life forms), invest in perennial stems as a generic solution for these challenges. The stems transport water in their xylem (plant tissue), where the need to avoid getting clogged by air bubbles ('embolism') in wide vessels under dry conditions (Domec et al., 2006) is balanced against enhanced transport capacity in such vessels under wet conditions. Wood density is negatively related to vessel size, with high growth rates generally associated with low wood density, early successional status, low drought and fire tolerance and short life-spans (Larjavaara and Muller-Landau, 2010).

The roots are the primary organs for water uptake, and their amount and distribution in the soil determine options for water and nutrient uptake and structural stability (with increased demands in trees). Yet, every unit of dry matter can be used for supporting only one of the three essential organs (i.e. leaves, stems and roots) and the allocation can be considered a strategic as well as an adaptive choice (van Noordwijk et al., 1998b, 2015a). The ability of trees to persist in dry or seasonally dry climates thus depends on a variety of eco-physiological adaptations to water scarcity (Breshears et al., 2009). Root patterns of trees present in natural vegetation differ in predictable ways based on climate and groundwater table depth (Fan et al., 2017).

Diversity of contexts and ecoregions

Different tree species have different water needs depending on their phenology (timing of green leaf presence, flowering and fruit production) and crown architecture, and have different access to soil water based on their root development, making them adjusted to one or more of the ranges of climates (Box 2.1). Competition for the same water resources is minimised through mixtures of species with canopies that do not overlap, that develop their leaves at different times of the year, or that have different

Box 2.1

Diversity of contexts for forests and trees

As is evident from the Holdridge (1967) climate and vegetation classification, a wide range of forest types occur in many hydro-climatic conditions. Based on the ratio of annual precipitation to potential evapotranspiration we can expect shrub (<0.5), dry forest (0.5-1), moist forest (1-2), wet forest (2-4) or rain forest (>4). This ratio reflects rainfall, ranging from superhumid (> 8000 mm/year) to superarid (< 125 mm/year), and latitudinal zones (tropical, subtropical, warm temperate, cool temperate, boreal, subpolar and polar) interacting with altitudinal belts (lowland to montane and alpine) in determining temperature and potential evapotranspiration. The latitudinal zones also determine the pattern of seasonality (Dewi et al., 2017). A global hydro-climate map shows a wide range of P/E_{pot} ratios (P = precipitation; E_{pot} = potential evapotranspiration, both at annual time scales) (Figure 2.2A). For the 33.6% of the global land area with a P/E_{pot} ratio < 0.5 there is only sufficient water for episodic rivers; for the 35.3% with a P/E_{pot} ratio between 0.5 and 1, water supply is limited part of the year, and rivers often are strongly seasonal. For the 31.2% with a P/E_{pot} ratio > 1 there usually is sufficient water to support permanent rivers.

The main climatically determined forest categories are:

Hot and wet. Consistently warm, never freezing. Includes tropical rainforest, tropical peat swamps (Gumbricht et al., 2017), tropical montane cloud forests (Bruijnzeel and Veneklaas, 1998), and lowland moist forests. Source of the world's largest rivers.

Hot and dry. Consistently hot leading to water stress and temporary river flow only after occasional storms that exceed the infiltration capacity of the soil.

Subtropical. Warm, with wet (monsoon) and dry season and associated seasonal rivers. Growth limited by seasonal moisture availability (D'Odorico et al., 2010; D'Odorico and Porporato, 2006; Newman et al., 2006). In the subtropical dry forest/Guinea savannah/ Sudan savannah/Sahel gradient wet and warm seasons may coincide, while in the Mediterranean zone, winter rains determine a relatively cool growing season (Llorens et al., 2011). In either, forest vegetation may depend on deep water storage (Bastin et al., 2017).

Moist temperate. Hot and cold seasons are clearly differentiated and precipitation can be seasonal or evenly distributed throughout the year, supporting deciduous, evergreen, or mixed forests with modest year-round river flow.

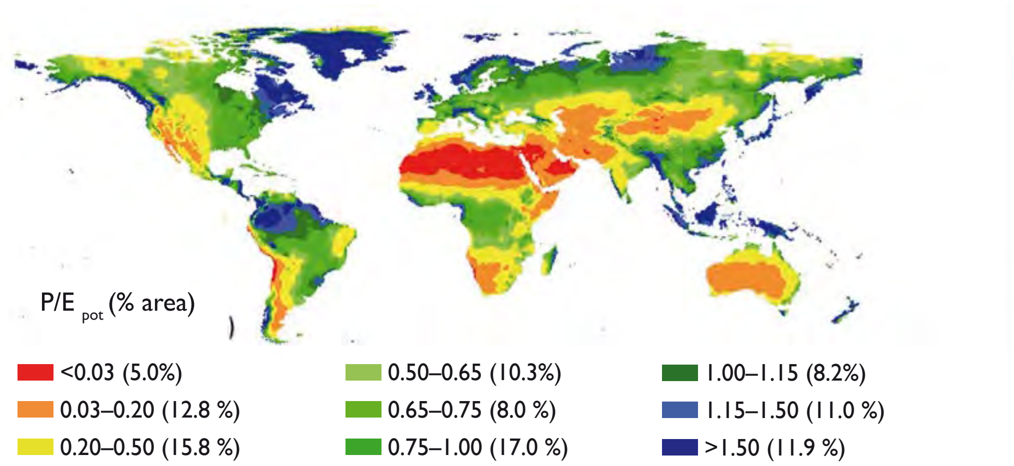
Cold and wet. Precipitation is much higher than potential evaporation, generating abundant river flow. Includes mountain climates with snow and ice.

Cold and dry. Snow and ice-dominated. Permafrost may limit rooting depth, accelerating runoff. The shallow thaw zone may be subject to drought and prone to wildfire (de Groot et al., 2013).

More differentiated schemes exist. In defining ecoregions, Olson et al. (2001) identified 867 unique terrestrial areas that are relatively large units of land containing a distinct assemblage of natural communities and species with boundaries that approximate the original extent of natural communities prior to major land-use change. These ecoregions are contained within 15 biomes, as reflected in Figure 2.2B. The trees in these various vegetation types differ both above- and belowground in key properties affecting their hydrological functions.

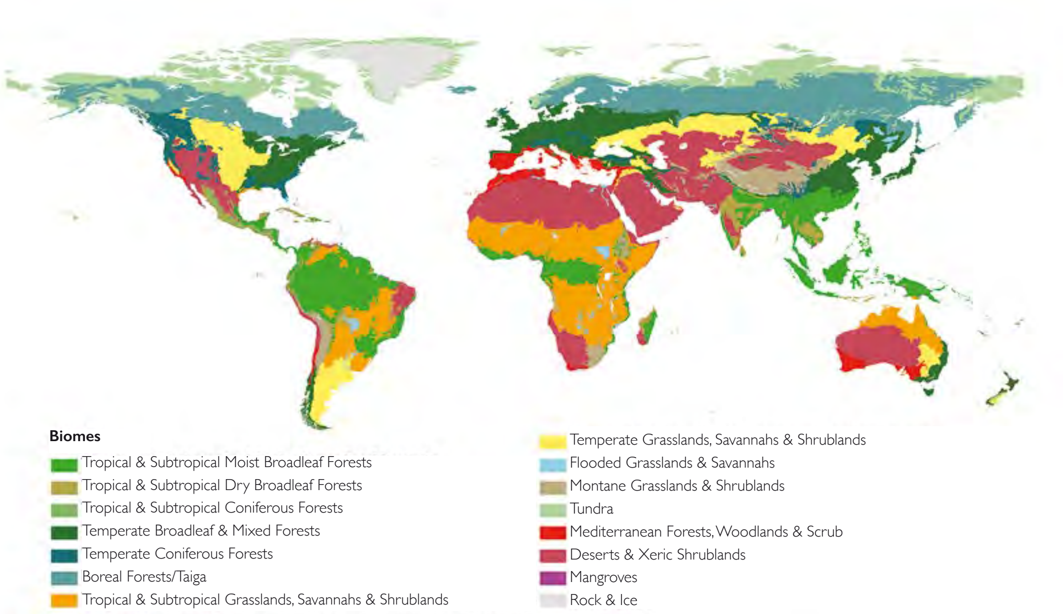
Global maps of A. Hydroclimate (precipitation (P) relative to annual potential evapotranspiration (E_{pot}) and B. Fifteen terrestrial biomes derived from 867 ecoregions Figure 2.2

A)



Source: Zomer et al., 2007; 2008, based on CGIAR-CSI Global-Aridity and Global-PET Database (http://www.cgiar-csi.org)

B)



Source: Adapted from Olson et al., 2001

rooting depths (González de Andrés et al., 2017). As a result, compared to single-species stands of trees, natural mixed forests can use water more efficiently (González de Andrés et al., 2018) and, on aggregate, respond less strongly to climate variability (Creed et al., 2014; Blanco, 2017; Laskurain et al., 2018; Kotlarz et al., 2018). Such ecological diversity effects can be mimicked in mixed tree-crop (agroforestry) systems in dry zones that can use more water compared to trees or crops alone (Bayala and Wallace, 2015; van Noordwijk et al., 2015a).

Tree rooting depth may also affect phenology, even in wet tropical climates with low seasonal variation, as it allows trees to benefit from dry (sunny) periods (Broedel et al., 2017). Tree phenology effects on hydrologic processes may be more pronounced under single-species, even-aged forests, where phenology is synchronised, compared to mixed-species, multi-aged forests, with diverse phenology (Wright et al., 2017). Forest stands dominated by evergreen species tend to impact dry season low flows to a greater extent (in terms of proportional reductions in streamflow) than annual streamflow totals (Scott and Smith, 1997). Actively growing conifer forest plantations are associated with up to 50% reductions in summer streamflow relative to old-growth conifer forests (Perry and Jones, 2017). Naranjo et al. (2011) documented for forested watersheds of western North America that trends in the observed water balance can be associated to land cover disturbances well before the start of hydro-climatic observations, a century ago.

Leaf Area Index (LAI)

The LAI of a forest or collection of trees is the total (one-sided) leaf area per unit two-dimensional ground surface area. If leaves were evenly spread out, an LAI of 1 would represent full coverage of soil and complete light interception. Given the architecture of plants, an LAI of 2-3 is typically needed for capturing 95% of radiation. An LAI of 5-6 is common in closed-canopy forests and allows only a small fraction of incoming radiation to reach the forest floor. Similarly, the leaf area also intercepts a large share of precipitation before it reaches the ground, but most of this drips off the leaves and continues its downward journey. The LAI of a forest or trees is influenced by a range of factors that impact the physical attributes of the canopy (i.e., canopy leaf density):

- inherent characteristics of forest/trees (e.g., species composition, age class distribution, tree size, tree density, canopy architecture, and canopy phenology);
- availability of, and competition for, light, water, and nutrients, which influences the spatial arrangement of the forest/trees (e.g., riparian, upland, forest margin effects); and
- anthropogenic effects and management practices (e.g., genetic modification, landscape alteration, weed control, harvesting, fertilisation, pruning, thinning and irrigation); and any disturbance that alter the area of leaves on trees (e.g., drought, wind, pests and diseases, pollution and temperature extremes).

Because transpiration takes place through stomata present on plant leaves, leaf area and water use are correlated (Gebhardt et al., 2014). In general, the higher the LAI, the greater the transpiration potential of the vegetation. However, increases in LAI are not directly correlated with rates of transpiration, but are moderated by water and energy availability, vapour pressure deficit (moisture demand of the air), and resultant variation in stomatal conductance (water and CO_2 fluxes) within the canopy. The LAI of a forest, such as a monoculture plantation, or mixed species/mixed age forest, consequently has significant effects on forest hydrology. As LAI, and the resultant transpiration potential, increase (usually with increasing tree age) so does the potential for extraction of water from the soil profile, through the trees' stems and leaves, into the atmosphere. Resultant changes in soil water at different depths in the soil profile subsequently affect infiltration, groundwater recharge and ultimately, streamflow. LAI also affects other hydrologic processes such as throughfall, stemflow, evaporation of leaf-intercepted rainfall and air turbulence (Hall, 2003).

Seasonal variation influences forest hydrology through the timing of leafing and associated interception and transpiration. Forest phenology includes leaf flush, senescence, flowering and fruiting, and can be understood as balancing the photosynthetic opportunities of a low cloud cover, high-radiation season, with water availability in the wet season. It is modulated by the presence of pollinators, seed dispersers and predators, pests and diseases. In deciduous forests, phenology strongly influences seasonal patterns of evapotranspiration, groundwater recharge and streamflow, but phenology of evergreen trees also produces noticeable seasonal variations in streamflow and streamflow response to forest change (Jones and Post, 2004).

Rooting depth

Forests and trees obtain most of their water through their roots, extracting it from soil pores. Root length density (length per unit volume of soil) determines the degree to which roots have access to all soil water; it varies with species, age, stand density, and soil characteristics. Woody vegetation usually has deeper roots than grasses, allowing it to take up water from deeper groundwater as well as soil moisture in the unsaturated zone (Moore and Heilman, 2011). While short-lived annual species in desert biomes have shallow roots, perennial species (including trees) in seasonally dry regions generally have deeper root systems than in those in permanently wet regions. Also, root systems in coarse-textured soils with rapid infiltration and limited water storage are generally deeper than those in fine-textured soils (Collins and Bras, 2007). These differences can be found as adaptive responses within any plant species ('functional equilibrium' theory; van Noordwijk et al., 2015a), but also between species most commonly found in these various environments. Rooting depth is not static and may change dynamically through the year. Water below the deepest roots can still be accessible to plants through capillary transport.

Shallow root system in moist but nutrient poor tropical peatland forests in Kalimantan, Indonesia
Photo © Daniel Murdiyarso

Plant roots move water from wetter to drier layers. Such equilibration usually consists of hydraulic lift, the process of bringing water to the soil surface from deeper rooted layers, or downward siphoning, the process of bringing fresh precipitation to deeper layers (Bayala et al., 2008; D'Odorico et al., 2010). Hydraulic equilibration by forests and trees is most effective at night when stomata are closed and transpiration has stopped (Bayala et al., 2008; Prieto et al., 2012). In this way, water can rehydrate drier zones connected by a single root system (Manoli et al., 2014), but can also leak out of roots in dry soils and be captured by roots of other species, as has been demonstrated with isotopic tracer experiments (Caldwell et al., 1998). Estimates of the extent of hydraulic redistribution of water by trees vary by nearly two orders of magnitude and depend on the combination of root architecture, soil physical properties, and gradients in water potential in the rooted part of the soil profile (Neumann et al., 2012). In temperate and semi-arid environments, hydraulic redistribution can contribute 17-81% of water transpired (Sardans et al., 2014) and may account for up to 30% of transpired water on dry late summer days in seasonally dry and wet forests. It may also enhance seedling survival and maintain overstory transpiration during summer droughts (Brooks et al., 2002; Domec et al., 2010). Hydraulic redistribution has been documented in Amazonian rainforests (Oliveira et al., 2005), neotropical savannahs (Scholz et al., 2002), semi-arid shrublands (Ryel et al., 2002), desert shrubs (Hultine et al., 2004), seasonally dry conifer forests (Domec et al., 2004), semi-arid savannahs (Barron-Gafford et al., 2017) and Sahelian agroecosystems (Bayala et al., 2008; Kizito et al., 2012).

Variation in root length density and rooting depth between tree species has direct relevance for soil moisture dynamics (Wilcox et al., 2011). As different species have different capabilities to explore soil layers, water out of reach for some species could still be available for others (Hardanto et al., 2017). Through different root system architecture, different tree species sharing the same stand can avoid competition and complement each other, using water from different soil layers or at different times of the year (Xu et al., 2011; Forrester and Bauhus, 2016; González de Andrés et al., 2018).

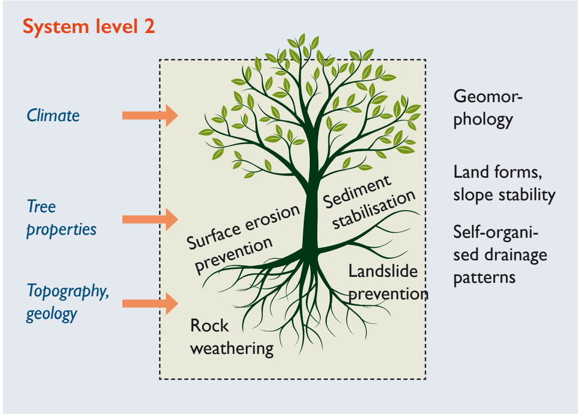
Water excess may be a problem for trees in some settings. Trees in mangroves and peat swamps have adapted roots to enable them to maintain adequate supplies of oxygen and remove gases such as ethylene and methane. Stilt roots, pneumatophores and aerenchym are common adaptations in mangrove (Pi et al., 2009) and peat swamp forests (Farmer et al., 2011; Pangala et al., 2013).

2.2.2 Forests, Soils and Climate: Sponge Effects

Forests and soils: a two-way relationship

Forests depend on soil, but also play a major role in soil formation by bedrock weathering, maintaining soil on-site (reducing landslides and erosion), and capturing it in sedimentation sites. Globally there is major variation in the depth and nature of soils, even when the forests look similar, causing variation in hydrologic responses.

The depth of a soil profile, together with its texture and soil organic matter content, determine the 'sponge'



effect (and its spatial variation) of buffering water availability within the reach of root systems. Forests influence soil formation and soil retention in the landscape (Brantley et al., 2017). Soils, in turn, impact forest hydrology through retention of water, infiltration, percolation, soil moisture storage, release, erosion, sediment deposition, landslides, and as a medium for roots. Deforestation effects on streamflow depend on soil type, soil depth and terrain features that are often ignored when seeking generic patterns. Spatial information on subsurface hydrology, added to remote sensing that has so far focussed on land cover, is currently filling a major research gap (McDonnell et al., 2018).

Soil water storage capacity

Part of the literature and much of the modelling done to date rely on a 'rooting depth' concept that assumes all water above a certain depth is available and all water below is unavailable to vegetation. This simplified approach assumes that water below rooting depth will either exit the ecosystem as subsurface flow or recharge the groundwater stocks. Using this approach, it is possible to estimate the water storage capacity of the root zone (Wang-Erlandsson et al., 2016). In a recent drought in California, specific forests where trees were found to have access to deep weathered bedrock were found to remain green (Rempe and Dietrich, 2018). Using data from 300 diverse catchments in Thailand and the USA, Gao et al. (2014) estimated the effective soil moisture storage to vary from around 50 to 500 mm, representing 25 to 250 days in which evapotranspiration rates of 2 mm/day can be sustained in the absence of precipitation or lateral inflows. Root-zone water storage capacity was reduced by logging in catchments with long-term monitoring data, and took a decade or more after forest regrowth to recover (Nijzink et al., 2016).

Macroporosity and water infiltration

The soil's infiltration capacity below the surface is influenced by soil porosity. Porosity defines the spaces between soil particles and aggregates, and thus the two primary biotic influences, are soil aggregation (related to organic matter and fungal hyphae) and the balance between disappearance and generation of macropores by roots and soil macrofauna ('engineers') (Bünemann et al., 2018). In some regions, with porous soils and relatively low precipitation or snowmelt rates, almost all water infiltrates. In these regions, overland flow is generally not a consideration, except where water has accumulated in the soil (e.g., at the base of hillslopes with shallow soils). In these places, soil saturation means that there is no room for more water to infiltrate, so all incoming water remains at the surface, creating saturation excess overland flow (potentially leading to flooding). In other regions, a combination of low infiltration capacity and/or high rates of precipitation can lead to infiltration excess overland flow which will contribute to flooding, with the risk of erosion. Overland flow at the soil surface – whether created by infiltration excess or saturation excess – does not contribute to subsurface water storage, which can sustain both streamflows and plant growth during drier periods. It can, elsewhere, lead to excess soil moisture, waterlogging and vegetation dieback.

Soil moisture storage depends on the pore size distribution of the soil. Very large pores (macropores) associated with roots, animal burrows, arthropods and earthworms are specifically sensitive to soil compaction but where present enable rapid infiltration and limit overland flow (Beven and Germann, 2013; Vereecken et al., 2016; Barrios et al., 2018); intermediate size pores (mesopores) associated with sand- to silt-size particles contribute to soil water holding capacity against gravitational drainage, and tiny pores (micropores) within organo-mineral aggregates or clay particles hold water very tightly. Dominance of vertical (infiltration) or horizontal (interflow) processes can depend on pore distribution, but also on precipitation and season (Grayson et al., 1997). Tightly bound micropore water can be differentiated from mobile water that tends to enter the stream via 'interflow' and is taken up most readily after a rainfall event (Brooks et al., 2010; Berry et al., 2017; Evaristo and McDonnell, 2017).

Coarse-textured (sandy) soils have low water storage capacity, but often high infiltration capacity, except where they develop water repellency and induce overland flows (Doerr et al., 2002). Fine-textured (clay and silt) soils have high storage capacity, but low infiltration capacity, except where cracks and biogenic macropores develop. A soil with a wide range of pore sizes has both high infiltration and high water storage capacity. In many landscapes the most agriculturally suitable soils have been converted and forests are left on the less favourable sites.

Loss of forest cover and forest disturbance generally reduce the capacity of soils to absorb and retain moisture. In the short term, forest harvest or forest removal can lead to macropore enlargement as roots decompose, facilitating infiltration (van Noordwijk et al., 1991; Noguchi et al., 1999), but subsequent collapse of macropores without new ones being generated reduces infiltration rates and increases overland flow. High runoff from bare patches combined with high interception and infiltration by shrubs or trees effectively partitions scarce soil moisture among plants in patchy dryland vegetation (Crockford and Richardson, 2000; Llorens et al., 2011; Li, 2011; Maestre et al., 2016), creating 'resource islands' (Roberts and Jones, 2000). Positive tree influences on soil macroporosity and infiltration can last years or decades after the tree has died (Ilstedt et al., 2007; 2016).

Litter layer and overland flow

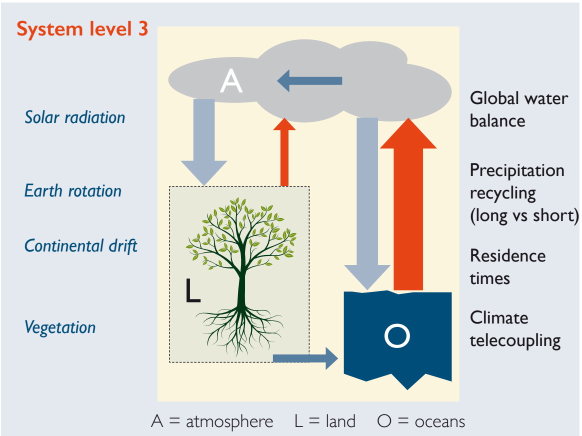
Once precipitation water passes through the tree canopy it encounters a critical interface at the litter/soil layers. Here, partitioning occurs between that water which infiltrates further downward into the soil, and that which does not. The ratio between the rate at which water reaches the soil surface (throughfall or snowmelt) and the rate at which the soil allows water to infiltrate determines this partitioning. Litter is composed of decaying leaves and needles, but also fungi (including mycorrhizal hyphae), soil arthropods, and earthworms, whose activities produce organo-mineral aggregates. Roots, animal burrows, arthropods, and earthworms create macropores, which in turn promote rapid water infiltration and limit overland flow (Barrios et al., 2018). In the short term, the presence of a litter layer is a store of water, and it also protects soil surfaces from the erosive capacity of direct rain droplet impacts (Hairiah et al., 2006). Where litter layers are dependent on trees, reduction of soil evaporation will partially offset increased transpiration (Wallace et al., 1999). In the longer term, the contribution of litter to soil organic matter will influence both infiltration capacity and porosity. Litter removal and grazing reduce infiltration rates and increase overland flow (Ghimire et al., 2014a). Forest and tree presence is a pre-requisite for the existence of litter, but the tree characteristics (e.g., species composition, age class distribution, tree density and deciduousness) as well as management activities (e.g., timber harvesting and under-canopy burning) also influence the properties of the litter layer in relation to how much accumulates, hydrophobicity and carbon pools (Paul et al., 2002; Bargués Tobella et al., 2014).

Soil litter layers and associated surface infiltration rates can be restored quicker than the organo-mineral aggregates, root channels and soil biota of the upper soil layers that are needed in larger rainfall events. Recovery of interflow will depend on soil macroporosity rather than surface characteristics and will take longer (Bruijnzeel, 2004; Ghimire et al., 2014b). The time frame (e.g., years, decades) at which forestation (used here to refer to an increase in tree cover, regardless of previous landuse, methods or species used) can restore soil infiltration capacity remains an active research frontier (Ghimire et al., 2014b), with a range of site-specific results (Marín-Castro et al., 2017; Qazi et al., 2017; Zwartendijk et al., 2017).

Groundwater

Near-surface water – also called water table – is dynamic and is affected by biotic (e.g., vegetation type, leaf area, rooting pattern) and abiotic factors (e.g., precipitation timing, intensity, and amount; air and soil temperature). It can be an important contributor to the water supply at all temporal and spatial scales (Issar and Simmers, 1990; Lerner et al., 1990). Aquifer water, beyond the reach of current vegetation, on the other hand is typically considered to be reflecting a much longer history ('fossil';

decades to millennia) of recharge, having a composition that is often isotopically different to near-surface water. Near-surface water can be indirectly affected by aquifer water if there is hydrological contact between the two. If aquifer water is used for human activities, the water enters the dynamic hydrologic cycle while the aquifer from which the water originates may be permanently reduced or depleted (Custodio, 2002; Konikow, 2013). Lateral groundwater flow, which is generally simplified or excluded in Earth system models, is important in many landscapes and may provide a missing link for reconciling observations on stable isotope patterns and global models of terrestrial water fluxes (Maxwell and Condon, 2016).



2.2.3 Atmosphere, Oceans and Terrestrial Vegetation: Global Water Fluxes

Global water cycle

The hydrologic cycle has been described as such for hundreds of years (Box 2.2), but most of hydrology has been based on the perspective that incoming precipitation is seen as an external variable rather than a variable that both influences and is influenced by vegetation.

Two and a half percent of the world's water is freshwater, with the largest proportion of freshwater existing in glaciers and permanent snow (Shiklomanov, 1999). Water available in streams, rivers, lakes, (surface and subsurface beyond reach of root systems) and reservoirs is considered blue water and has been the historical starting point of hydrology. However, on average, only about 35% of precipitation becomes blue water, with the other 65% used on-site by vegetation as green water (Falkenmark and Rockström, 2004; 2006). Blue water can be used for irrigation, drinking water or industry, while green water is used by plants for production of biomass (Sood et al., 2014). Recently the term 'rainbow water' has been suggested as atmospheric moisture, which is the source of all blue and green water, and the direct destination of all evapotranspiration (van Noordwijk et al., 2014a).

Partitioning of precipitation over streamflow ('blue water' – integrating overland, interflow and groundwater-based pathways) versus evapotranspiration by vegetation ('green water'), and the subsequent use of blue water downstream were the primary concern for science as well as practitioners. While it is hard to imagine how a national economy would be managed if it considered only monetary flow rather than a monetary cycle, the full hydrologic perspective has been slow to emerge in quantitative studies. The last two decades have seen major progress, however, facilitated by global data sets that reconcile measured atmospheric moisture flows, precipitation and evapotranspiration, supported by models to fill gaps (Trenberth et al., 2011). These datasets themselves are subject to improvement and refinement (van der Ent and Tuinenburg, 2017), but allow direct comparisons of atmospheric moisture concentrations, air movement (wind), precipitation and evapotranspiration, over oceans as well as land.

Box 2.2

Water cycle, forest-climate relationships and desiccation theory

Around the time William Harvey clarified blood circulation in the human body (1628), the study of plants found water to move mostly from the roots in the soil to the leaves where it evaporated. It was clear that water in the soil derived from rainfall, but where did the rain originate? The physics of evaporation and condensation made clear that water vapour, although invisible, was the 'missing link' in the hydrologic cycle, but how far and how long did it travel as water vapour before returning as rainfall? The idea of a hydrologic cycle composed of a 'short cycle' (over land) and a 'long cycle' (involving oceans) was born (Perrault, 1674; Nace, 1975). Around 1693, the astronomer Edmond Halley asserted that evaporation from the oceans was sufficient to explain all rainfall, strengthening the case for the 'long cycle'. Stephen Hales ('Vegetable Staticks' 1727) quantified transpiration, leaf areas and root lengths, and consolidated the understanding of plants as part of the hydrologic cycle. Around the same time, John Woodward started to link vegetation and climate through the hydrologic cycle. This became the basis of the 'desiccation' theory (Grove, 1996). In the 18th century, the effects of deforestation on small islands (St Helens, Mauritius, Tobago) used as stop-overs in the Asian-Europe trade became clear: rainfall was affected. A speech by Pierre Poivre in 1763 in Lyon may well have been the start of widespread climatic concerns over human impact on (tropical) forests. While widespread forest clearance by European settlers in temperate North America was seen by them as climate improvement, replacing damp air by healthier drier air, similar effects in the tropics were seen as negative and forest protection policies started in Mauritius found their way in the French and English colonial expansion in the tropics (Grove, 1994). The desiccation theory became part of the discourse in Africa, undergoing drastic changes after its incorporation in the colonial world, as documented by Endfield and Nash (2002). A specific form of the desiccation theory became the basis of explanations for the historical decline of land productivity in the Middle East, cradle of cereal-based agriculture (Kubat, 2011).

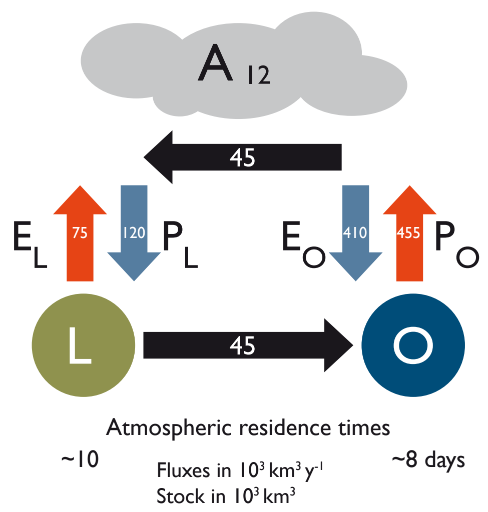
Box 2.3

Global water balance

Based on global data averaged over at least ten years, Figure 2.3 suggests a net ocean-to-land transfer of around 45,000 km^3/year balanced by a similar return flow of rivers and groundwater into oceans. As the annual precipitation over land is around 120,000 km^3/year, the net contribution of terrestrial evapotranspiration to terrestrial precipitation is, on average (120 - 45)/120 = 75/120 = 63%. If one would be able to 'tag' the water molecules from the two sources (land and ocean) of evapotranspiration (which isotope analysis allows only to a very approximate degree) one may find that the fraction of precipitation most recently derived from land rather than oceans varies between 13% (if atmosphere is fully mixed) and 63% (if there would be no land to ocean transfer of atmospheric moisture). The relevant point is that an average water molecule crossing the ocean-to-land boundary in the atmosphere may fall 2.7 (120/45) times as precipitation over land, once as original ('long cycle') rainfall plus 1.7 (2.7-1) times as terrestrially recycled ('short cycle') rain, before flowing back to the ocean in a river. There is no compelling reason why this is not either more (which would imply more rainfall) or less (less rainfall), even if the conditions of the oceans do not change. This is the core of the 'hydrologic space' argument posed by Ellison et al. (2012).

A first estimate of the global mean residence time is obtained by dividing the time-averaged stock of precipitable water (i.e. 12,000 km^3) by the mean daily average precipitation (530,000/365 km^3/day), yielding 8.2 days. Spatial variation around this average has been mapped (van der Ent and Tuinenburg, 2017), with a more accurate global mean of 8.9 days as current estimate. There is however some uncertainty in this conceptual model and its numerical results, linked to assumptions about the degree of layering of atmospheric transport.

Figure 2.3



Source: Own elaboration simplified from van der Ent and Tuinenburg (2017) where uncertainties of the various numbers are discussed

Summary of current understanding of global water cycle involving atmosphere (A), land (L), oceans (O), precipitation (P) and evapotranspiration (E) as key exchange processes

Short and long cycle rain

The short cycle only involves terrestrial systems and the atmosphere (Figure 2.3). In contrast, the long cycle includes atmospheric moisture that is derived from both terrestrial and ocean sources. Current understanding of the global cycling of water between atmosphere, oceans, and land areas is based on a combination of data on evapotranspiration, precipitation, air movement, and the presence of 'precipitable water' (Bosilovich et al., 2002; 2011; Trenberth et al., 2003; Dirmeyer et al., 2009; Gimeno et al., 2012). Uncertainty around the long-term average values for the global balance is within a few percent of the estimates provided, as a number of different models used in combination with empirical data provide similar results (van der Ent and Tuinenburg, 2017).

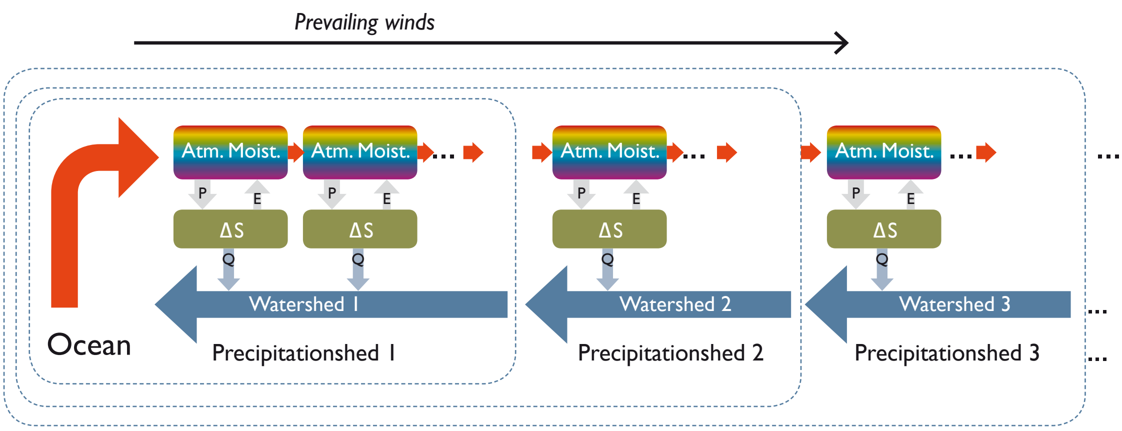
The higher the rate of evapotranspiration, the more a land area contributes atmospheric moisture to the short cycle. Land covers that excel in this function include open water, wetlands, irrigation agriculture, and forests (Ong et al., 2015). On average, forests on sufficiently deep soils can be expected to match the potential evapotranspiration of a site to the degree that precipitation allows, with little loss to rivers until this potential is reached. In other vegetation, part of rainfall comes in amounts that cannot be immediately absorbed by the soil and flow into the river, while vegetation may not be present throughout the year and shallower roots cannot fully use the soil reserve (Black et al., 2015; Bayala and Wallace, 2015). On average, the difference between forests versus other vegetation was estimated by Zhang et al. (2001) to be around 200 mm/yr. Spracklen et al. (2012) showed that rainfall is statistically associated with passage of air masses over forest in the days preceding a rainfall event, with the specific mechanisms still subject to debate (Spracklen and Garcia-Carreras, 2015).

Precipitationsheds

Watersheds are the land areas that contribute water to a given river, considering precipitation as the start of a flow (rather than cycle). Starting one step earlier in the cycle, precipitationsheds are the upwind surfaces of the Earth (whether oceans or land areas) that provide evaporation that later falls as precipitation in a given location (for example, a watershed). The source of atmospheric moisture responsible for, say 95%, of precipitation in a specified location (a point, a catchment, a nation or a region) provides an operational definition of these precipitationsheds (Keys et al., 2012), with recent specifications provided for countries and regions (Keys et al., 2017; Wang-Erlandsson, 2017). The precipitationshed of a watershed is considerably

Conceptual model of the water balance along a series of spatial units following the prevailing winds from an ocean-land interface

Figure 2.4



P = precipitation Q = river discharge E = evapotranspiration ΔS = variable storage term

Source: Authors' own elaboration based on Keys et al., 2012

larger than that watershed itself, and typically contains some part of the global oceans plus parts of one or more terrestrial watersheds (Figure 2.4). For example, watershed 3 in Figure 2.4 can contribute water to watershed 2, but its precipitationshed can include the ocean plus watersheds 1, 2 and 3. Thus, shifting the question from "what happens to the precipitation that a watershed receives?" to "where does this precipitation originate?", and hence "what factors might influence variability and trends?", implies a much stronger regional and global dependence and influence of forest-water relations.

The size of precipitationsheds depends on wind speeds and residence times of atmospheric moisture. As shown by van der Ent et al. (2010), depending on the location relative to global circulation patterns and the shape and size of continents, terrestrially evapotranspired water has a probability of returning as rainfall over land that varies between 0 and 100%. For any given location the uncertainty in this estimate is relatively small (van der Ent, 2010). Similarly, the percentage of rainfall in any location derived from terrestrial rather than oceanic sources varies from 0 to 100% with location, but uncertainty of the location-specific estimate is small.

Prevailing winds together with atmospheric residence time determine moisture recycling (van der Ent, 2014; van Noordwijk et al., 2014a; Ellison et al., 2017). The net transport distances of atmospheric moisture during a mean residence time of around eight days vary from less than 100 to several thousand kilometres. Strong short cycle precipitation in the Amazon and Congo basins and on the large island of Borneo is associated with low wind speeds[1].

The telecoupling (or spatial dependency of processes) that is quantified in a precipitationshed has geopolitical implications that only recently have been explored from a policy perspective (van der Ent et al., 2010; van Noordwijk et al., 2014a, 2016; Ellison et al., 2017; Keys et al., 2017). These are discussed further in Chapter 7.

Vegetation effects on precipitation

Satellite observations and atmospheric trajectory modelling increasingly permit research to disentangle the origin and immediate drivers of growing-season precipitation, and the extent to which ecoregions themselves contribute to their own supply of rainfall (van der Ent et al., 2010). While the amount of water recycled varies between wet and dry years, the recycling ratio increases in dry years (e.g., Miralles et al., 2016). For example, as much as 25% of basin-evapotranspired moisture may be recycled within the Congo basin (Dyer et al., 2017), with further rainfall occurring elsewhere. Recent analysis of rainfall records for Borneo (McAlpine et al., 2014) showed that watersheds with >15% forest loss had a >15% reduction in rainfall, as maritime influences are limited and measured wind speeds low. Weng et al. (2018) identified parts of the Peruvian Amazon and western Bolivia as the atmospheric moisture sink areas most sensitive to land use change in the Amazon. Water tagging studies indicate that continental recycling of water explains more intraseasonal variations in moisture in inland areas than in coastal areas (Risi et al., 2013). In the Amazon, rainforest transpiration enables an increase of shallow convection that moistens and destabilises the atmosphere during the initial stages of the dry-to-wet season transition, which drives moisture convergence and wet season onset 2-3 months before the arrival

[1] This can be verified for any part of the world at any day on a website such as www.windy.com for wind speeds at a standard height of 80m above the land surface.

of the Intertropical Convergence Zone (ITCZ) (Wright et al., 2017).

Variation in precipitation and the frequency of extreme events is likely to be as important as the annual mean precipitation. Degu et al. (2011) described cases where the construction of manmade reservoirs induced local extreme rainfall with negative effects. Such extreme events may be related to a relative scarcity of rainfall triggering agents – as the presence of these would induce more frequent and moderate precipitation rather than cloudbursts. This, however, represents the frontier of current science, as it requires atmospheric physics, chemistry, biology, and particle transport to be reconciled with global circulation models.

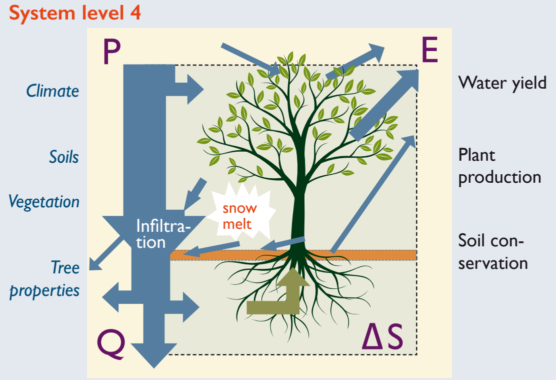
Human modification of the global water cycle

Humans modify the hydrologic cycle through the withdrawal of blue water for agricultural (92%), domestic (4%), and industrial (4%) uses from lakes and rivers (Hoekstra and Mekonnen, 2012). Partly to support these abstractions, humans affect the flow of water in the landscape through the construction of reservoirs for hydropower, flood control and irrigation. In addition, humans modify the hydrologic cycle through land use/land cover change. Human use of river flow in many cases (most directly in case of water abstractions for agriculture) leads to further evapotranspiration, making the blue versus green water partitioning (Falkenmark and Rockström, 2006) highly scale dependent.

Many human activities based on water use do not lead to evaporation but to impaired water quality, as described in the grey water footprint (Hoekstra and Mekonnen, 2012). This involves both point sources of pollution (e.g., industry or residential wastewaters) and diffuse sources of pollution (e.g., agricultural chemical and erosion loads to water). New insights on human influence on precipitation through land cover change, have yet to be incorporated in such footprint estimates.

2.2.4 Precipitation, Evapotranspiration and Discharge: Water Balance and Buffering

System level 4



P = precipitation E = evapotranspiration
ΔS = change in storage Q = river discharge

Box 2.4

Water balance equation

The water balance equation is:

$Q = P - E - \Delta S$

where Q = streamflow, P = precipitation (including rain, snow, cloud water interception), E = evapotranspiration, and ΔS = change in water storage. P, E and Q are expressed as depth (mm) per unit of time (day or year). ΔS can, depending on context, be split into ΔSS (change in soil water), ΔSN (change in snow and ice water storage, where snowfall is part of P), and ΔGWS (change in groundwater stores). The change in water storage in plants (ΔSP) may be non-negligible (Dietrich et al., 2018).

Box 2.5

Paired watershed experimental studies as gold standard of forest hydrology

Paired watershed experimental studies became key to the development of forest hydrology as a science, a century ago. Typically, data collection on at least two similarly-sized watersheds starts a few years before a major intervention is applied to one of the watersheds (i.e., the calibration period) with the other serving as a control. The response is monitored for as long as it takes until the difference in hydrologic response between the sub-catchments has disappeared. Data from paired watershed experiments have been mostly obtained in temperate moist climate zones (e.g., Hibbert, 1967; Bosch and Hewlett, 1982; Andreassian, 2004; Jackson et al., 2005). Where sets of paired watershed experiments have been compared over time with various treatment intensities, short-, medium- and longer-term effects of forest change on water yield have been attributed to changes in the E and ΔS terms of the water balance equation (Scott et al., 2000; Webb et al., 2012). Critiques of existing paired watershed studies often refer to the absence of 'mosaic' effects, where treatments are applied uniformly while in the real world many intermediate degrees of tree cover or mosaics are expected. The results from paired watershed experiments cannot be directly applied in large watersheds (>1,000km^2), as several scale-dependent processes need to be factored in.

Linking ecosystem structure and function

At the scale of a patch of land, the hydrologic cycle is reflected by three long-distance, one-way fluxes (precipitation, evapotranspiration and contributions to streamflow) and local two-way exchanges with water stored in soils, plants and/or snowpack. Four key ecosystem structure attributes (leaf area index, rooting depth, litter layer and soil macroporosity) determine vegetation effects on flow pathways, buffering and flow regime via the basic water balance equation (Box 2.4).

Scale and scaling

Paired-watershed experiments (Box 2.5), which test the effect of forest conditions on hydrology, are typically conducted in small watersheds, usually less than 100 km^2.

Recent advances in ecohydrology include scaling water fluxes from the leaf to the watershed and landscape, the effects of plant-soil interactions on soil moisture, and the influence of plant water use on streamflow regimes (Asbjorssen et al., 2011). Conceptually, studies of forests and water connect spatial scales from the leaf to the globe, and temporal scales from hours to multiple decades. Spatial scales of interest range from hillslopes and forest stands (0.001-0.1 km^2), to forest management units and small watersheds (0.1 to 10 km^2), meso watersheds (10 to 1,000 km^2), large watersheds (1,000 to 10,000 km^2), regions (10,000 to 1,000,000 km^2), and to continents and the globe. Reaching numerical agreement across scales is challenging (van Noordwijk et al., 2004, 2015d).

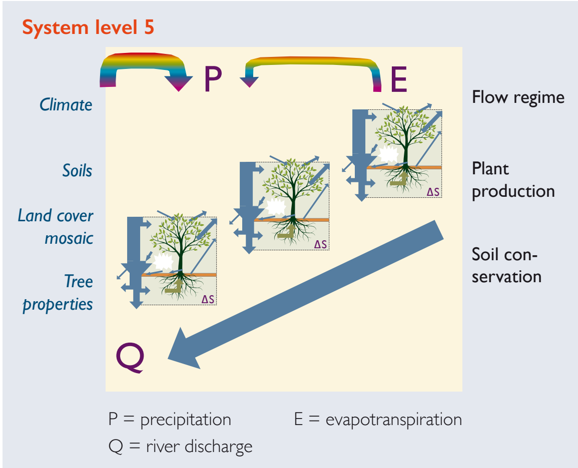
Annual means of precipitation, evapotranspiration, and streamflow may scale with area, but peak flows (defined as the "maximum instantaneous discharge of a given stream") have been found to scale with area to the power 0.7 (Rodríguez-Iturbe and Rinaldo, 2001) or 0.8 (Lin and Wei, 2008). Peak flows relative to mean flows decline with area: for an area that is 10 times larger, the mean flow will be 10 times larger, but the peak flow is expected to be five (equal to 10 to the power 0.7) times larger, so the peak-to-mean ratio halves. The scaling parameter (and its variation across landscapes) reflects both flow buffering in larger watersheds (with greater likelihood of riparian wetlands beyond head catchments) and spatial correlation of peak rainfall events (high for frontal rains, low for thunderstorms). Flood risks and its determinants strongly depend on scale of consideration (van Noordwijk et al., 2017a).

There have been many attempts to develop scale transfer functions, clarifying scaling rules for hydrologic variables and hydrologic effects across different sized watersheds (Blöschl and Sivapalan, 1995; Hrachowitz et al., 2013). Gupta and Waymire (1990) introduced the concepts of simple scaling (e.g., area-based or scale-independent fractal rules) and multi-scaling (more complex scale-dependent rules) to describe spatial structures of rainfall and floods (Blöschl and Sivapalan, 1995). Gupta and Dawdy (1995) showed that floods exhibit simple scaling in snow-dominated watersheds and multi-scaling in rain-dominated watersheds.

Evaporation versus transpiration

In the absence of a litter layer, soil evaporation can be a significant part of total evaporation in some forest types (Raz-Yaseef et al., 2010). At the forest stand and watershed scales it is difficult to distinguish evaporation from soils or intercepted canopy moisture from transpiration, but at the global scale this has been recently accomplished by combining water balance and isotope data. Globally, across all vegetation types, transpiration has been estimated to be 64 ± 13% of evapotranspiration (Good et al., 2015). Stand- and watershed-scale studies of water isotopes imply different cycling and terrestrial retention times of water involved in transpiration compared to streamflow (e.g., Evaristo et al., 2015), but such studies have not yet been paired with water quantity measurements to close the water balance. Only 38 ± 28% of surface water is derived from the plant-accessed soil water pool (Good et al., 2015), with the remainder reaching streams by overland flow, but these numbers are likely differentiated by land cover type.

2.2.5 Dynamic Landscape Mosaics: Streamflow



Flow regimes as landscape signature

Most forest hydrologic studies focus on understanding the response of homogeneous forest patches to specific treatments, but the reality is that land cover dynamics involve complex space-time patterns of roads, forest conversion, partial recovery of secondary forests, intensified agriculture, plantations and urbanisation.

The spatial pattern in land cover also matters for surface and subsurface lateral flows, modifying streamflow regimes (water quantity, quality, regularity of flow). The black-and-white language of 'deforestation' and 'reforestation' does not do justice to the many intermediate situations that influence streamflow in complex ways. Land cover transitions (e.g., the loss of natural forest and the subsequent return of trees – planted, spontaneously established and not removed, or spared during land clearing) matter for the four ecosystem structure attributes (LAI, roots, litter, soil porosity), with different response times for above-and belowground changes.

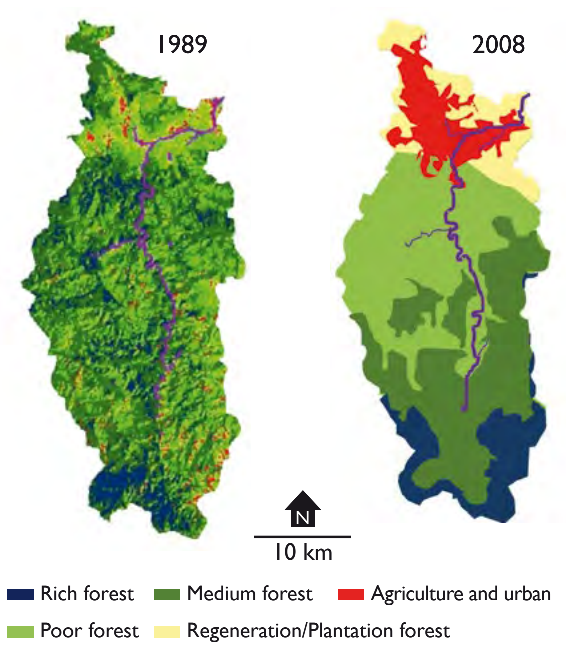
Forest (tree cover) transitions

Forests and tree cover are part of a three-dimensional space, where climatic zones and topography interact with an anthropogenic forest transition (Dewi et al., 2017). Many natural forests are converted, frequently to more open agricultural land cover types, but trees can come back (Meyfroidt and Lambin, 2011), either under pressures of 'push' (increased value of trees used in plantations or as part of agricultural and urban land use mosaics) or 'pull' (urbanisation, land abandonment). Under a 'push' scenario, most of the new trees may be planted, whereas under the 'pull' scenario, most of the trees will be secondary forests with spontaneously established trees (among which invasive exotic species may compete with native pioneer trees) (Ordonez et al., 2014).

The swidden-fallow mosaic landscape in Xishuangbanna of China has been replaced by monoculture rubber plantation
Photo © Xiaobao Han

Figure 2.5

Despite recovery of forests in the headwaters (lower part of map) of the Huong river basin in Vietnam from 1989 to 2008, expansion of agriculture in the lower portions of the basin (upper part of map) exacerbated flooding over the period



Source: Redrawn from Tran et al., 2010

The usual binary classification of land cover into forest versus non-forest, which is used in many studies of forest effects on water, obscures both the effects of forest quality and the effects of spatial arrangement of forest within a watershed, especially in landscapes where swidden/fallow (or secondary forest) cycles are subject to segregation of 'forest' and 'agriculture' (Malmer et al., 2005; van Noordwijk et al., 2012a, 2015b). For example, any possible flood-mitigating effects of forest expansion and growth in the headwaters of a large watershed were overwhelmed by agricultural intensification from the traditional swidden-fallow system in the lower reaches of the Huong basin in Vietnam over the period 1989 to 2008 (Figure 2.5), which experienced a statistically significant increase in the highest yearly flood peak in the lowland. Hence, the spatial distribution and character of forest and tree cover influence hydrologic behaviour in large watersheds, with conditions of the land outside the forest at least as important as that inside remaining forest. Concepts, as specified in the Indonesian spatial planning law, that 30% of forest is needed to guarantee watershed functions, regardless of what happens in the other 70% of land, have little empirical basis, even when occasional studies seem to confirm the 30% estimate (Tarigan et al., 2018).

Special forest niches

Beyond bioclimatic zones, topography is an important determinant of ecosystem structure and hydrologic function of forests and tree cover. Specific forests of interest include the following:

Water towers

Water towers are found at high altitudes and are areas where the ratio of precipitation to evapotranspiration is sufficiently high to generate streamflow. They are often the primary source of streams on which life in lower and drier zones depends (Viviroli et al., 2007). Tropical water towers tend to have relatively high human population densities and rates of forest conversion (Dewi et al., 2017); they thus are hotspots of conflict over water.

Cloud forests

Cloud forests – often the mountain tops of water towers – have a special place in forest hydrology as the vegetation plays an active role in trapping moisture from clouds, attaining higher precipitation than measured by standard rainfall gauges (Bruijnzeel et al., 2011). A recent study of cloud forests in Colombia, however, suggested that low evaporation due to foggy conditions is a key part of streamflow generation (Lawton et al., 2001; Ramírez-Correal et al., 2017a,b), making the continued functioning of such forests dependent on evapotranspiration of adjacent lowlands.

Wetland and riparian forests

For wetlands and riparian forests, factors that control the surface and subsurface flows of water may be at least as important as local precipitation in determining water availability to plants. This includes the large seasonal floodplains of the Amazon basin and smaller parts of many other river systems. Where wet conditions are permanent, peat forests may form based on trees with sufficient root adaptations to live in a permanently anaerobic environment. Wetlands and riparian forests can have an important flow-regulating effect on downstream river behaviour, as long as their water table level is allowed to move up and down. With conversion to agriculture or urban areas, changing water table levels become problematic and engineering solutions externalise the variability, implying a loss of flow buffering functions.

Vegetation around springs and wells

Due to obvious relations with water quality and public health, the vegetation around springs and wells has been protected by locally-developed resource use rules in many parts of the world with national legislation usually formalizing such rules (Galleani et al., 2011; German et al., 2013).

Mangroves

Along marine coastal zones, a specially adapted tree flora forms mangroves (see Box 2.8), providing flood and storm surge protection of the hinterland (Bayas et al., 2011), mitigating sea level rise and coastal erosion, as well as being a spawning ground for coastal fisheries or protecting other important ecosystems, such as seagrass and coral reefs.

Small island forests

On small islands, limited fresh groundwater impacts water availability for forests, agriculture and people (White and Falkland, 2010), making them especially vulnerable to climatic variability. Small island states have been strong advocates of global climate change mitigation, and they also are at the forefront of adaptation discussions (Duguma et al., 2014). For example, the Tobago Main Ridge Forest Reserve (proposed as a UNESCO World Heritage Site)[2] is on record as the oldest legally-protected forest reserve established specifically for water conservation purposes. It was established on April 13th, 1776 by an ordinance which states, that the reserve is "for the purpose of attracting frequent showers of rain upon which the fertility of lands in these climates doth entirely depend."

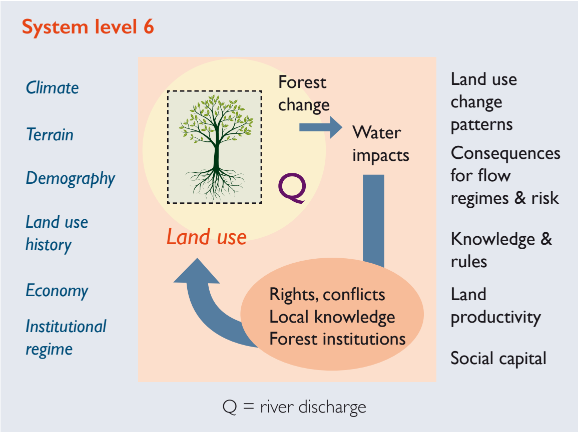
Trees outside forest

With 43% of the world's agricultural lands having at least 10% tree cover[3] (Zomer et al., 2016), the roles that these trees play for the local economy, as well as for the water balance and local climate, deserve attention (Ong et al., 2015). Agroforestry has seen a growing recognition that land use at the interface of agriculture and forestry has much to offer to sustainable development concepts (Garrity, 2004; Prabhu et al., 2015).

Urban trees

Trees and other vegetation in urban areas are essential for rainfall infiltration and storm surge abatement. They function as air conditioners, cooling surrounding air by producing latent heat through transpiration. This ecosystem service per unit biomass may be as high as that of the sparse trees in dry zones. The mechanical instability of urban trees (due to limitations to root development and functioning) is a problem, and the selection of suitable trees for urban environments is a specialised field of science (Pokorny et al., 2003). Perennial climbers on walls may combine the positive roles of a high leaf area index, with the absence of tree and branch fall risks (Alexandri and Jones, 2008); green walls as complements to urban trees have become popular, for example, in Singapore (Magliocco, 2018).

2.2.6 Land and Water Use Rights, Local Knowledge and Forest Institutions: Landscapes



Local rights and forest institutions

The forest (tree cover) transitions described and analysed as statistical phenomena with hydrologic consequences in the preceding system level are in fact a consequence of complex interactions between social and ecological aspects of a dynamic interaction that changes people as well as landscapes.

Rights, conflicts, multiple knowledge systems, the emergence of forest institutions of various types, all controlling what individual actors can or cannot do to forest and tree cover (Freeman et al., 2015), indirectly influence streamflow regimes (van Noordwijk et al., 2015e). In the history of land use change, evolving local institutions on forest and water use rights have restrained private benefit maximisation, often progressing from 'first come, first served' rules towards collective action, stewardship and shared responsibility. In many countries, state-based forest use rights ('concessions') have been applied without reference to local or traditional rights.

The historical evolution of forest institutions in relation to local rights has reflected issues of national security (including shipbuilding, navigable rivers, accessible ports), economic gain (logging), watershed protection (depending

[2] https://whc.unesco.org/en/tentativelists/5646/
[3] Within the global climate convention countries were asked to specify their tree cover threshold (between 10 and 30%) to be used in distinguishing forest from non-forest

on downstream interests), biodiversity protection and conservation, and recreation, with shifts in the public-private balance of power. Conflict resolution, more participatory forms of forest management and transparency of landscape resource monitoring have changed the forest-water relation over time and its role in national development strategies.

Water and forest rights

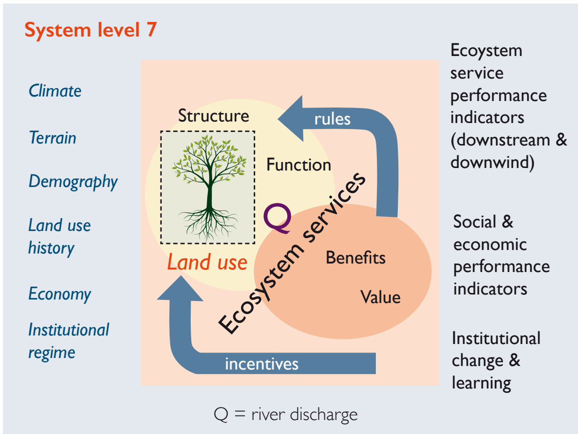
Water is among the resources with the longest history of clarifying public, club, and collective rights and responsibilities. At the most basic level of rights, there is a concept of 'settler rights', where the first to claim establishes a long term right, and a 'riparian right' where all those with land bordering a stream or lake have collective rights and responsibilities to share and manage the resource. Given their military importance, navigable rivers have been claimed by states from the start of codified law. In the establishment of 'forests' as a state resource, the concept of 'terra nullius' (land without settler rights) provided the opportunity, while public concern over water flows became a justification (Williams, 2003; Galudra and Sirait, 2009). With many post-independence nations inheriting strong 'state' claims from indigenous peoples, conflicts have occurred over what are 'club' collective rights, versus 'state' prerogatives. In subsequent 'privatisation' of state claims of resources, e.g., through concessions for water use or drinking water distribution, a new arena for conflicts was opened (Boelens, 2009).

Schlager and Ostrom (1992) in their foundational analysis described five property rights with respect to natural resources: the right to access, the right of withdrawal, the right of management, the right of exclusion, and the right of alienation. Recent stocktaking (Galik and Jagger, 2015) of progress in the understanding of property rights added a sixth category (the right to alter) to those defined by Schlager and Ostrom. Regulating the right to alter land cover and land use is central to efforts to manage public functions of water, alongside private rights to 'harvest' and 'manage'. A delicate balance exists in water resource management between plot-level issues that are better handled with private tenure security versus those that require collective action at the levels of streams and rivers (Swallow et al., 2001). This has become an important issue in South Africa, where the introduction of licences for 'stream flow reduction activities' were introduced to control large scale plantation activities and their downstream impacts (Gush et al., 2002). Climate change provides a new complication at the public/private interface where forest and water resources are involved.

Local and traditional knowledge

Traditional knowledge is typically transferred between generations as part of local culture, whereas local knowledge can be accumulated by a person or community merely by experiencing local conditions for a period of time. Both can involve component (ethnobotany, ethnozoology) and explanatory knowledge (Joshi et al., 2004). There is an inextricable link between traditional ecological knowledge systems and forest-water interactions that emerges from historic ties to cultural landscapes (Xu et al., 2009). For example, many 'globally important agricultural heritage sites' from the Andes and Asian highlands show the complex but coupled linkage between the forest-village-terraced-rice paddy and river systems (Camacho et al., 2010; Jiao et al., 2012) involving local world views, knowledge systems, norms and institutions, trials and innovations, teaching and learning. Various government policies and the expansion of regional and global markets play important roles in shaping the landscape and associated cultural influences (Xu and Grumbine, 2014a). More recently, there has been considerable discussion on ways to integrate local knowledge with government policies for managing forest-water interactions (Jeanes et al., 2006; Xu, 2011; Rahayu et al., 2013; Leimona et al., 2015b).

2.2.7 Social-Hydrological Systems: Ecosystem Services as Valued Human Benefits



Typology of services

The Millennium Ecosystem Assessment (MEA, 2005) has popularised a classification scheme of ecosystem services that is based on the type of human benefits (provisioning, regulating, cultural, supporting) that are derived from functioning ecosystems (De Groot et al., 2002; see Chapter 5).

Both the anthropocentricity of the definition of ecosystem services and the association with economic representation of value and proposed alternative concepts used in the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services studies have become the subject of intensive debate[4] (Tomich et al., 2010; Pascual et al., 2017; Diaz et al., 2018; Braat, 2018; Peterson et al., 2018). Yet, the ecosystem services concept has sparked new ways of combining rule-based approaches with economic incentives. Such incentives may

[4] http://science.sciencemag.org/content/359/6373/270/tab-e-letters

Table 2.1

Water-related ecosystem functions provided by vegetation and potentially perceived as 'ecosystem services'

	Functions	Metrics
Generic		
W1	Water transmission	Total water yield per unit rainfall
W2	Buffering peak river flows	Wet- and dry-season flow persistence (van Noordwijk et al., 2017a,b) or flashiness (Holko et al., 2011) River discharge per unit above-average rainfall
W3	Gradual release of stored water supporting dry-season flows	Dry-season flow persistence Aquifer recharge
W4	Maintaining water quality (relative to that of rainfall)	Pollutants per unit volume of water Biological water quality indicators
Site-specific		
W5	Stability of slopes, absence of land-slides	Woody roots for topsoil binding and anchorage Non-erosive pathways for overland flow
W6	Controlling soil loss by erosion	Surface runoff pathways Volume of trapped sediment in filter zones Infiltration of topsoil and subsoil (macro porosity due to worms and roots)
W7	Microclimate effects on air humidity, temperature and air quality	Wind speed; reduction in daily maximum temperature; land surface temperatures
W8	Coastal protection from storm surges, tsunamis	Retardation of waves, reduced maximum run up height
Frontier of science		
W9	Ecological rainfall infrastructure and biological rainfall generation	Recycling of atmospheric moisture; height above vegetation of rainfall generating events; ice-nucleating agents

Source: van Noordwijk et al., 2016; Lusiana et al., 2017

'nudge' (Thaler and Sunstein, 2008) land use decisions, rather than impose them. Wunder (2015) differentiated payments for ecosystem services from regulation-based (command-and-control) efforts to protect and enhance ecosystem services by emphasising that payments for ecosystem services are a realistic, voluntary, and conditional contracts between at least two parties. In practice, a balance between 'efficiency' and 'fairness' had to be found to make the concept operational (van Noordwijk et al., 2012b; Kerr et al., 2014; Leimona et al., 2015a; Lapinski et al., 2017).

A classification of water-related ecosystem services that is closer to hydrologic function (rather than the way people benefit, as in provisioning, regulating or cultural services) has been used in recent reviews (Table 2.1; van Noordwijk et al., 2016; Lusiana et al., 2017).

2.2.7.1 Generic Functions

Function W1: Water transmission

The commonly observed association of streamflow and forests is the combined effect of the high-precipitation places where forests tend to occur and the way water is partitioned over streams and recycled to the atmosphere (Box 2.6). When total water yield is the primary performance criterion for a watershed (e.g., where a large reservoir is to be filled and sediment loads are not an issue), less trees will lead to more blue water. Overall, studies in both small and large watersheds indicate that removal of forests reduces evapotranspiration (ET) and increases streamflow, while reforestation does the opposite (Moore and Wondzell, 2005; Andréassian, 2004; Li et al., 2017a).

In a summary of hydrological research in 30 long-term ecological research sites in the US and Canada (Jones et al., 2012; Figure 2.6 A and C), the E_{act}/E_{pot} ratio was close to the P/E_{pot} ratio when the P/E_{pot} ratio was less than 1, indicating water-limited ET and plant growth, and around 1 when the P/E_{pot} ratio was greater than 1, indicating energy-limited ET. E_{act}/E_{pot} ratios > 1 point to uncertainties in the calculation of E_{pot} (Lu et al., 2005), timescales where ΔS is not negligible, or situations where groundwater flows support E that are not accounted for in P. These E_{act}/E_{pot} ratios for natural vegetation in the dataset of Jones et al. (2012) are higher than the average for 'forest' in the Zhou et al. (2015) data set, and may point to heterogeneity of what is included in forests when compared to non-forests.

Canopy interception may contribute to higher E_{act}/E_{pot} ratios of forests compared to other vegetation. Water

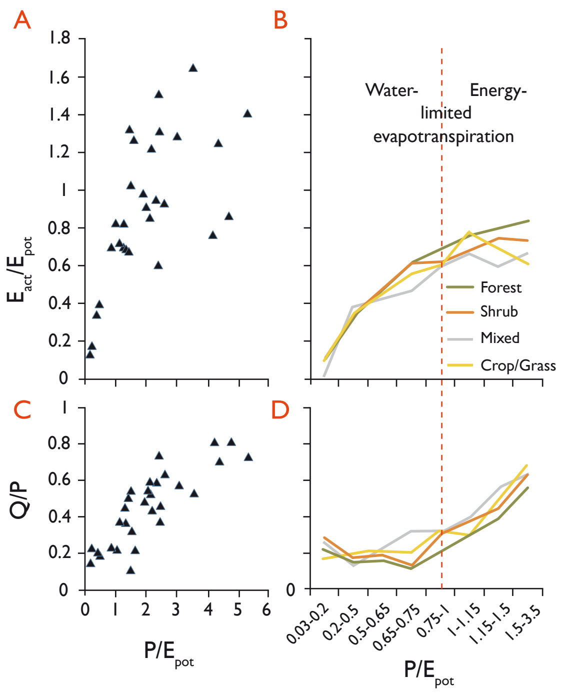
Box 2.6 Figure 2.6

Blue water yield in relation to vegetation and precipitation

Forests occur mostly in places with relatively high precipitation. However, relative to most other vegetation, evapotranspiration for a given precipitation is higher in forests, implying less water transmission to streams (and more to 'rivers in the sky'). The net effect on streamflow of these two findings has been debated. In the most comprehensive global dataset of watershed studies, where P, Q, and E have been assessed across the main continents at annual time scales for a range of land cover types (Zhou et al., 2015; Zhang et al., 2017), an approximately constant water transmission or Q/P ratios of 32.7%, 34.5%, 34.5% and 30.5% were obtained for forest, shrub, mixed land uses and crops/grass, respectively. The forests were associated with the highest precipitation, with P/E_{pot} ratios for the four land covers of 1.17, 1.07, 0.92 and 0.88, respectively. This compensated for the higher E_{act}/E_{pot} ratios of the four land covers: 71.8%, 59.2%, 53.3% and 55.2%, respectively. Both the Q/P and E_{act}/E_{pot} ratio depend on local climate (Figure 2.6). The weighted average based on the global distribution of P/E_{pot} ratios (Figure 2.2A) indicates a global mean water transmission fraction for forests and non-forests of 33.8 and 40.8%, respectively (and E_{act}/E_{pot} ratios of 70.4 and 60.9%, respectively). In the wettest part of the data range, the difference in E_{act}/E_{pot} ratio is up to 20%. The averages for the four land cover classes are midpoints of a rather wide statistical distribution, and the stated differences may not hold for specific land covers compared in a given location. The lower E_{act}/E_{pot} ratios for crops reflect annual assessments; within the growing season, closed crop canopies can operate at E_{act}/E_{pot} ratios of close to 1 if the soil is sufficiently moist.

Figure 2.6 D shows that the average Q/P ratio for all vegetation types at low rainfall does not drop below 15% and may actually increase when the lowest P/E_{pot} ratios are considered. This is likely due to peak rainfall events that exceed the instantaneous infiltration capacity of the soil.

Part of the variation in annual data analysis like this is that groundwater stocks carry over from wet to dry years, depending on substrate and topography (Condon and Maxwell, 2017). Gudmundsson et al. (2017) challenged the continuous functions used in the analysis of these data by Zhang et al. (2017); the current analysis is based on means for P/E_{pot} class to avoid the assumptions of continuous functions.



Source: Authors' own elaboration based on datasets from Jones et al., 2012; Zhou et al., 2015

Relation between precipitation relative to potential evapotranspiration (P/E_{pot}) and actual relative to potential evapotranspiration (E_{act}/E_{pot}) (A, B), and relation between streamflow Q relative to P (C, D) for two datasets. A and C are from 30 long-term ecological research sites in the US and Canada (Jones et al., 2012) and B and D are from a global dataset (Zhou et al., 2015).

intercepted by forest canopies may evaporate without being measured (Sahin and Hall, 1996; Carlyle-Moses, 2004; Brown et al., 2005; Wei et al., 2005, 2013). Leaf area index, thickness, and characteristics (i.e., waxiness, hairiness and drip tips) determine the absolute amount of water stored after any precipitation event (Gash, 1979), and thus water available for evaporation from the canopy.

Forest species composition and age influence the E_{act}/E_{pot} ratio of forests. Paired watershed experiments (Box 2.5) have shown larger effects on streamflow for changes in evergreen forest than for changes in broadleaf deciduous forest (Bosch and Hewlett, 1982; Brown et al., 2005), but age of the experimental stands may have influenced these results (Jones and Post, 2004). Changes in forest water yield over time have been attributed to shifts in forest species composition between low and high-water using (mesophytic) tree species (Caldwell et al., 2016; Elliott et al., 2017).

In a review of forestation effects on streamflow by Filoso et al. (2017), most studies reported decreases in water yields following the intervention. However, most studies referred to plantation forestry rather than forest restoration with mostly slower growing native species. Furthermore, studies were especially limited for the humid tropics and subtropics. One of the challenges in interpreting such data is that actual precipitation over forests is not readily measured, as standard climate stations measure away from trees, while wind-corrections on rain gauges are not consistently applied across data sets (Chang, 2006). If regional vegetation influences precipitation, then its effects are implicit in the data and may not be explicitly considered in the data analysis and conclusions.

Function W2: Buffering peak river flows

Unanticipated floods create major damage (Brauman et al., 2007; Bishop and Pagiola, 2012; Winsemius et al., 2013) and the human and economic costs of floods, particularly where cities are built on floodplains, can be huge (Farber et al., 2002; Turner and Daily, 2002). While floods may originate from factors exogenous to the landscape of interest (such as heavy precipitation, earthquakes inducing dam collapse, tsunamis or coastal storm surges (van Noordwijk et al., 2017a)), they may also be caused by land use patterns, such as low infiltration capacity, limited soil water storage, logging practices, forest roads (Wemple and Jones, 2003) or accelerated snow melt (Jones and Perkins, 2010; Schulte et al., 2015). Avoided flood damage may translate into high economic value, justifying an 'insurance' approach to maintaining or restoring forests, if effects can be sufficiently quantified.

Forests and their management can affect the peak flows that cause flooding downstream (Rogger et al., 2017; Jacobs et al., 2018), but the degree to which this function is achieved in any given context remains subject to debate and uncertainty. Most of what has been presented from correlational studies as direct evidence of a relation between forest loss and increased flood risk has alternative interpretations in relations with human demography and remains contested (van Dijk et al., 2009). However, the analysis of Malaysian data by Tan-Soo et al. (2014) with adequate controls of confounding factors showed increased flood risk after conversion from natural forest to plantation crops and urbanization. Elsewhere natural forest was shown to be more effective in reducing floods than plantations on former agricultural lands (Nadal-Romero et al., 2016).

Forestation may reduce flooding by rapidly increasing evapotranspiration and enhancing infiltration more slowly once soil macroporosity increases (Bresson and Valentin, 1993; Ilstedt et al., 2007). The relative importance of these two effects varies with context, and is a challenge for analysis of empirical data, as is the statistical distribution of peak precipitation events that are the direct cause of floods.

Efforts are needed to relate the more readily observable response to less-extreme events to what can be expected in extremes. An index of 'flashiness' of streams has been used in evaluating streamflow records (Baker et al., 2004; Holko et al., 2011); it quantifies the relative day-to-day changes in flow. A recently introduced method goes a step further, as it provides a direct link between the part of a peak rainfall event that comes directly into the stream and the 'flow persistence' (flow regularity) that can be observed in the day-to-day changes in flow (van Noordwijk et al., 2017a, b). Instantaneous peak flow, which is relevant for flood risk management, can be derived from the maximum mean daily flow in various ways (Jimeno-Sáez et al., 2017), connecting flood assessments to daily flow accounting schemes. New ways of estimating flow duration curves for ungauged catchments have been developed (Poncelet et al., 2017) using geographic similarity.

In the temperate zone, floods can be caused by snowmelt in spring as well as by peak rainfall events in summer, with different opportunities for forests to provide function W2. The energy relations of forests also cause snow to accumulate and melt differently than in openings, so forest cover may mitigate snowmelt peaks (Bergström, 1995; Seibert, 1999; Varhola et al., 2010). The first quantitative studies that related forest cover to flooding risks were carried out in Switzerland in the 1920s (Mather and Fairbairn, 2000). By comparing flooding responses in the valleys with varying degrees of conversion of forests to alpine meadows and/or agricultural lands, a safe threshold of forest cover of 30% was derived. In valleys with more than 30%, forest snowmelt was more gradual and flooding risk was lower, than in valleys where all snow could melt simultaneously.

Although it is difficult to assess statistical significance for rare, extreme events, forest harvest was associated with significant increases in peak flows in both small and large (100 - 1,000 km^2) basins (Jones and Grant, 1996; Jones, 2000). Partial forest harvest may produce smaller effects on peak flows (Troendle et al., 2001). Forest harvest also is associated with increases in peak flows in watersheds ranging from 1 to 1,000 km^2 (Jones and Grant, 1996). Engineering measures (dams, reservoirs, canals and dykes) can significantly alter the flow regime of streams (Poff et al., 1997). The life expectancy of such structures depends, however, on the sediment load of incoming streams and thus on upper watershed conditions (Graf et al., 2010).

Function W3: Gradual release of stored water supporting dry-season flows

Gradual release of water stored in the 'sponge' of forest soils primarily depends on the geomorphological context (Section 2.2.2) rather than on the more visible part of the forest.

After Hamilton and King (1983) and Bruijnzeel (1990; 2004) drew attention to the soil, rather than the trees, as the most hydrologically relevant part of a forest, forestation research has tried to clarify the increase in infiltration that is needed to have a positive effect on dry-season flows, offsetting additional water use by fast-growing trees. While annual streamflow is likely reduced by forestation, effects on groundwater release are uncertain, as they depend on the balance of infiltration and (deep) water uptake by trees (Ma et al., 2009; 2010).

Forest soils typically have a litter layer that retains water on the surface and increases the time available for infiltration and protects soil surfaces from the erosive capacity of direct rain droplets (e.g., Hairiah et al., 2006). In peri-urban environments, leaf litter, root channels, and animal burrows can detain and absorb water, reducing erosion and turbidity (Seitz and Escobedo, 2011). Loss of forest cover is associated with loss of soil organic matter and associated aggregates that lead to reduced moisture holding capacity (Allen, 1985).

Intermediate tree densities provide a solution for the tradeoff between enhanced infiltration and increased water use due to trees (Ilstedt et al., 2016). When clearing land for crop production, farmers in the parkland agroforestry systems of the Mediterranean and the Sahel retain

Box 2.7

Riparian forests and water quality

Forests can have direct influence on water quality in streams, including temperature, sediments, nutrients, and biological oxygen demand (Stelzer et al., 2003; Moore et al., 2005). First, direct microclimate effects influence stream temperature, critical for 'cold water' fish (Groom et al., 2017). Secondly, riparian forests act as buffer zones that filter sediment, nutrients, and contaminants before they reach the water (van Noordwijk et al., 1998a; Ranieri et al., 2004). For example, riparian forests can retain soil and limit sediment erosion that would otherwise transport unwanted mineral soil particles to the water, consequently darkening and decreasing its quality (Neary et al., 2009). Nutrients (nitrogen and phosphorus) and contaminants (pesticides and pathogens) that could also be transported to the water can be adsorbed in the forest soils or taken up by plants and microbes (Gilliam et al., 2011). Thirdly, organic matter from forests gets washed into waterways (Para et al., 2010). It provides shade, which prevents excessive growth from aquatic plants and algae, and consequently regulates oxygen levels and water clarity (Thrane et al., 2017). Additionally, these terrestrial inputs to the food web are either directly ingested by zooplankton and fish or are decomposed by sediment microbes that release bioavailable carbon into the water (Berggren et al., 2009). Together, these processes support as much as 20% to 85% of secondary production in freshwater systems (Karlsson et al., 2012). In order to meet their energy requirements, biota in less productive waters are particularly dependent on these terrestrial subsidies that supplement within-lake primary production (Tanentzap et al., 2017). The surrounding species of trees, land-use, seasonality, and the communities present within the water regulate how strongly these terrestrial inputs will impact the aquatic ecosystem (Cole et al., 2006).

old trees, especially those of a number of species with valued products (fruits, edible young leaves; Bayala et al., 2015). The ratio of beneficial effects and water use is likely higher for old than it is for young trees (van Noordwijk and Ong, 1999). Actual tree densities may be close to what is optimal from a perspective of groundwater recharge: more trees would imply higher water use, less trees would affect infiltration (Ilstedt et al., 2016).

Function W4: Maintaining water quality

The association between natural forests and good water quality is based on a number of aspects:

- lower sediment loads, as erosion is largely confined to shallow landslides and much of the soil involved can become incorporated in surrounding vegetation rather than reaching streams;
- tight nutrient cycling with little nutrients lost to streams (when compared to agricultural land with recurrent nutrient inputs); and
- scarcity of pollutant point sources, although bacteria such as *Escherichia coli* can be present whenever vertebrates are in close contact with streams.

However, the general association between forest conditions and good water quality needs to be contextualised. Retaining riparian zones of native forest can reduce some of the negative effects of plantation forestry on flow regimes and water quality (Little et al., 2015). Relatively small strips of riparian vegetation can act as sediment filters in overland flows from uphill agricultural plots and make a subwatershed behave 'forest-like' (van Noordwijk et al., 1998a; Ranieri et al., 2004) in terms of sediment load (Box 2.7).

2.2.7.2 Topography-Dependent Functions

Function W5: Stability of slopes, absence of landslides

A large amount of literature links forestry to increased occurrence of landslides, debris slides, and debris flows in steep landscapes as a result of logging or forest roads (Swanson and Dyrness, 1975; Swanson and Swanston, 1976; Amaranthus et al., 1985; Wemple et al., 2001; Sidle et al., 2006). Landslides, however, are a natural part of landform evolution, but forest condition and soil type influence their occurrence (Verbist et al., 2010). Landslides are triggered by positive water pressures within soil pores, facilitated by macroporosity and high instantaneous infiltration rates (Sidle and Bogaard, 2016). Vegetation, especially undisturbed native forest, promotes cohesion of steep hillslopes through root systems (Hales et al., 2009), by decreasing peak rainfall intensities through canopy interception and by reducing soil water content through evapotranspiration, which promote slope stability (Turcotte and Malamud, 2004; Sidle and Bogaard, 2016); however, large trees can add weight and increase landslide risks when uprooted by strong winds. Increased land sliding is particularly likely within a window of a decade (or two decades in cold climates) after logging or forest conversion, depending on rates of root decay and root development by new vegetation (Dhakal and Sidle, 2003). Forest cover also modulates avalanche risk on mountains with snowpack; forest conditions that reduce likelihood of avalanche include a crown cover of >30%, the absence of gaps >25 m in length, and an increased terrain roughness associated with standing or downed trees that exceed snow depth (Bebi et al., 2009).

Function W6: Controlling soil loss by erosion

Forests with understory vegetation and intact litter layers have low rates of erosion, but forest harvest and roads increase erosion (Wemple et al., 2001; Sidle et al., 2006). Removal of the forest litter layer increases overland flow of water, and hence, surface erosion (as described for Nepal by Ghimire et al., 2014a). Forest plantations without understory can increase the kinetic energy of throughfall beyond that of rainfall and increase detachment of soil particles as a first stage of erosion (Wiersum, 1991). Riparian forests are particularly important to limit streambank erosion (Verbist et al., 2010). Reforestation has been associated with reduced erosion and sedimentation in major river basin systems in China (Miao et al., 2010; Ma et al., 2014; Yang et al., 2015).

Function W7: Microclimate effects on air humidity, temperature and air quality

Many processes influence how forests and trees outside forest (in agricultural lands or urban environments) affect local air temperature, and effects depend on the climate zone. In boreal forests, a large amount of literature has debated the effects of forest and snow albedo (reflection of incoming radiation), forest change, and climate change on energy balances. Boreal forest albedo is very low both in summer and under snow (Betts and Ball, 1997; Manninen and Stenberg, 2009), contributing to warmer temperature under these forests in winter compared to other vegetation cover types, and these differences are not expected to be sensitive to anticipated climate change, including reduction in snow cover (Kuusinen et al., 2012). Furthermore, the effects of tree cover on reduced night-time cooling can offset day-time effects of increased evapotranspiration (Peng et al., 2014). Differences in albedo between forests and clearings in the tropics are relatively small (Pinker et al., 1980; Teixeira et al., 2015) and cooling associated with evapotranspiration may dominate the energy balance, making forest canopies cool relative to other cover types (Ellison et al., 2017). Cooling effects of trees and open water were first described for 'urban heat islands', but these effects are now recognized in agricultural landscapes with various degrees of tree cover (Bayala et al., 2015; Sida et al., 2018).

Function W8: Coastal protection from storm surges and tsunamis

Coastal protection by mangroves and other forests may well represent the highest ecosystem services of trees per unit tree biomass, as coastal areas can have high human population densities (Box 2.8). Empirical evidence for the benefits of such protection during the December 2004 tsunami in Southeast Asia, however, has been mixed with trees blocking exit pathways for people living between the tree cover and the coast for example (Bayas et al., 2011). Nevertheless, interest in ecosystem-based coastal defence in the face of global change is increasing (Gedan et al., 2011; Temmerman et al., 2013), if only for financial reasons, as construction of alternative protective sea walls is expensive (Gunawardena and Rowan, 2005).

2.2.7.3 Frontier of Science

Function W9: Ecological rainfall infrastructure

Forests and trees outside forest may influence four factors required for precipitation at a given time and place: 1) the presence of atmospheric moisture; 2) phase shifts from vapour to water droplets (clouds); moist air has to get into cooler higher atmosphere layers for ice nucleation (and thus cloud formation) to happen, but just how cold (and thus how high) it has to be depends on ice nucleating agency (e.g., dust and bacteria that live on the leaves of plants) which can increase the temperature threshold (from minus 30ºC in clean air to around minus 5ºC); 3) local capture of atmospheric moisture (ending the atmospheric residence of a specified unit of moisture) that might otherwise move elsewhere; and 4) mass flow of moist air during and between rainfall events that depends on modifications of prevailing winds (Makarieva et al., 2009, 2013).

Capturing atmospheric moisture in plant available form can occur at a number of scales. Water droplets in the air that are too small to fall can be captured by vegetation. For example, in cloud forests, epiphytic lichens, mosses, and hairy leaf structures strip 'horizontal rain' (Holwerda et al., 2006) from the atmosphere. The presence of cloud forests, often the highest parts of water towers (Viviroli et al., 2007; Dewi et al., 2017), can thus actively increase precipitation (Hamilton et al., 1995; Bruijnzeel, 2001; Ramirez et al., 2017; Domínguez et al., 2017; Regalado and Ritter, 2017). The loss of cloud forests can lead to reductions of water yield, opposite to the increases expected otherwise. Locally-generated moisture can also be captured as dew by hairy plants growing in dry environments with large diurnal temperature fluctuations that increase relative humidity at night (Stone, 1957; Zhuang and Zhao, 2017). Dew is a major source of green water rather than blue water (Ben-Asher et al., 2010), but can help in establishing 'ecological rainfall infrastructure' in dry environments (Zhuang and Zhao, 2017). Forest cover may affect cloud height and cloud cover (Millán et al., 2005), slow down winds, and therefore influence the likelihood of rainfall triggering (Fan et al., 2007; Poschl et al., 2010; Pöhlker et al., 2012; Morris et al., 2014, 2016; Bigg et al., 2015).

'Rainfall triggering' tends to have a physical component in cooling that follows the rise of air masses due to turbulence or orographic effects as well as a chemical and biological component. Forests, and especially forest edges, have been shown to influence turbulence and as

Box 2.8

Mangroves and land building in the river deltas

Mangroves often dominate the estuaries of tropical river basins, providing significant services including trapping and accumulating sediments and eventually elevating surface and forming deltas. They are often considered land builders and in many places the accretion rate often exceeds sea level rise. Global estimates of the accretion rate are 4.0 + 3.5 mm/year (Breithaupt et al., 2012), while sea level rise under a high emission scenario ranges between 2.6 and 3.2 mm/year (Church et al., 2011). The rate and extent of accretion depend on the hydro-geomorphic settings of the coasts and estuaries. Tidal range, topography and geological formation of the watershed and coastal areas are important determining factors (Balke and Fries, 2016), as well as anthropogenic influences through coastal development (Alongi, 2008). The unique nature of mangrove root systems not only supports the trees to withstand sea currents and waves but also secures the stability of the coast itself. The ability of mangroves to successfully adapt to changes in sea-level depends on accretion rate relative to rate of sea-level change.

such can bring moist air to heights where it is sufficiently cold to form ice nuclei and raindrops (Degu et al., 2011; Pielke, 2013). Aerosols (e.g., dust or hygroscopic salts as used in ‘cloud seeding’) and volatile organic substances derived from vegetation (Stopelli et al., 2015; Fröhlich-Nowoisky et al., 2016) interact with biological cell wall material (e.g., ice-nucleating bacteria, pollen, fungal spores) that can act as catalysts for ice nucleation (van Noordwijk et al., 2015c; Morris et al., 2016).

Forests influence winds with their frictional resistance tending to reduce wind speeds. Wind speeds over the Amazon, Congo Basin and forested parts of insular Southeast Asia are remarkably low, allowing local evapotranspiration to be recycled as local rainfall before it is transported hundreds or thousands of kilometres.

The mechanism by which developing rainstorms can attract moisture from adjacent areas by creating low pressure systems (Makarieva et al., 2009, 2013) is not yet adequately represented in global circulation models and the debate over its significance continues (Sheil and Murdiyarso, 2009; Sheil, 2018).

The concept of tree planting in order to increase precipitation, such as in ‘Great Green Walls’ in China and the Sahel remains controversial, but recent advances in science make it open to further analysis.

2.3 Research Gaps and Conclusions

2.3.1 Research Gaps

For each of the seven system delineations there is a need for continuous refinement of the concepts, models, and methods as knowledge of the multiple relationships influencing forest-water relations grows. Research progress can especially be made at the interfaces between the various system delineations. These include: (1) estimates of evapotranspiration that can be scaled from tree-level sapflow, vegetation-level eddy-covariance and watershed-level water balances; (2) estimates of water storage and groundwater fluxes (including as it relates to soil type, soil depth and terrain features and may correlate with forest types); (3) estimates of atmospheric moisture recycling reconciling isotope-based and mass balance approaches; (4) estimates of both abiotic and biotic aspects of rainfall triggering; and (5) metrics that capture the effects of land cover change on flood (and drought) risk at various scales and in various contexts with confounding factors controlled.

2.3.2 Conclusions

A broader context that considers the interactions of climate, forests, water, and people is needed to assess current risks of not achieving the water quantity, quality and regularity of flow needed for the SDGs. At each of the seven system delineations of the climate-forest-water-people system, there are some globally valid conclusions, but also many statements that depend on the specific context:

1) At the watershed scale, four major determinants of ecosystem structure need to be considered – leaf area index, condition of the soil surface, infiltration patterns dependent on soil structure, and rooting depth – to understand hydrologic functions of forests and tree cover outside forests, and responses to ongoing and anticipated changes.
2) At the landscape scale, streamflow regulation through dams and reservoirs that tend not to occur evenly over larger watersheds and water abstractions can mask or strongly influence any positive effects forests in upper watersheds have on streamflow regimes. Unless one understands the physical basis of deviations from area-based scaling, it is risky to extrapolate beyond the scale range over which scaling rules were calibrated. This applies especially to peak flows, flooding risks and the degree of flood protection that intact natural forests and/or plantation forestry provides.
3) Tradeoffs between total water yield (expressed as fraction of precipitation) and the regularity of flow and water quality are to be expected for most contexts, as the rate of evapotranspiration in forests tends to be closer to the potential value than it is for most other vegetation, with the exception of wetlands and possibly irrigated agriculture.
4) Forest-derived atmospheric moisture mixes with ocean-derived moisture in spatially explicit patterns that have been well-documented on the basis of atmospheric measurements, and that lead to strong geographic variation in the percentage of precipitation derived from the long versus the short hydrologic cycle, as well as in the contribution a forest makes to short-cycle precipitation downwind. If confirmed by further scientific analysis, the idea that forests contribute to downwind rainfall could be of overriding importance for the prevention of water shortages, flood mitigation and design of forest restoration activities.
5) The hydrologic functioning of forests and landscapes with partial tree cover translates to a wide range of ‘ecosystem services’, with direct links between human benefits classified as provisioning, regulating, supporting and cultural services. The biophysical basis of the hydrologic functions and their variation in space and time may well be better understood than the social dimensions of associated rights, value concepts and regulations.

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Chapter 3
Determinants of the Forest-Water Relationship

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CONTENTS

3.1 Introduction

As outlined in Chapter 2, our analysis of forest-water relations addresses four important subsystems of a linked planetary social-ecological system: climate, forests, water and people. In this chapter, we consider how each of these subsystems is changing (trend) and what is causing the change ('determinant'). We discuss the critical determinants of change in forests as they relate to water quality and quantity. Chapter 4 then presents the impacts of these changes on water quality and quality.

3.1.1 What is a Determinant of Change?

In this chapter, interactions between forests and water are examined. The biophysical factors that significantly influence those interactions are termed determinants of change. They include, for example, gravity, soil pedology or climate change. Determinants of change occur over different scales both temporal and spatial. Some essential determinants of change for forest water use and yield may rarely occur but have a substantial impact; while others have a more frequent or constant impact on forest hydrology. Certain determinants of change operate on a very small scale, while other determinants of change may impact water resources across basins, regions or even globally. Each of these temporal and spatial scale determinants of change on forest water will be discussed separately.

Leaf area is an important measure for the water use of trees
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As described in more detail in Chapter 2, for almost 30 years, global studies have shown that trees evapotranspire (i.e., use) most of the precipitation that they receive (Running et al., 1989) through evaporation from their leaves via stomata (Whitehead, 1998). Stomata are the very small openings on the leaf surface through which carbon dioxide (CO_2) diffuses into the leaf, and water and oxygen (O_2) diffuse out of the leaf. Diffused atmospheric CO_2 is converted into carbohydrates while water vapour diffuses out of the leaf resulting in increased atmospheric relative humidity and atmospheric cooling (Li et al., 2015). Any factor that increases tree leaf area and the fraction of time stomata are open will thus create more sites for water loss, cooling and carbon gain. Conversely, any factor that decreases tree leaf area or leads to stomatal closure reduces the number of sites for transpiration and also reduces carbon gain (Tyree, 2003). Changes in leaf area thus comprise a standard measure by which changes in water use by the forest (or vegetation in general) may be gauged (Sun et al., 2011). As a control of forest water use, leaf area can serve as a proxy for assessing forest water use and yield (Caldwell et al., 2015). Leaf area may also have an impact on forest water quality through changes in soil erosion and stream turbidity by buffering forest soil from the direct impact of precipitation. The canopy absorbs much of the precipitation energy during the fall of a raindrop (Kang et al., 2008). The erosive force of the raindrop is reduced after the precipitation falls from the leaf onto the forest floor and therefore so is the erosive capacity of the water (Karamage et al., 2016). Forest canopies further protect water quality by reducing stream temperature and maintain higher levels of dissolved oxygen during warmer months (Moore et al., 2005). In addition to leaf area, other factors (e.g., previous land use history, slope, soil parent material) control forest-originated stream water quality (Neal, 2002; Clinton, 2011). However, as a single determinant of change, leaf area index (LAI) will be used as a measure of forest water use and yield throughout the chapter.

Although leaf area index is a useful vegetation cover indicator, there are other vegetation cover indices, including:

- Forest cover and deforestation rate (Achard et al., 2002; Mayaux et al., 2005). Forest cover rate is simple and easy to use, but it does not include any other types of vegetation. More importantly, from the forest hydrological perspective, it does not consider hydrological recovery due to forest regeneration after disturbance, which is a significant drawback for assessing forests and hydrology, particularly in large watersheds.
- Remote sense-based NDVI (normalised difference vegetation index, Matsushita et al., 2007) and equivalent clear-cut area percentage (Lin and Wei, 2008). Like LAI, NDVI is useful for vegetation changes at a relatively coarse level in vast regions of the globe, but it also suffers from 'saturation effects' of remote sensing spectrum data (Liu et al., 2012).
- Equivalent clear-cut area (ECA) is defined as the area that has been clear-cut or naturally-disturbed, with a reduction factor (ECA coefficient) to account for the hydrological recovery due to forest regeneration. It is an integrated indicator that combines all types of forest disturbances spatially and temporally and considers the vegetation and hydrological recovery following disturbance. ECA has been successfully used in forest hydrological research in British Columbia and elsewhere (Lin and Wei, 2008; Wei and Zhang, 2010; Lewis and Huggard, 2010). However, the demand for detailed data at the plot level makes it difficult to apply at the continental or global scale.

3.1.2 Three Dimensions of Determinants of Change

All determinants of change may be defined by the three dimensions of time, space and condition state. Time impacts a determinant in two ways: length of time and frequency (or how often a determinant of change is active, also known as 'return time'). As with time, there are two components for defining the spatial dimensions of determinants of change: resolution describes the primary scale at which a determinant operates, and ranges from the microbial to global scale; extent addresses the area over which a determinant of change typically occurs. Some forest determinants of change may be very impactful within a very limited spatial area. Although resolution describes the scale at which a determinant of change impacts on forest hydrology, the extent describes how common a particular determinant is across an area. A finer spatial resolution does not necessarily equate to a significant extent. For example, the cutting of trees for wooded figurine carving may have a significant local impact on forest hydrology, but the extent of such a practice might be insignificant if considered on a regional or global scale. Conversely, increasing atmospheric CO_2 would be a determinant with a global impact on forest growth and water yield. In this example, the CO_2 determinant acts at a microscopic spatial resolution (i.e., leaf stomata), but a vast extent (i.e., global).

The 'condition state' is the final dimension required to define a determinant of change as a function of relative impact on forest hydrology. Substantial changes in specific determinants may have little impact on forest hydrology and vice versa. A change in a determinant's condition state is, therefore, an indication of a determinant's stability and sensitivity. For example, methane (CH_4) is a much more efficient absorber of solar radiation compared to CO_2 (Lashof and Ahuja, 1990). Therefore, small increases in atmospheric CH_4 may have more impact on global warming and forest water use than significant increases in atmospheric CO_2. Each of these determinants of change will be discussed in more detail below.

3.2 Determinants of Change by Temporal Scale

3.2.1 Why Does Temporal Scale Matter?

Trees have a lifespan, from germination through seedling development, into sapling stage, eventually maturing, reproducing and ultimately dying as a result of natural or anthropogenic causes. The duration of this lifespan varies considerably, ranging from short-term fast-growing tree plantations, which may be clear-felled as quickly as six years after planting (Hinchee et al., 2011), through to ancient forest trees surviving for over a thousand years (Eifert, 2000). The lifespan of an individual tree is dependent on the environmental condition of the forest in which the tree is growing. A forest may take the form of a cohort of evenly-aged trees all established at approximately the same time and developing in unison, as is the case in a tree plantation, re-forested stand, or natural forest recovering after a catastrophic disturbance (e.g., wildfire, hurricane, tornado) (Lines et al., 2010). Conversely, some ecosystems may experience very infrequent, large scale, stand killing disturbances that can lead to multiple age class forests (Dale et al., 2001). The temporal scale under which these changes occur can impact the stability of the stream water quality and quantity as occasional small gaps in the forest cover have less impact on hydrology than do large areas of tree loss (Hansen et al., 2008).

3.2.2 Temporal Duration

The temporal duration of a determinant of change can be an essential contributor to forest hydrology. Short-term disturbances can have significant, long-term impacts on water yield (e.g., wildfire, Hallema et al., 2017). There is no general rule regarding temporal disturbance duration and impact, but an understanding of how each scale can impact forest hydrology is vital to effective water management. The next sub-section examines how short, medium and long-term temporal duration determinants of change influence forest water use and yield.

3.2.2.1 Short-Term / Event-Based (e.g., days or months)

Event-based determinants of change in forest ecosystems are of short duration (days or months) and may or may not have long-term consequences for water use and water yield. For example, floods, resulting from extreme

Flooding in forest after a heavy storm
Photo © iStock:VioNet

rainfall events, have short-term impacts of varying severity (Chen et al., 2015). However, if there is no substantial change in leaf area or soil condition of the affected stand, then the forest/water relationship should stabilise and return to a steady state in a relatively short time (Chen et al., 2015). On the other hand, an event-based determinant such as a wildfire – also a short-term event – may have long-term impacts even if only a small area of the forest is impacted (Hallema et al., 2017). The resultant decrease in leaf area will have immediate consequences through reduced evapotranspiration (water use) and lead to increased streamflow from the deforested watershed (dependent on antecedent soil moisture levels, recharge within the soil water profile and soil water infiltration capacity). The hydrological response following wildfire will impact both water quantity (e.g., average daily, seasonal and annual flows) and water quality through the potential for increased stream sedimentation (Richter et al., 1982). Nitrate inputs (Riggan et al., 1994) and water temperature can increase due to a loss of forest stream shading (Hitt, 2003). Recovery from these impacts will be dependent on the reestablishment of trees and restoration of leaf area and litter cover within the stand, which may take years before a hydrological response is restored to pre-fire conditions (Brown and Smith, 2000; Cuevas-González et al., 2009). Another important short-term determinant of change having long-term impacts on forest and water relations is logging (Gilmour and Gilmour, 1971; Storck et al., 1998).

3.2.2.2 Medium-Term (e.g., years)

Medium-term determinants of change that impact forest and water relations are numerous. They include disease/pest infestations (and associated leaf area changes linked to defoliation or mortality); changes in population density/demographics (Yin et al., 2017). Urbanisation can in turn increase the need for timber and other forest products with resultant changes in road and infrastructure development (Debel, et al., 2014). All of the above, result in changes in leaf area to a greater or lesser extent, with resultant impacts on streamflow. Some determinants of change result in maintaining or even increasing forest coverage, such as conservation and afforestation (Zhang et al., 2017b) efforts or a move towards alternative energy sources (e.g., photovoltaic, wind or biogas), leading to reduced deforestation and increased leaf area (Maiwada et al., 2014). However, there are exceptions to this, such as in Brazil, where a developing bioethanol industry led to forest clearing for sugar cane, with reduced forest leaf area (Lapola et al., 2010).

3.2.2.3 Long-Term (decades to centuries)

Long-term (i.e., decades to centuries) determinants of change having impacts on forest and water relations include elevated CO_2. While increases in tree water use efficiency due to elevated atmospheric CO_2 have been well established (Keenan et al., 2013), nutrient limitations may reduce the efficiency of tree water use (Oren et al., 2001). Additionally, increases in tree water use efficiency may not translate into increased stream flow as trees may increase leaf area and therefore total water use (and productivity) given the available water resource (Tian et al., 2010). Long-term changes in forest exposure to ground level ozone (O_3) can increase forest water use (and reduce stream flow) by causing leaf stomata to remain open and thus increase water diffusion from the leaf (Sun et al., 2012). Global climate change (i.e., long-term temperature and precipitation changes, changes in relative humidity, climate extremes) is one issue of significant concern regarding changes in forest water use and yield (WEF, 2017). Changes in precipitation and increasing air temperature will have significant impacts on global to local hydrology with or without forests being present (IPCC, 2014). The changes in the distribution, timing and amount of precipitation are still mostly unknown due to uncertainty regarding how quickly reductions in GHGs can be achieved (Kirtman et al., 2013). Globally, precipitation has increased during the 20th century as the atmosphere has warmed and the hydrologic system has accelerated (IPCC, 2014). At a smaller scale, current regional patterns of precipitation change may persist, intensify or dissipate in the years and decades to come (Kirtman et al., 2013). Likewise, global air temperature has increased by approximately 1°C since the 19th century, and all projections are for continued global warming with regional areas of minor warming (or even cooling) (IPCC, 2014). All warming will increase the forest potential evapotranspiration (PET) (Lu et al., 2005). The combination of increased precipitation and forest stream water flow, along with uncertainty regarding the frequency that the determinant of change will occur, the seasonality of change, and other factors (e.g., increased wildfire) make predictions of climate change impacts on future water yield difficult (IPCC, 2014).

Forest area increases (e.g., Indonesia, Hansen et al., 2013) could further stress areas receiving reduced precipitation as leaf area and evapotranspiration (ET) increase. Increasing water vapour associated with increasing ET could promote additional precipitation downwind,

Water towers project in Mau Forest, Kenya - Eucalyptus tree plantation
Photo © Patrick Shepherd/CIFOR

but the amount, location and timing are uncertain (Sheil, 2018). Establishment of a commercial forestry industry using introduced tree species (e.g., South Africa, Brazil), bush encroachment or infestations of invasive tree species (alien or indigenous) have all contributed to increased atmospheric water vapour (Stanturf et al., 2014). Further examples that impact forest water use and yield include changes in species composition (genetic changes/genus exchange) and associated water use/yield changes within commercial forestry or pollutant deposition (acid rain). Various governance and management measures – such as protecting water towers – all have an impact on leaf area (see Chapters 6 and 7).

3.2.3 Temporal Frequency

The temporal frequency of a determinant of change can be more impactful in altering water quantity and quality than duration. Infrequently triggered determinants of change can have significant long-term impacts on water yield. For example, major wildfires and hurricanes may only occur once every several decades in a particular forest, but a single event can result in substantial changes to forest structure. These structural changes can have significant implications for water yield and quality (Riggan et al., 1994; Brown and Smith, 2000; Cuevas-González et al., 2009; Hallema et al., 2017). Aside from the structural and functional forest changes, infrequent event-based determinants of change may alter forest management and risk perception. If an event has a small annual chance of occurrence, less preparation may be given to resistance and resilience measures before the event (Pilling, 2005). As climate variability increases, previously rare disturbances will become more common (IPCC, 2014); preparing for the extreme will become more critical moving forward.

3.3 Determinants of Change by Spatial Scale

No determinant of change will likely fit into only one spatial scale, but any given determinant will be more commonly observed at one scale over another. For example, drought can occur at either the basin or regional spatial scale, and across the short, medium and long-term temporal scales (Breshears et al., 2005; IPCC, 2014). As previously stated, tree leaf area will be the standard by which changes in forest water yield will be discussed for each determinant of change.

3.3.1 Why Does Spatial Scale Matter?

The understanding of the determinants of change of forest water quantity and quality by scale allows for the consideration of strategic and operational planning. Strategic planning provides guidelines for adapting or mitigating adverse impacts of large-scale or large spatial extent determinants of change (FAO, 2013). Once developed, these guidelines can provide extensive decision-supportive information across a range of forest conditions. Knowledge about the smaller scale or smaller spatial extent determinants of change is very useful for developing location-specific forest management practices. The details associated with operational planning are needed to put general knowledge regarding water resource management into practice (Cosgrove and Loucks, 2015). Consideration of spatial scale thus facilitates risk assessment and mitigation (primarily at a large scale) and optimisation of water production (primarily at a small scale, with potential for extrapolation).

3.3.2 Spatial Differentiation of Change

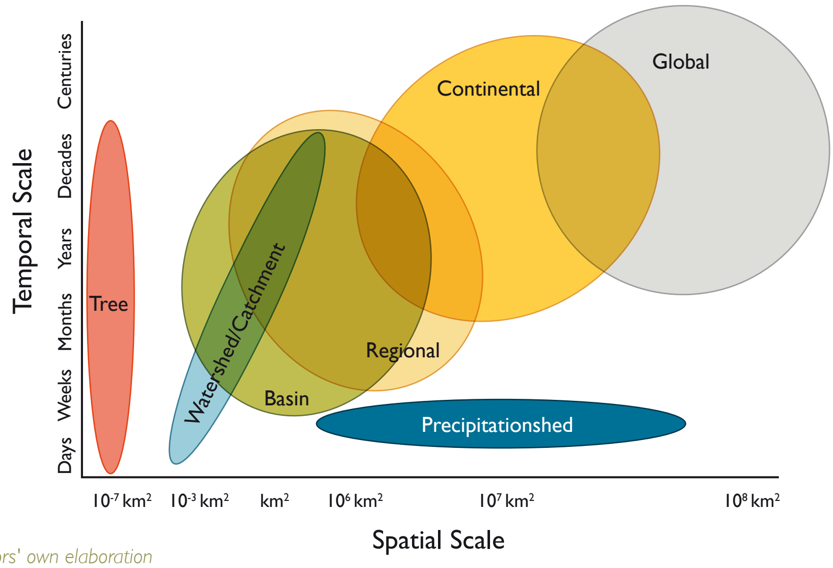
The differentiation between spatial scales can be used to examine forest hydrology (Figure 3.1). There are many temporal and spatial scales for defining and assessing forest ecosystems (Flipo et al., 2014; Figure 3.1). At its lowest common denominator, any determinant of the forest/water relationship could be considered at the level of a single tree, as every determinant fundamentally impacts one tree at a time. The tree is the scale at which individual changes and the resultant impacts on water resources can be multiplied and/or extended spatially. However, this extremely fine resolution has limited practical benefit and is too complicated to account for variations in responses across space (Lovell et al., 2002). Consequently, for strategic and operational planning purposes, risk assessments or management decision-making purposes, it is usually necessary to plan over a larger area (Schulze, 2000; Environment Agency, 2009). In hydrological terms, these might be referred to as Hydrological Response Units (HRUs). In this report, three spatial units are adopted which are common within much of the published literature, namely (in decreasing order of scale): continental scale; regional scale; and basin/watershed/catchment scale (Lovell et al., 2002). These delineations relate to ecological, geopolitical, meteorological, hydrological and operational separations that facilitate the understanding and prediction of the potential changes (impacts) on forest/water processes that may be wrought by respective determinants of change (Edwards et al., 2015).

3.3.2.1 Continental Scales and Global Scales

Our understanding of land use practices, land-atmosphere interactions (and the role of trees and forests, in particular), in the hydrologic cycle across land surfaces has increased over the past 80 years (Dooge, 2004; Suni et al., 2015). We expect larger scale change in land use practices to have an impact on the total amounts of atmospheric moisture that are circulated across terrestrial and continental surfaces. Sheil and Murdiyarso (2009), suggested that continuity of forest cover from upwind coasts helps to sustain transport of atmospheric moisture deep into continental interiors (e.g., the Amazon basin). However, it is challenging to estimate the amount of continuous forest cover necessary from upwind coasts to supplement atmospheric moisture in continental interiors. The continuous and ongoing anthropogenic transformation of the ecosystem, in particular, increasing leaf area, presumably contributes to significant changes in land-atmosphere

Primary temporal and spatial resolutions of ecosystem hydrologic scales as defined in this document. The relative differences are more important than the absolute range values

Figure 3.1



Source: Authors' own elaboration

interactions and thus to the cross-continental hydrologic cycle (Ellison et al., 2012).

Long-term and large-scale increases in forest evapotranspiration may increase precipitation and cross-continental transport of atmospheric moisture. The notion that forests produce massive amounts of atmospheric moisture, and more than most other land cover types, is not controversial. Decades of paired-catchment basin studies have focused on the role of forests in allocating precipitation over evapotranspiration and streamflow. Many studies have concluded that evapotranspiration in forests is close to the energy-determined potential rate with the remainder exported as streamflow (Bosch and Hewlett, 1982; Lu et al., 2003; Brown et al., 2005; Farley et al., 2005; Filoso et al., 2017). Most literature labels forest and cropland evapotranspiration as 'consumption' (Hoekstra and Mekonnen, 2012; Schyns et al., 2017), but from the atmospheric moisture perspective, trees, forests and other forms of vegetation are producers (Ellison et al., 2012).

Several researchers (Nobre et al., 2014; Keys et al., 2016; Keys et al., 2017; Wang-Erlandsson et al., 2017; Ellison et al., 2017) are exploring whether reductions in forest cover reduce continental scale precipitation. The concept assumes that terrestrial interiors are heavily dependent upon upwind land-atmosphere interactions and the production of atmospheric moisture through precipitation recycling (Bosilovich, 2002; van der Ent et al., 2010). If correct, the spatial organisation of a land use practice may have significant implications for downwind water availability (Ellison et al., 2017), and suggests that their impact increases as one moves further away from upwind coastal frontiers. The further from upwind coasts an individual catchment basin is located, the more it will depend on upwind terrestrial evapotranspiration and the smaller the impact of oceanic evaporation. Likewise, the more conversion from forest to urban settlement and other land uses occurs in upwind locations; the more downwind basins are likely affected by the change in land use practices. However, specifics of location relative to global circulation matters (van der Ent et al., 2010). Ecosystems outside of strong prevailing, moisture-laden, winds will have less precipitation compared to other areas where the influx of additional atmospheric moisture is more common (Figure 3.2).

3.3.2.2 Regional Scale

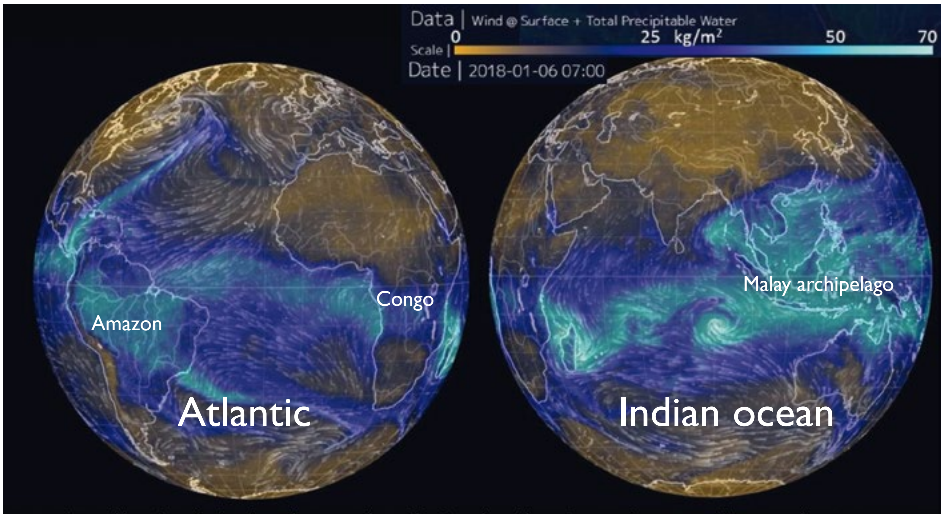
While determinants of change at continental and global scales are essential for understanding whole Earth processes, their role at a scale appropriate to forest management has not yet been adequately studied and quantified (Ellison et al., 2012; Sheil, 2018). For example, forest carbon sequestration slows global warming but competes with other forest environmental services such as efforts to increase forest water yield (Sun et al., 2011). Consequently, the regional resolution is considered the most extensive scale at which determinants of change of forest/water relationship can realistically still be managed. Some determinants of change that could be considered principally regional in scale include large-scale deforestation, afforestation or reforestation with resultant changes in forest/water interactions (Burt and Swank, 1992; Caldwell et al., 2012).

3.3.2.3 Basin and Watershed/Catchment Scale

Basins are smaller than regions, so there is a higher likelihood that an individual determinant of change could impact the entire spatial domain of a forested basin compared to one that is regional (Caldwell et al., 2012).

Wind speeds and total precipitable water is reduced over most forested land areas compared to oceans

Figure 3.2



Source: https://earth.nullschool.net/#2018/01/06/0000Z/wind/surface/level/overlay=total_precipitable_water/orthographic

However, similar to regions, there is a higher likelihood that a determinant of change will impact individual forests within a basin rather than the entire area. As the spatial area of a determinant of change decreases, so does the frequency and severity of impact on forest water yield and quality. For example, the probability of a cyclone occurring within a specific basin is less than the probability of a cyclone occurring within a region in which there are many basins. Likewise, the probability of a severe cyclone within a specific basin is less than the probability across all basins. Individual forest basin disturbance risk to water resources thus decreases from the region to the basin scale.

The watershed is the finest delineation of forest area that will be discussed as a determinant of forest water and represents the finest scale by which forest changes in water resources can be observed. Stands are the geographic scale below watersheds, but stands are often not delineated by water flow (Edwards et al., 2015). Instead, stands may present a particular forest or species type. A watershed may have one or many stands. The size of a watershed varies: as topography increases, the size of the watershed becomes smaller. Therefore, flat areas such as a coastal plain would likely have a more extensive watershed delineation than a mountainous forest. Management practices focus on either the watershed or stand scale, and determinants of change can be watershed specific. If water resources are managed at a watershed scale, then understanding evapotranspiration processes associated with the watershed is very important. For example, watershed management is essential in South Africa, where streamflow reductions (from high evapotranspiration rates) resulting from commercial tree plantations have been quantified per watershed (Gush et al., 2002), and commercial plantations are regulated/restricted according to their watershed-scale water resource impacts.

3.4 Determinants of Change by Condition State

Determinants that experience a large change in their condition state can often be very disruptive of water resources and are often the focus of forest management and restoration. For example, a trend toward more frequent and severe droughts can reduce forest water yield. Initial measures to eliminate water scarcity may include forest thinning (Douglass, 1983), while longer-term solutions may include tree species replacement (Burt and Swank, 1992). In total, there are three types of condition state: static, variable and trending. A fourth condition state termed 'new normal' (see Chapter 1) combines aspects of the previous three states. Each condition state will be defined separately.

3.4.1 Static Condition State

Static condition state determinants of change are essential for forest structure and function, but often (with notable exceptions) receive little attention. Such determinants of change may be considered permanently fixed (e.g., gravity), or, if they do experience change, such change will occur over very long timeframes, such as thousands of years (e.g., soil pedology). Changes in static condition state would likely have enormous implications for forest hydrology but the forces needed to change these determinants of change would also cause other significant changes (probably cataclysmic concerns).

3.4.2 Variable Condition State

The condition state of most determinants of change is variable. Historically, variable condition determinants of change of forest hydrology are centred on a mean value.

Moist forested landscape in Morne Trois Pitons National Park in Dominica
Photo © Andre Purret

However, the average is seldom observed. Instead, variability either increases or decreases the value centred on the mean. One of the primary concerns related to anthropogenic climate change is that variability is increasing, even if (for some parameters) the mean remains the same or similar. For example, annual precipitation may have remained constant over the past century in some regions, or without significant trend over the full measurement period, but seasonality or precipitation intensity has changed (or fluctuated between 'episodes'). More intense rain events followed by more prolonged periods of drought could produce the same amount of annual precipitation as more evenly distributed and less intense rains, but the impact on forest hydrology would be very different. For this reason, variability of determinants of change serves as a growing area of concern among forest managers.

3.4.3 Trending Condition State

A trending condition state is difficult to determine, as identification requires years of careful measurement and observation. Unlike a variable condition state, the mean of the trending condition state changes over time. If the factors impacting the determinant of change are well known and predictable, then changes in the trending condition state can also be predicted. However, if factors are not well known, then the rate of change, magnitude and even direction of the trending condition state cannot be anticipated. The changing condition state represents a fundamental shift in forest function. Forest managers and water users must, thus, also change their practices if forest water resources are to be sustainably managed under such changing conditions.

3.5 Atmospheric Determinants

Atmospheric determinants of change are the most important with regards to the extent, frequency and severity of forest water resources. In Chapters 2 and 5, climate appears as one of our mega determinants of change; clustered under 'global environmental change', which comprises one of the axes for the scenario analysis undertaken in Chapter 5 and referred to in Chapter 2. The interaction of precipitation and air temperature are the two most significant determinants of forest type and distribution. For these reasons, changes in atmospheric determinants of change have large impacts on forest hydrology (Novick et al., 2016). Figure 3.3 shows which forests globally are experiencing the highest rates of climate change. The spatial scale of atmospheric determinants of change range from global (e.g., carbon dioxide) to stand level (e.g., tornado and hail). Predominant airflow patterns in combination with topography determine climate (IPCC, 2014).

3.5.1 Climate

Some components of climate, including air temperature, precipitation, relative humidity, and wind speed, are determinants of change of forest water quantity and quality (Aber et al., 1995; Furniss et al., 2010). Also, there are many ways to examine temporal climatic change determinants of forest water including daily, monthly, annual, seasonal and event-based. Finally, there are different attributes of each component including, minimum, maximum, average and extreme. Even this non-exhaustive list would produce 80 (4 components x 5 temporal scales x 4 attributes) possible combinations, and there would be thousands of combinations of climate determinants if all were considered. That level of analysis is beyond the scope of this report. However, a few of the most frequently cited climate determinants are discussed.

3.5.1.1 Precipitation

Precipitation is the most robust single determinant of stream flow (Sun et al., 2011). Regardless of the change in other factors, reductions in precipitation will result in reduced streamflow. Dry forest types require a minimum of 300-400 mm of annual precipitation for full canopy cover (Ricklefs and Relyea, 2014); at this level, there will be no streamflow (Caldwell et al., 2012). The intensity and duration of precipitation also determine the timing of streamflow. Intensive or long duration rains can cause soil saturation and a significant proportion of fast flow (i.e., the percentage of rain that drains from the forest within 48 hours of a storm event). Conversely, frequent, gentle rains can allow most of the precipitation to be absorbed by the forest soil and slowly released over many months.

3.5.1.2 Air Temperature

Air temperature also serves as a significant determinant of forest water quality and quantity (Sun et al., 2011). Foliar cover provided by forest prevents direct solar radiation on streams (Dugdale et al., 2018) However, a lack of forest cover can significantly increase water temperature, leading to reduced water oxygen concentrations and water quality, especially under climate warming (Matthews, 2016). Additionally, as air temperature increases so does the vapour pressure gradient and tree demand for water (Zhang et al., 2015). Therefore, all other determinants of change being constant, increased air temperature reduces forest streamflow through increased tree evapotranspiration and stream water evaporation (Sun et al., 2011).

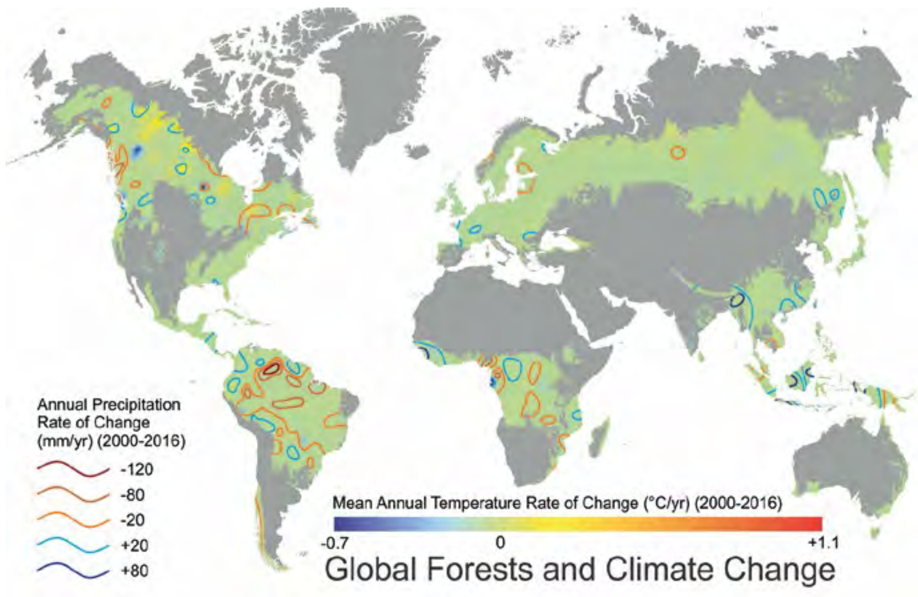
3.5.1.3 Wind Speed

Standard meteorological stations measure wind speed at ground level, with results relevant for evapotranspiration of short vegetation, but not for taller tree canopies. Wind speed depends on the height in the atmosphere and the surface roughness of the vegetation (Irwin, 1979), as well as season and location on the globe (van der Ent et al., 2010). A recent 'stilling' or reduction of measured wind speed data over the northern hemisphere could be

Figure 3.3

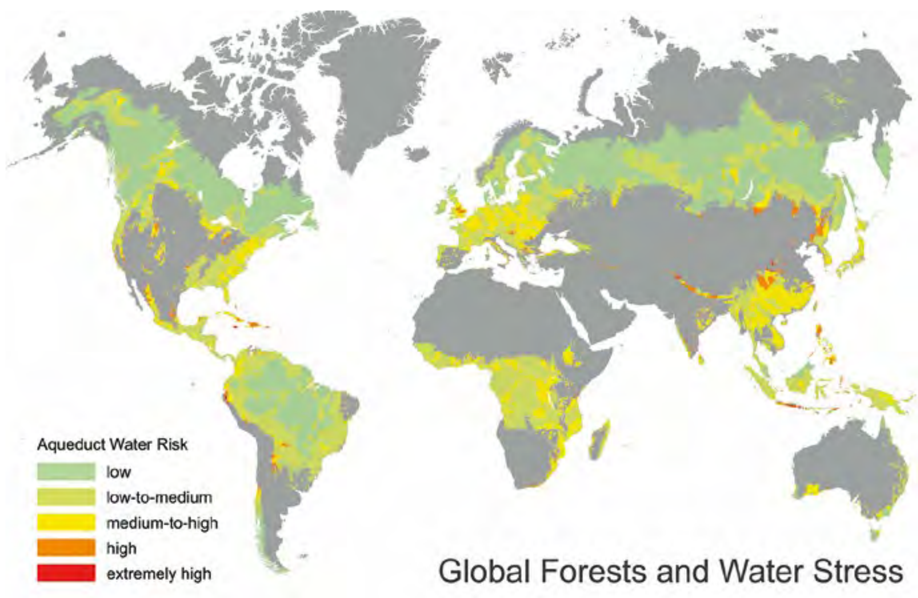
Changes in precipitation (Figure 3.3 (a)), do not necessarily correlate with water shortages (Figure 3.3 (b)) due to water demand and absolute differences in precipitation and forest water use

a)



Sources: Hansen/UMD/Google/USGS/NASA; CPC Global Daily Temperature; CPC Global Unified Gauge-Based Analysis of Daily Precipitation

b)



Sources: Hansen/UMD/Google/USGS/NASA; WRI Aqueduct Global Maps 2.1 Data

Box 3.1

Which forests are experiencing the most substantial rates of climate change over time?

Forests provide ecosystem services by protecting water supplies. Over 80% of global forest cover is in areas of low or low-to-medium water security risk mapped by the World Resources Institute (Gassert et al., 2014); less than 4% of global forest cover is in areas of high or extremely high water risk primarily because forests tend to occur in areas of low human population density.

Also, forests provide climate services by removing carbon from the atmosphere and, in tropical regions, mitigating warming through evaporative cooling (Bonan, 2008). At the same time, carbon removal through forest growth requires water, affecting the partitioning of water supplies and altering hydrologic cycles and atmospheric water exchanges at regional and continental scales (Ceci, 2013).

The complex forest-water-climate interactions occur in the contexts of both deforestation and climate change; alterations in forest cover or climate can lead to deviations from, or intensification of, the feedbacks between forest, climate and water. Changes in temperature and precipitation can directly alter the long-term composition of forests (Rustad et al., 2012). Changes in forest composition can lead to increases in the frequency, duration and intensity of natural disturbances – such as drought, fire and pest outbreaks – that can increase tree mortality and alter the structure of forests (Dale et al., 2001; Allen et al., 2010). Boreal forests in Canada and Russia have faced the most significant stress of increased temperature since 2000, while tropical forests in the Amazon basin have faced the most significant stress of decreased precipitation since 2000 (Boisvenue and Running, 2006).

partly attributed to an increase of vegetation roughness (Vautard et al., 2010), with trees outside forest increasing roughness more than closed forest stands. Increasing tree roughness and decreasing windspeed would reduce forest transpiration (Fisher et al., 2005) and therefore increase forest stream flow.

3.5.2 Atmospheric Chemistry

3.5.2.1 Air Pollution

Air pollution can increase or decrease forest water yield. Nitrogen deposition from the burning of fossil fuels can fertilise forest and increase leaf area (Pregitzer et al., 2008; Quinn et al., 2010), leading to reduced water yield. However, too much nitrogen can lead to a condition of nitrogen saturation (as observed in the northeastern US and parts of Europe) (Aber et al., 1989). The progression of nitrogen saturation leads to forest mortality, reduced leaf area and increased streamflow (Lovett and Goodale, 2011; McNulty et al., 2014). Nitrogen deposition can also be converted into highly leachable nitrate through soil nitrification, and negatively impact water quality (Aber et al., 1989). Additionally, ozone formation in the troposphere occurs when nitrogen oxides (NO_x), carbon monoxide (CO) and volatile organic compounds (VOCs) react in the presence of sunlight (Krupa and Manning, 1988). Ozone can damage forest leaf stomata that regulate carbon dioxide intake and water loss, making trees less water use efficient (McLaughlin et al., 2007). Reduction in forest water efficiency translates into increased forest water use and decreased streamflow. Black carbon (i.e., soot), can also impact hydrology by changing the albedo and therefore melting of glacial water (Box 3.3).

Box 3.3

Forest fires and their impacts on glaciers, snow cover and hydrology

Forest fires, both natural and human-induced, are frequent globally and their incidence and spread are increasingly affected by climate extremes (Kale et al., 2017). Studies from the Tibetan Plateau and the Indian Himalayas suggest that up to 40% of all black carbon emissions come from biomass burning, including forest fires (Zhi et al., 2011). When light absorbing impurities like black carbon settle on white snow or glacier surface, they reduce snow albedo and enhance glacier and snowmelt, and thus affect the overall hydrological regime. A study in the Indian Himalayas found that black carbon aerosols could potentially heat up the Himalayan atmosphere by 0.04-0.06 K/day and that could result in a 5-20% reduction in snow cover over a decade (Bali et al., 2016). The deposition of black carbon on snow increases surface temperature by approximately 1°C, which has a more significant impact on snow melt than CO_2-induced atmospheric temperature rise (Qian et al., 2015), reducing snow and ice cover in the region (Barnett et al., 2005)

3.6 Anthropogenic Drivers of Forest Change

Temporal and spatial drivers of change of forest water can each be further divided into 'direct' (or 'proximate') and 'indirect' (or 'ultimate', 'root' or 'underlying' causes) drivers (Lambin et al., 2003). Proximate causes of land-use change constitute human activities or immediate actions that originate from intended land use and directly affect land cover (Ojima et al., 1994) and typically involve a physical action on land cover. Indirect causes are fundamental forces that underpin the more proximate causes of land-cover change and operate more diffusely or at a different scale (e.g., national or global economy), often by altering one or more proximate causes (Lambin et al., 2003).

3.6.1 Forest Transitions and Land Use Change

Deforestation, forest degradation, plantation development and increases of trees outside forest have altered the distribution of trees and mixture of forests (Ordonez et al., 2014). Such trends have been linked to anthropogenic factors in various parts of the world (Lambin et al., 2001; Turner et al., 2007; Haberl et al., 2007; Zomer et al.

2016), with strong time dependence of patterns in many instances. Forest-transition theory describes and explains non-linear changes in tree cover (i.e., the loss of natural forests followed at some point by an increase in planted and managed trees) as a country develops (Mather and Needle, 1998; Dewi et al., 2017). Forest transitions to other cover classes occur at continental scale, but also at a finer-grained basin scale (Dewi et al., 2017). Rather than a one-way human land cover change relationship, humans and natural systems interact to create changes in forest cover (Liu et al., 2007). For example, Meyfroidt et al. (2014) and Robbins et al., (2015) linked tropical tree crop expansion and commodity agroforests.

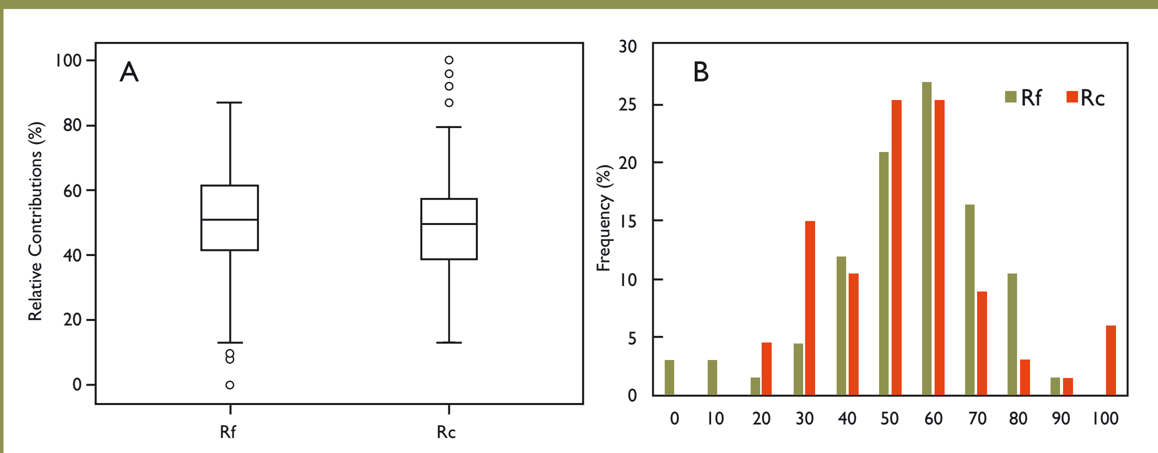
Determinants of change of land use (and land cover) change have increasingly become global (Lambin and Meyfroidt, 2011), with commodity markets connecting patterns of change across many locations. Protecting forests in one location without changing demand for products that caused the forest change is likely to deflect rather than reduce forest conversion (Meyfroidt et al., 2013; Dewi et al., 2013; Minang and van Noordwijk, 2013). Intensive debate on the scale at which agricultural intensification slows down or speeds up deforestation has focussed on the drivers that can be used for leverage in the coupled and globally connected social-ecological systems (Byerlee et al., 2014; Carrasco et al., 2017; Law et al., 2017).

Box 3.2 | Figure 3.4

Interactions of climate and land cover changes as determinants of hydrological change

Forest cover change and climate variability are commonly viewed as two significant determinants of change for hydrological variations in forest-dominant watersheds. The influences from climate must be either removed by possible methods such as paired watershed experimental studies (PWE) or explicitly accounted for to assess the effects of forest cover change on hydrology, (Wei and Zhang, 2010; Zhang and Wei, 2012; Liu et al., 2014). Any research in large watersheds (>1000 km^2) has to explicitly include climate into the analysis so that the relative effects of forest cover change on hydrology can be quantified because the PWE approach is not suitable for large watersheds. Thus, the relative contributions of forest cover and climate variability to hydrology are often assessed in large watershed studies, while these are not ordinarily available in PWE studies. Also, there are essential feedbacks between those two determinants of change. For example, forest changes can also affect hydrology through their impacts on climate alteration due to their cooling effects and atmospheric recycling (Ellison et al., 2012). These feedbacks may not affect the assessment of the above-mentioned relative contributions as they are already reflected in climate data collected.

Numerous studies on separating the relative contributions of forest cover change and climatic variability to annual water yields have been conducted in the past few decades (Zhang et al., 2017a). A recent review based on 168 studies from large watersheds (i.e., > 1,000 km^2) around the globe shows that forest cover and climate variability play a co-equal role in annual water yield variations (Figure 3.4, Li et al., 2017). Also, the effects of forest cover change and climate variability on annual water yield variations can be additive or offsetting due to their directional influences. The effects of deforestation (more) or reforestation (less) annual water yield (AWY) variations are mono-directional, and their effects are cumulative over a specific period. In contrast, the effects of climate variability on AWY variations tend to fluctuate or be multi-directional and consequently may lead to possible cancellations or additions over the deforestation or reforestation period (Aber et al., 1995). Thus, the difference in the impact directions may make the hydrological effects of forest cover change more pronounced. Both the magnitude and direction of two determinants of change must be considered for assessing and managing hydrological changes.



Source: Cited from Li et al., 2017

(A) Boxplot of the relative contributions of forest cover change (Rf) and climate variability (Rc) to large scale (i.e., > 1,000 km^2) annual watershed water yield variations; and

(B) Histogram of relative contributions of forest cover and climate variability to annual water yield variations. The averaged Rf and Rc are 50.1 ± 18.9% and 49.1 ± 19.5% respectively

Changes in forest cover, especially conversion to agriculture, can have significant impacts on water quality (Scanlon et al., 2007).

3.6.2 Demographic Change and Urbanisation

Two processes of demographic change can drive tree cover (or forest) transitions (see previous section) in opposite directions, with hydrological consequences as discussed in the next chapter: increasing human population and urbanisation. An increase in human population density has historically always been associated with a reduction of forest cover (Köthke et al., 2013). A decrease in rural population, started primarily since the industrial revolution in the 19th century, may present an opportunity for forest regeneration in some areas (e.g., Agnoletti. 2014; Box 3.4). At the same time, urbanisation is associated with a change in lifestyles which can exert more pressure on the forest for production (DeFries et al., 2010). In a pantropical data set, Dewi et al. (2017) found the two patterns combined, with a tree cover of 20-30% for the highest population densities in (peri)urban sub-watersheds, a 'more people, less forest' part of the curve and a 'more people, more trees' phase. The nuance depends on the operational forest definitions used (van Noordwijk and Minang, 2009; Chazdon et al., 2016). A recent change in the eastern states of the US suggests a new period of forest cover loss, after earlier re-expansion (Drummond and Loveland, 2010), linked to shifting lifestyles.

While drivers of land abandonment are more or less well understood, impacts on forest regeneration and biodiversity are only partially understood and are very context specific – in some places, farmland abandonment leads to regrowth of natural forests and subsequent increases in biodiversity, in other instances, invasive species take over. Given this dearth of literature, more studies are needed that directly link land abandonment and regrowth of natural vegetation with local water resources.

3.6.3 Conflicts

In addition to the drivers of change associated with demographic variability, as discussed above, wars both displace populations and physically disturb forest ecosystems (Orians and Pfeiffer, 1970; Nackoney et al., 2014; Daskin and Pringle, 2018). Historically, war and conflict often place considerable pressure on the need for natural resources, including water and wood products (Homer-Dixon, 1994; McNeely, 2003). Displaced populations may seek forests for shelter, refuge and fuel (Daskin and Pringle, 2018). When such actions increase the need for fuelwood and timber, this causes a reduction in tree leaf area, which in turn may increase river flows and water yield. However, under conditions of conflict, forest use is generally (although not always) sporadic and uncontrolled, and proper forest practices that protect water quality are unlikely to be followed (DeWeerdt, 2008). Poor forest management is likely to bring about increased sedimentation and a reduction in water quality, regardless of timber loss (Fergusson et al., 2014).

The widespread use of defoliants in forested areas during war significantly reduces forest cover (Westing, 1971;

Box 3.4

Land Abandonment

Abandonment of agricultural land and subsequent natural re-growth of vegetation is a common phenomenon across all mountain regions of the world. Most of these documented cases are from the Alps (Gellrich and Zimmermann, 2007) and other mountain ranges in Europe (MacDonald et al., 2000; Sitzia et al., 2010; Tarolli et al., 2014; Regos et al., 2015; Latocha et al., 2016) where the process of land abandonment started at least a century ago in some places. In Europe, primary drivers of land abandonment were rural to urban migration and related de-population in mountain areas; lack of profitability of mountain agriculture; forest fires and in some cases, unsustainable land management practices that led to soil erosion and associated hazards. In recent years, several provisions of the Common Agricultural Policy have also led to the abandonment of farmland, especially in the mountains and such marginal areas (Regos et al., 2015; Latocha et al., 2016). In Japan, land abandonment in mountain areas started in the 1950s and was driven by macroeconomic shifts and demographic transition (Palmer, 1988) with a positive impact on biodiversity and forest regeneration (Osawa et al., 2016; Katayama et al., 2015). In the Hindu Kush Himalayas, abandonment of agricultural land through outmigration is a relatively recent phenomenon, starting in the 1990s driven by macroeconomic factors, including opening up of earlier insular economies. In Nepal and China, outmigration and labour shortages in mountain villages are the main cause of land abandonment (Jaquet et al., 2015; Zhang et al., 2016). In the Indian Himalayas, new ecosystems preservation plans that ban traditional animal husbandry practices are known to have led to the abandonment of pastures (Nautiyal and Kaechele, 2007).

Abandoned land in previously terraced landscapes was found to be particularly prone to gully erosion and landslides (Tarolli et al., 2014), while in other instances, land abandonment and increase in the area of forests and grasslands led to a decrease in soil erosion (Latocha et al., 2016). Sitzia et al. (2010) looked at 53 case studies of land abandonment and subsequent natural forest recovery and found that the results were mixed. Overall, there was a decrease "in semi-natural habitats such as meadows or pastures due to natural reforestation" and therefore, an overall loss of landscape-level diversity (Sitzia et al., 2010). None of the studies looked at the relationship between secondary forest regeneration and local level water resources.

Meyfroidt and Lambin, 2008). While this may lead to an increase in stream flow and water yield, long-term legacy on land and water pollutants may remain for some years or decades. However, there have also been instances where situations of conflict and social unrest have brought about a reduction in the use and overuse of forest areas, thus allowing forests to regenerate (Davalos, 2001; Alvarez, 2003).

3.7 Outstanding Gaps and Research Priorities

Forests are complex ecosystems even when forest structure and function are relatively stable (i.e. in steady state). Understanding the interaction of determinants of forest water quality and quantity is therefore challenging. Assessment of current and prediction of future forest water resources becomes even more challenging under the ever-changing conditions of the 'new normal'. Climate serves as the most critical determinant of forest water availability. Improved models and support for the use of short, medium and long-term weather and climate forecasting would provide the single most significant benefit for improved forest water forecasting. Beyond climate, improvement in demographic, economic and technology forecasts would also help support improved forest water management. Management options are further expanded in Chapters 6 and 7 to follow.

3.8 Conclusions

Determinants of change in the climate-forest-water-people system vary over space and time. Additionally, the relative interaction between determinants is also changing making it difficult to predict forest water flows. Under a changing climate, these factors are changing more than ever, sometimes in unanticipated ways.

The magnitude of each determinant of change influences the degree of hydrologic impact on an ecosystem. Not all determinants of change have similar impacts on forest water use and flow regime. By better understanding which determinants of change have the most significant impact on forest function, estimates of water supply can be improved while minimising assessment costs.

No single factor determines forest resources, but changes in climate are the most important determinant of hydrology, regardless of the ecosystem. In addition to differences in precipitation and other factors such as forest leaf area, air temperature and management practices can also, secondarily, impact forest water use and yield. Under a changing climate, the variability of precipitation is increasing, so more extreme ranges in water flow in all terrestrial systems should be expected.

The appropriate temporal and spatial scale for assessing and managing forest water use and yield depend on the question being asked. Questions related to regional water availability across average or extreme environmental conditions require long-term predictions of climate variability and understanding of inter-basin atmospheric and terrestrial water flow (Ellison et al., 2017). Our ability to understand the complexities and interactions of large-scale forest hydrology is not complete due to limitations in large-scale measurement, monitoring and prediction (Sun et al., 2011). Conversely, the determinants of change of local water availability have been studied for over 80 years and are well understood (Douglass, 1983).

Historical paradigms regarding seasonal weather patterns, rainfall amounts and intensity are becoming outdated, as new patterns, limited patterns or no pattern emerge under the 'new normal' (Thornton et al., 2014). This continually evolving context makes it very difficult to establish a baseline by which determinants of change of forest water quantity can be evaluated (Carpenter and Brock, 2006); and yet, the establishment of such a baseline is critical.

The ability to forecast how adaptive management can contribute to the stabilisation of forest water quality and quantity has never been more important, nor more challenging. Fortunately, while non-antecedent conditions are contributing to this notion of a 'new normal', the principles of ecosystem science still apply.

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Chapter 4
Forest Landscape Hydrology in a 'New Normal' Era of Climate and Land Use Change

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CONTENTS

4.1 Introduction

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

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

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

4.2.1 Hydrological Consequences of Natural Forest Disturbance

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

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

4.2.1.1 Invasive Weeds, Insects and Pathogens

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

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

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

Mountain Pine Beetle (MPB) infestation in Baskerville, British Columbia, Canada,
Photo © John L. Innes

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

Box 4.1

Wildfire and 2018 Southern California mudflows

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

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

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

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

4.2.1.2 Wildfire

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

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

In the immediate aftermath of wildfires, the burnt soil is bare and dark, and highly susceptible to erosion,

La Tuna wildfire in Los Angeles, CA in 2017
Photo © iStock: Jorge Villalba

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

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

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

4.2.1.3 Ice Storms

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

4.2.2 Hydrological Effects of Human-Driven Forest Changes

4.2.2.1 Silviculture

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

Diversity of species

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

Age

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

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

Thinning

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

Harvesting

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

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

4.2.2.2 Plantations

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

Pine plantation in South Africa
Photo © Mark Gush

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

Monoculture plantations also have less biodiversity compared to natural stands (Brockerhoff et al., 2008) which can increase the risk of episodic insect and disease outbreaks, or fire that can threaten the health of the entire stand (Mitchell et al., 1983; McNulty et al., 2014). While complete stand or catchment mortality can significantly increase stream flows, tree mortality may also decrease water quality (Hibbert, 1965; Swank et al., 2001).

4.2.2.3. Forestation

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

Box 4.2

Reforestation/afforestation programmes in China

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

4.2.2.4 Agroforestry

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

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

4.2.2.5 Urban and Peri-Urban Forestry

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

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

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

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

4.3.1. Climate Change and Future Forest Hydrology

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

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

At the same time, forests are important for sequestering atmospheric CO_2. Many studies have shown that old-growth forests and old trees can continue to accumulate carbon in vegetation and in soils, while harvesting old-growth forests results in net carbon release (Harmon et al., 1990; Zhou et al., 2006; Luyssaert et al., 2008; Stephenson et al., 2014).

Box 4.3

Potential impact of rising treelines on water resources

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

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

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

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

Box 4.4

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

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

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

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

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

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

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

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

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

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

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

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

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

4.3.3. Changes in Natural Disturbance Regimes and Future Forest Hydrology

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

4.3.4. Forest Succession and Future Forest Hydrology

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

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

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

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

4.4 Data Needs and Knowledge Gaps

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

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

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

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

Box 4.5

Trees as history books reflecting past climate variability

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

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

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

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

Box 4.6

Stable water isotopes and sources of precipitation

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

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

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

4.5 Conclusions

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

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

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

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

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

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Chapter 5
Current and Future Perspectives on Forest-Water Goods and Services

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CONTENTS

5.1 Introduction

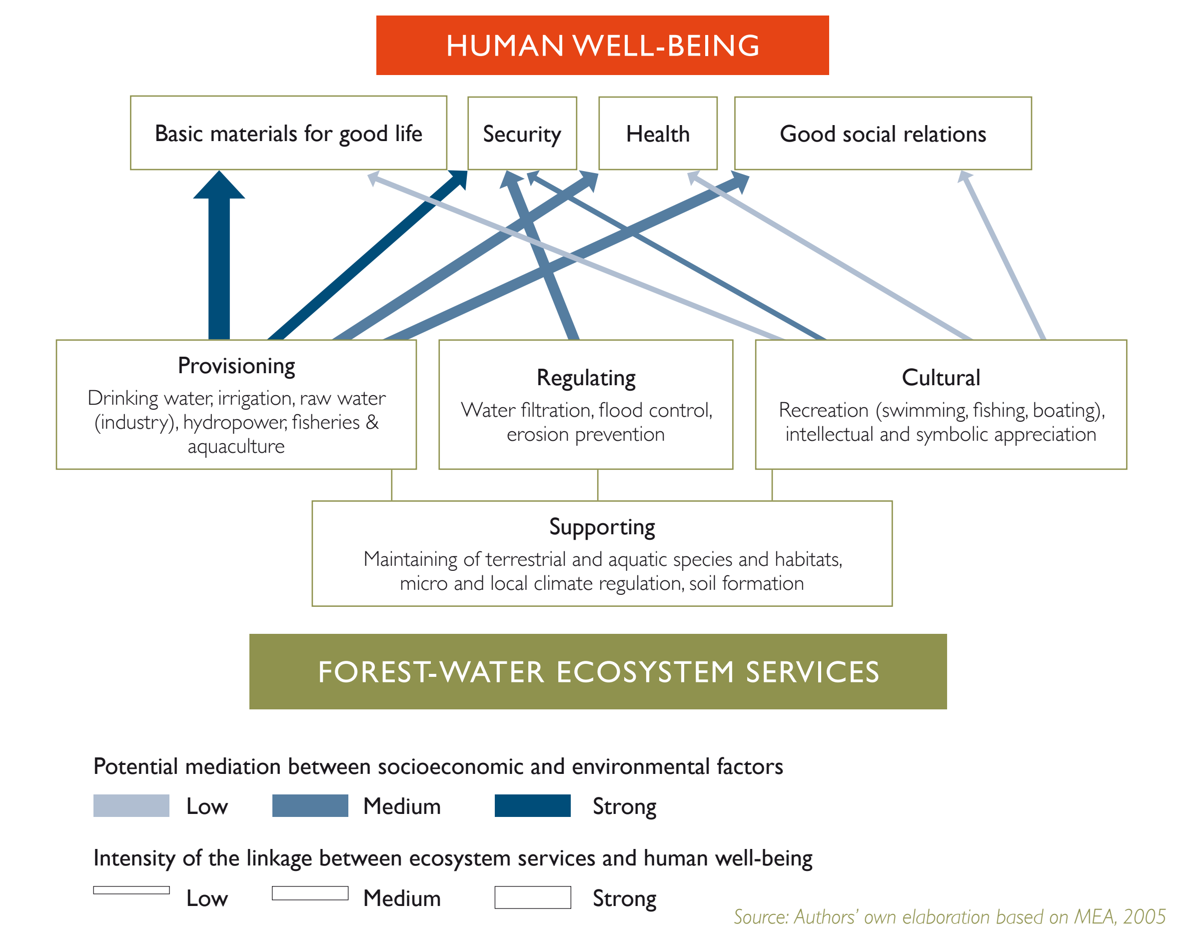
This chapter utilises the ecosystem services framework to understand the consequences of change in forest ecosystem functions and water-related implications. Using a scenario analysis, the chapter explores the likely changes in attributes of forest-water systems (and associated services) that will translate to exogenous impacts, and their consequences in the future. The narrative provides a foundation for the analysis of management options and policy responses that will be discussed in Chapters 6 and 7. Ultimately, these responses are likely to affect the drivers of change and thus highlight the interconnectedness of coupled forest-water systems.

5.2 Conceptualising Forest-Water Relationships in Terms of Ecosystem Services

5.2.1 Origins and Evolution

The dependence of human life and well-being on finite natural resources has long been acknowledged (Malthus, 1888; Meadows et al., 1972), and different conceptualisations of human-nature relationships have emerged over time (Raymond et al., 2013). The term ecosystem services (ES) represents one such conceptualisation (Martin-Ortega et al., 2015). The ES concept was coined in the 1960s primarily to raise awareness among policymakers about the implications of biodiversity loss and environmental degradation by emphasising the benefits that nature freely provides to society (Gómez-Baggethun et al., 2010). The "Tragedy of the Commons" framed by Hardin (1968) triggered the debate about open access to natural resources. The natural processes of environmental degradation which have impacts on social-ecological systems, therefore, generate social change (Eckholm, 1975). Literature on ecosystem services grew exponentially from 1997 onwards, when Daily (1997) defined the term as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life" and Costanza et al. (1997) estimated the total economic value of the planet's ecosystem services at USD 33 trillion/year. Despite criticisms on methodological grounds (e.g., El Serafy, 1998), further publications consolidated this body of research (e.g., De Groot et al., 2002), until it firmly entered the policy arena when the UN Secretary-General Kofi Annan called for a global assessment of the world's ecosystem services

Linkages between forest-water ecosystem services and human well-being Figure 5.1



Source: Authors' own elaboration based on MEA, 2005

(Millennium Ecosystem Assessment report, MEA, 2003, 2005). Ecosystem services were then defined as "the benefits that people obtain from ecosystems" (MEA, 2003) and the dominant classification scheme of ecosystem services was established. In this scheme, ES were divided as supporting (services required for the production of other ecosystem services), provisioning (products that can be directly obtained from the ecosystem), regulating (benefits that can be indirectly obtained from the regulation of ecosystem processes), or cultural services (non-material benefits that people obtain from ecosystems), which all directly or indirectly contribute to human well-being.

Further to the MEA, there has been a proliferation of ES frameworks and applications, including a multitude of novel research directions and refined definitions and classification of the ES domain (Ojea et al., 2012). A major difference between the ES frameworks is how intermediate ecosystem processes are treated. Some frameworks only include final services consumed or valued directly by humans (e.g., Hein et al., 2006; Haines-Young and Potschin, 2013), while others also include intermediate environmental processes that contribute indirectly to human-welfare (e.g., Boyd and Banzhaf, 2007). As ecosystems depend strongly on the water cycle, the complex inter-linkages between ecosystems and the water cycle make the classification of water-related services as supporting, regulating, or provisioning particularly complex (Ojea et al., 2012). For example, water flows can be regarded as supporting services for maintaining terrestrial and aquatic species and habitats, or micro and local climate regulation; or they can be regarded as regulating services for aquaculture production or as provisioning services for agriculture or drinking water supply; in a way that simultaneously affects different components of human well-being (Figure 5.1). Box 5.1 illustrates how an ecosystem services-based approach would apply to the understanding of water-related forest ecosystem services.

Box 5.1

An ecosystem services-based approach to the understanding of water-related forest ecosystem services:

- … recognises that structural changes to forests can influence several watershed processes (e.g., erosion rates, sediment load, water chemistry, peak flow levels, total flow, base flow, or groundwater recharge) – in different ways and that, in turn, these changes result in different kinds of impact on human well-being (e.g., increased costs of water purification, increased fertilisation of floodplain lands, decreased reservoir capacity due to siltation, flood damage, changes in agriculture), (Lele, 2009).
- … requires the understanding of the biophysical processes that determine the way forest cover, forest structure, soil-vegetation dynamics, etc. affect the amount and quality of freshwater to the extent that it impacts on human well-being (through use or non-use) by the beneficiaries.
- … combines knowledge of the service delivery processes that are based on natural sciences (e.g., plant physiology, ecology, hydrology) with information from social sciences (e.g., economics, psychology, political science) and (local) stakeholder knowledge (e.g., farmers, domestic water users, floodplain residents, hydropower companies, regulators) that jointly help to understand, for example, where benefits arise in relation to where ecosystem change takes place.
- … requires at least some degree of quantification of changes in the final services delivered (e.g., increase of the flow of water associated with forest cover) coupled with a qualitative interpretation of the implications for human well-being, or the valuation of associated benefits through, for example, willingness to pay for increased water availability, so that these benefits can be incorporated into decision making (for example, on afforestation or the creation of protected forest areas) (Martin-Ortega et al., 2015).

5.2.2 Valuing Ecosystem Services

Values and associated processes of valuation have been of interest to researchers and philosophers since ancient times, and the term has been ascribed a multiplicity of meanings (Schulz et al., 2017). On the one hand, values can be conceptualised as abstract guiding principles (fundamental or held values) that may inform preferences and decision-making. Examples are security, achievement, or self-direction. On the other hand, values can be understood as measurements of a certain quality or of importance (i.e. assigned values). The ecosystem service paradigm and environmental economics, which are rooted in neoclassical economics, are examples of strategies to describe assigned values. Human beings are seen as rational actors that aim to satisfy their substitutable preferences and maximise their personal utility through their choices (Pearce and Turner, 1990; Dietz et al., 2005). Value is then defined as "the change in human well-being arising from the provision of [an environmental] good or service" (Bateman et al., 2002). These welfare changes can be compared through conducting monetary valuation studies that estimate relative values and people's willingness to pay to achieve an environmental change, such as improved water quality from forest conservation practices.

While being the most widespread conceptualisation of (environmental) value, the neoclassical definition has also attracted a lot of criticism for epistemological and moral reasons (Gómez-Baggethun et al., 2010; Norgaard, 2010). Other critiques reflect misgivings about related concepts such as markets, capitalism, commodification and/or neoliberalism derived from monetisation (Brockington and Duffy, 2010) and have been rejected by those defending more eco-centric conceptualisations of human-nature relationships (Martinez-Alier, 2002; Schulz et al., 2017).

Another recent branch of literature on values focusses on shared and social values, which Kenter et al. (2015) present as those values that an individual holds on behalf of a community or group of which they are a part. More recently, relational values, which are understood as ethical and moral principles that guide 'good' human-nature relationships and may differ across cultures (Chan et al., 2016), have emerged as a new conceptualisation more coherent with the pluralistic views promoted by new approaches to ecosystem services (Muradian, 2017), as advocated by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES).

Here, when we refer to the value of water services provided by forest, we refer to assigned values (i.e. the values that people derive from forest and the water services they provide) but recognise that these are underpinned by fundamental values that critically shape world views and the relationship between people and forests.

5.2.3 Criticisms and New Conceptualisations: Nature's Contributions to Humans

The ecosystem services concept has arguably inspired novel avenues for environmental research, enhanced communication, debates and cooperation between scientists from diverse disciplines, policymakers, conservationists and practitioners. Beyond the MEA, the global TEEB initiative (The Economics of Ecosystem Services and Biodiversity; Kumar, 2010), and related national ecosystem assessments (e.g., the UK NEA, 2011) are testimony of the concept's wide-ranging appeal.

Inevitably, the popularisation of the ES approach has also led to the emergence of new debates and criticisms. While not questioning it, some see gaps in the practical implementation of the conceptual advances made (Nahlik et al., 2012), such as deficient monitoring, and some see the risk of the concept of ecosystem services losing its original (or any) meaning as pre-existing environmental management approaches are simply relabelled. More critically, many point at the risk of oversimplification of ecological, economic and political processes (Norgaard, 2010). Ecological economists are critical of the neoclassical conceptualisation of environmental values and argue that some values cannot be measured with a single measurement unit such as money (Martinez-Alier et al., 1998). Ethical concerns have also been raised about the potential misuse of the ecosystem services concept for the commodification of nature where artificial markets are created for public environmental goods (Kosoy and Corbera, 2010; Peterson et al., 2010), as well as about the marginalisation and crowding-out of non-anthropocentric (often non-Western/utilitarian) ethical frameworks for nature conservation (Raymond et al., 2013).

The criticism is extended to the consideration of equality in the distribution of ecosystem services, and also to the interpretation of benefits in different socio-cultural contexts. The power, gender and labour relationships which mediate access and capability to manage ecosystem services need to be highlighted in an ecosystem service approach. The degree to which any individual benefits from ecosystem services thus depends on a complex range of mechanisms of access including natural and social capitals, both traditional as well as emerging and evolving rights to natural resources (Ribot and Peluso, 2003). Also, the ecosystem services approach often does not sufficiently take traditional ecological knowledge into account (Xu and Grumbine, 2014a,b). Some argue that a practical alternative to the problems of conventional valuation would be to make use of a multi-criteria approach, enabling the inclusion of a wider range of issues (Fontana et al., 2013). Others propose a less anthropocentric conceptualisation of values that encompasses other worldviews (such as those of indigenous communities). For example, in Australia, indigenous people believe that all of the environment is interlinked and they are part of that interlinkage (Altman and Branchut, 2008), having been created with forests and water, and all within them at the beginning of time, remaining as custodians of nature (Flannery, 1994; Skuthorpe and Sveiby, 2006). Even today they engage in living cultural landscapes and waterscapes, where water and forests are central to culture, spirituality and identity (Bark et al., 2011). A major challenge remains as to how such deep understandings can be incorporated into modern policy and institutional arrangements relating to the management of forests and water resources.

Uluru, also known as Ayers Rock, is a large sandstone rock formation in central Australia, Uluru-Kata Tjuta National Park. Uluru, including its rock caves, ancient paintings, and surrounding springs and waterholes, is sacred to the Pitjantjatjara Anangu, the Aboriginal people of the area

Photo © Pixabay: Wallula

To address some of these challenges and criticisms, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has recently developed a new framework that seeks to integrate different knowledge systems regarding human-nature interactions, including indigenous and local perspectives alongside western scientific models (Pascual et al., 2017). It consists of six interlinked elements constituting a social-ecological system that operates at various scales in time and space (Díaz et al., 2015). These are nature; nature's benefits to people; anthropogenic assets; institutions and governance systems and other indirect drivers of change; direct drivers of change; and good quality of life. While shifting the focus towards relational values, a good quality of life and cultural specificities, the IPBES framework essentially maintains the original anthropocentric perspective, but emphasises a less utilitarian philosophy and pluralistic values.

In this assessment we adopt the ecosystem services conceptualisation as the currently dominant way of expressing the relationship between humans and nature, so that any existing evidence can be integrated more effectively here. However, we acknowledge other visions and the fact that this is an evolving paradigm.

5.3 Consequences of Change

5.3.1 Consequences for the Delivery of Water-Related Forest Ecosystem Services

Forest ecosystems provide timber, energy, food, fodder and other goods while maintaining diverse ecosystem services and functions (see Section 5.2) that are relevant for human well-being.

Forests are spatially heterogeneous areas in which the trade-offs and synergies in the provision of goods and services are governed by complex interactions of environmental factors and processes, with social and economic forces operating at different spatial scales. The spatial pattern of land uses, management intensity, land use changes, climatic conditions, the resilience of forest ecosystems, and natural and anthropogenic stressors, such as droughts, extreme climatic events, wildfires, atmospheric pollution, or invasive species are the main factors affecting the provision of forest goods, and environmental services and functions (Lawler et al., 2014; Millar and Stephenson, 2015; Newbold et al., 2016; Castello and Macedo, 2016; Seidl et al., 2016). Changes in coupled forest-water systems can thus have significant impacts on biota, and ultimately on human-well-being. Yet the consequences of natural- and anthropogenic-driven changes on forest-water systems depend on their scale and intensity (see Chapter 3). Where forest and water are concerned, changes in land use and management would mainly affect water quality and quantity.

The most significant contribution forests make to water for all living beings is in maintaining its quality (FAO, 2008). The role of forests in filtering sediments and other pollutants from water before it reaches the stream has increased the interest in conserving forest and restoring riparian vegetation to protect water quality (Sweeney and Newbold, 2014). Brogna et al. (2017) found that forest cover has a positive effect on water quality, using a long-term and spatially distributed monitoring data set that covered more than half of Belgium's territory. The contribution of forest in protecting water quality can have economic implications. For example, Fiquepron et al. (2013) and Vincent et al. (2016) found that a higher forest cover can be translated into lower drinking water supply costs in France and Malaysia, respectively.

On the contrary, a decline in forest cover may have a negative effect on water quality. Large scale deforestation can affect the physicochemical properties of downstream waters (Dessie and Bredemeier, 2013). In studying the impacts of deforestation in Amazonia, Langerwisch et al. (2016) found that deforestation will decrease riverine particulate and dissolved organic carbon amount by up to 90% and the discharge of organic carbon to the ocean will be reduced by about 40% under a severe deforestation and climate change scenario. This will have local and regional consequences on the carbon balance and habitat characteristics in the Amazon Basin itself as well as in the adjacent Atlantic Ocean. Changes in forest structure can also affect water temperature, with the removal of riparian canopy, generally leading to increased energy loading to the stream and higher stream temperatures (Bladon et al., 2016). Likewise, forest management can affect water quality. Higher management intensities can raise concentrations of suspended sediment and nutrients following silvicultural operations (e.g., Eriksson et al., 2011; Laudon et al., 2011; Siemion et al., 2011). The effects of harvesting will be higher when timber and biomass extraction bares the soil surface, thereby increasing the erosion risk (FAO, 2008).

Where timber and water are concerned, researchers tend frequently to think in terms of trade-offs between timber and water provision. Those trade-offs may go beyond timber and water, as trade-off between carbon sequestration and water provision services have been also reported in areas with water scarcity problems (Chisholm, 2010; Ovando et al., 2017). Similarly, forests also provide non-timber products, which should be considered in evaluating options (see Box 5.2). Trade-offs can also involve erosion regulation and water yields, whereas afforestation can provide relevant erosion reduction benefits while reducing water yield (Dymond et al., 2012). Large scale forest plantations can control sediment and nutrient loads and protect water quality (depending on their management), but this can lead to conflicts between beneficiaries of upstream plantations and downstream water users, where there is a demand for irrigation water (e.g., Nordblom et al., 2012).

Changes in coupled forest-water systems have significant impacts on biota. For instance, Ricketts et al. (2004) found that forest-based pollinators increased coffee yields by 20% and improved coffee quality within 1 km of forests in Costa Rica. Similarly, coupled forest-water systems support a large variety of birds, with a high percentage being dependent on forest habitats. Among the benefits that birds provide are pollination, insect pest control, seed dispersal and nutrient cycling (Wenny et al., 2011), but they also add substantial value to the economy through tourism, with bird-watching being one of the faster growing subsectors of ecotourism (Callaghan et al., 2018). The direct dependence of aquatic biodiversity on water quality and quantity render it specifically vulnerable to change. One of the ways for citizens to support informed policy development and decision-making is through applying local and traditional knowledge to local solutions and feeding those through to policy and management domains (see Box 5.3).

Box 5.2

The importance of non-timber products for millions of forest dwellers and indigenous people

Poor people throughout many parts of the world depend heavily on the direct use of natural capital for their livelihoods. The multi-functional use of natural capital can be exemplified by Non-Timber Forest Products (NTFPs) which are essential inputs to forest people. In the burgeoning discussion around ecosystem services, the safety net that NTFPs provide for subsistence households must not be forgotten (Vira et al., 2015).

In the development of more effective macroeconomic policies for forest and water management, it is vital that we recognise that contrary to conventional production theory (Simon and Khan, 1984; Beckerman, 1995), other forms of capital (finance, infrastructure) cannot infinitely be substituted for natural capital (Daly and Cobb, 1989; Daly, 1999). As such, in the development of forest and water policy, we must recognise the multitude of disparate values held by different social groups in different ecological contexts. In an example provided from a study of Amerindian communities in Guyana (Sullivan, 2002), the ways people use forest resources have been identified and quantified. The monetary value of this use is calculated using an income accounting approach, and these use-values demonstrate the importance of non-timber forest products and services to people in these subsistence communities.

Findings of this study show that depending on location, the value of forest inputs amounts to between 33% and 63% of productive values in these communities. Whereas benefits derived from the harvesting of timber products are limited by forest production, the livelihood support that can be generated from the use of non-timber forest products and services are not limited in the same way. This has significant implications for sustainability, and it is important that if the quality of this income stream is to be preserved for future generations, action must be taken to ensure that this dimension of forest resources is not depleted by the decisions and actions of either local residents or global policymakers.

Shoebill (*Balaeniceps rex*) in Mabamba swamp in Uganda. The shoebill attracts many bird-watchers

Photo © Marius Claassen

Forest-water services are about hydrological dynamics, and the socially-constructed relationships that underpin humans and ecosystems; for example, the rules, infrastructure and access to benefits and substitutes. The ability for humans to receive ecosystem services varies from place to place and from time to time, with the social-ecological system or political economy often playing a role in shaping the distribution of benefits of ecosystem services (Ostrom, 2009). For instance, frequent extreme weather and floods cause more loss of human life and property due to poor land use practices, poor planning and urbanisation on flood-prone areas and the poor often suffer the most due to lack of protection (Agrawal et al., 2008). Since these impacts are more severe at downstream locations, such communities are often willing to pay more to upstream communities for forest/water services. Therefore, the impacts of change on ecosystem services are varied in space and time and case-specific due to the ecological relationship between forest and water, as well as the socioeconomic relationship between humans and nature. Water is perceived to be useful only when people have access to it or have the ability to benefit from it (Brauman et al., 2007), however, the indirect benefits of water to people go beyond this narrow view.

A limitation of the literature in this area is that many recent forest hydro-economic studies apply engineering-oriented bottom-up models based on a combination of empirical relationships and theoretical control factors, such as behavioural responses or hydro-ecological processes (e.g., Garcia-Prats et al., 2016; Susaeta et al., 2016; 2017; Ovando et al., in press). These have been used to estimate the response of a production function (and associated revenues and/or costs) to specific interventions (Brouwer and Hofkes, 2008). Such relatively simple input-output production relationships examined at the individual plot level provide only a static view of the supply or demand of water ecosystem services (Harou et al., 2009). Only a few studies explicitly analyse the effect of forest interventions on water supply and demand, water prices and their welfare effect on economic sectors competing for water use (e.g., Nordblom et al., 2012; Garcia-Prats et al., 2016).

In arid and semiarid areas, the substitution of natural grasslands, shrubs and croplands with fast growing plantations, is often associated with decreases in streamflows and groundwater recharge, leading to potential conflicts between upstream plantations and downstream water users (e.g., Nordblom et al., 2012). At the local scale, increased tree cover can also be associated with reduced streamflow (Chapter 3). However, depending on the baseline conditions, the opposite may be true. Reforestation of degraded agricultural lands with heavily compacted soils may raise dry season stream flows by increasing infiltration rates and soil water holding capacity (Garcia-Chevesich et al., 2017). The role of forests in filtering sediments and other pollutants from water before it reaches the stream, has increased the interest in using forest to increase water quality, thus reducing drinking water supply costs (Abildtrup et al., 2013; Vincent et al., 2016). An analysis of the relative magnitude of ecosystem services provided by forests can inform decisions on species selection, forest density and management options in relation to the regulation of water flow and quality under changing climatic conditions (Box 5.4).

5.3.2 Consequences of Change at Different Scales

Forest-water related services are dynamic and complex across scale and time. Each service has attributes of quantity, quality, location and timing of flow (Brauman et al., 2007). Scale is still considered to be the unresolved problem in the relationship between forests and water (Malmer et al., 2010). Knowledge about hydrologic services from forests are often based at catchment scale (Bruijnzeel, 2004). Such forest and water relationships are varied or dynamic in terms of the time scale. While in some cases (particularly in tropical areas), forest cover may be restored in a relatively short time, many forest-related water services (such as reducing sediment and enhanced water quality) may take much longer to recover (Malmer et al., 2010).

The impacts of changes in coupled forest-water systems are sensitive to boundaries in biophysical systems as well as jurisdictional boundaries. Natural boundaries include climatic zones (defined by altitude, rainfall, temperature, ocean proximity, etc.), surface water catchments (defined by topography) and groundwater basins (defined by hydrology, geology and topography). Anthropogenic alterations to these boundaries include inter-basin water transfers and changes in land cover. Eco-regions, discussed in Chapter 2 represent the combined impact of the above boundaries. Societal boundaries include differences in cultures and practices, economic conditions and jurisdictional boundaries. Since social values and practices can vary within and between communities, the impacts of change in forest and water systems and related services will also vary between different societal contexts, whereas economic circumstances influence development priorities and options. Jurisdictional boundaries, which can be sub-national (e.g., districts), national (sovereign countries) or regional (e.g., regional economic commissions) are associated with different policy contexts, economic conditions and societal perspectives. The determinants of change (Chapter 3), changes to the forest/water system (Chapter 4) and response options (Chapters 6 and 7) are all sensitive to natural and societal boundaries. Temporal scale is another type of boundary, since the time scales of political, policy, economic and social process are not always aligned with the time frames of environmental (forest/water) impacts and responses.

The delivery of water-related forest ecosystem services is scale-dependent in terms of biophysical processes (Chapters 3 and 4), but also in terms of governance processes. More recent literature on forest-water interactions and dynamics suggests that the boundaries for the

Box 5.3

Toad's eye views and water quality

Science and technology as concepts, howsoever varied might be the understanding behind them, underlie any discussion of water and forest management. It is when deconstructing these concepts with questions like whose science? What kind of technology? Which methods? What inherent capacity for maintenance? etc. that the hegemony of particular technologies and scientific approaches expose themselves. Research methods can be classified as those primarily aimed at external learning ('extractive science'), those primarily supporting local learning, and methods that explicitly aim for both (Mehmood-Ul-Hassan et al., 2017).

Methods that match local concerns over water and are yet understandable by scientists and forest management officials can play a substantial role in negotiations on clarifying sources of pollution and changes in flow regime linked to local land uses (Tomich et al., 2004; van Noordwijk et al., 2016). Biological water quality monitoring methods (Rahayu et al., 2013a, b) have been used to support local stakeholders in forest mosaic landscapes where land use patterns are contested.

In water management, there is growing awareness that there are traditional technologies, some of them centuries old, which are perfectly well-suited to the hydro-ecology of the region in which they are found in as well as to the availability of local raw material and skills for their use. On the other hand, modern ferro-cement technologies or piped water systems may be efficient, but beyond the capacity of the local people or their raw material resource base to maintain or restore if damaged by a disaster. Brushwood dams still account for over two-thirds of actual irrigation in the Himalayas and the technology they deploy is dependent on the collective capacity of the local irrigation community. These dams are built at the start of the dry season to divert water to the fields and are washed away during the monsoon, only to be re-built in the next dry season. When they are replaced by modern ferro-cement dams, the modern technologies are very efficient as long as they operate, but when a major flood occurs that damages or washes them away, they are abandoned as unrestored relics and people revert back to their traditional technology (Gyawali, 2004). A similar story is unfolding in the semi-arid zones of western India with traditional water harvesting technologies as modern technologies fail to deliver (Agarwal and Narain, 1997)

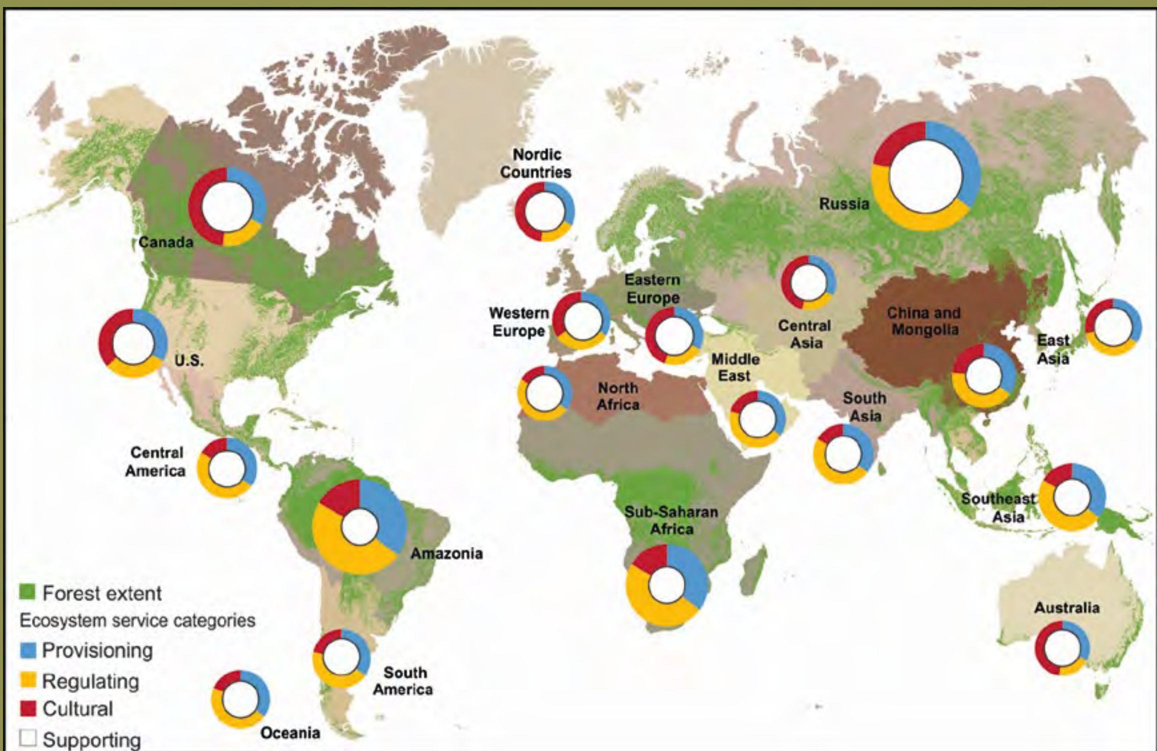
If technology is defined as science which has commercial implications, the disjuncture between modern and traditional technologies is explained by the power and bias of the most powerful market players. The decision-making process remains dominated by investments backed by modern technologies involving cement, steel, petroleum-based plastics and other such powerful artefacts, while local communities are alienated and disempowered, and their traditional technologies marginalised. Ethnographic and anthropological studies of science and technology have tried to distinguish between 'toad's eye' or civic science and 'eagle eye' or modern western science to reveal this contrast in approaches to knowledge.

Box 5.4 Figure 5.2

An attempt to quantify the value of forest-based ecosystem services

An approach to quantify service domains is to divide the world into regions and calculate the amount of tropical and temperate/boreal forest for each region. These estimates are then grouped into provisioning, regulating and cultural ecosystem services (ES), summed to estimate the total valuation per category, and then multiplied by the area of tropical and temperate/boreal forest in each region to produce an estimate of the total ecosystem service category value for each region. While the value of supporting services can also be calculated in this way, these services are deemed as intermediate services that are implicitly embedded in the final value of regulating, provisioning and cultural services (Hein et al., 2006).

The portfolio and relative magnitude of the different ES types, based on the above calculations and ES values from De Groot et al. (2012) are presented in Figure 5.2, with the supporting services being superimposed on the other three types of services to denote their intermediate nature. The figure illustrates the variation in different ecosystem services provided by forests in different regions of the world. Based on this approach, northern regions generally provide more cultural services than southern regions, which may reflect differences in sustainable forest management between these two regions (Fisher et al., 2009; Chan et al., 2012). These north/south differences may be attributed to the differing ecosystem service values for tropical and temperate/boreal forests, with tropical forests providing a larger amount of provisioning and regulating ecosystem services and temperate/boreal forests providing a larger amount of cultural and supporting ecosystem services. The loss or degradation of a forest in one region may have different consequences for water security than in another region. Forest provisioning and regulating ecosystem services – in particular – have important implications for water security. To mitigate the potential loss of specific ecosystem services, a portfolio of functions should be maintained on the landscape. A portfolio of function approach ensures that there exist forests in a specific region that provide low, medium and high levels of each of the categories of ecosystem services. This portfolio approach illustrates that the ecosystem can better buffer against change and ensure that forests provide a suite of ecosystem services to the landscape's inhabitants.



Source: Authors' own elaboration based on calculations from De Groot et al., 2012

Figure 5.2. Map illustrating the portfolio and relative magnitude of ecosystem services provided by forests. The relative magnitude of ecosystem services for each region is illustrated by the relative size of the circles, whereas the relative size of each segment represents the value of each service

political governance of these interactions need to be greatly extended in space (Ellison et al., 2017). Most cross-regional and international water management frameworks for negotiation consider only the catchment (watershed) boundaries and include actors situated at least in part within the catchments. However, in order to adequately address water availability concerns and impacts with respect to forests and land use change, it is necessary to redesign these frameworks such that they can actually take into account the principal contributions from a much broader concept of hydrologic space. The 'precipitationshed' approach is currently perhaps the best example of this concept. Since land use practices, both upwind and within the given catchment, ultimately influence the total amount of water that is either consumed locally or redistributed onto other downwind basins (Dirmeyer et al., 2009), it is of explicit interest to be able to harness these factors in the service of the larger framework of forest and water management strategies. Thus, both local, regional and larger forest and water management strategies and institutional systems need to find meaningful ways of not only incorporating and involving up- and downstream interests, but also of involving up- and downwind communities in the larger overall forest and water management framework.

5.3.3 Consequences for Human Well-Being

As described earlier, changes in forest status can lead to significant changes in hydrological functions, which in turn translate into changes in the provision of ecosystem services (Lele, 2009). Besides the biophysical repercussions, these changes have direct and indirect socio-economic consequences well beyond forests' boundaries (Gregory, 2006; Wang-Erlandsson et al., 2017). For example, in China controversial resettlement schemes have been a key instrument for the government to address poverty and environmental degradation in the past two decades, with up to 6 million of the 120 million internally-displaced people qualifying as environmental migrants (Myers, 2002). These schemes have been associated with both ecological and social consequences (Fan et al., 2015) with some cases showing that resettlement promotes ecosystem recovery by removing human pressures (notably from grazing livestock) and improving access to infrastructure, education, and health care. In other cases, however, there are also negative social and ecological impacts in newly resettled areas, including a disruption of the coupled social-ecological system among resettled communities.

One conceptualisation which helps to understand the well-being implications of changes in hydrological functions is derived from neoclassical economics, based on the measurement of welfare changes in monetary units (Pearce and Turner, 1990; Bateman et al., 2011). Under this conceptualisation, changes in well-being are directly linked to the value that humans attach to ecosystem services, measured through the monetary trade-offs that individuals are willing to undergo to secure the service. As an illustration, Box 5.5 provides current evidence of the monetary value of water ecosystem services delivered by forests, focusing on two forest systems of global relevance: tropical forests in Central and South America and mangrove forests in South East Asia. This evidence provides some basis for the general understanding of the welfare benefits that forest conservation provides in relation to water ecosystem services and, as a corollary, of the welfare loss associated with the decline in the state of ecosystems. However, it should be noted that this literature is very heterogeneous in purpose and approaches, providing a very fragmented view of the value of forest water services (Lele, 2009; Ojea et al., 2012).

Despite their limitations, a growing number of studies offer some insights into the economic implications of forest conservation and management for the provision of water ecosystem services. For example, some econometric

Box 5.5

Evidence of the monetary value of water services provided by forests

Tropical forests in South and Central America

Ojea and Martin-Ortega (2015) undertook a meta-analysis of 25 primary valuation studies of water services of tropical forests in Central and South America, which served to identify some factors that systematically influence forest values.

The review of this literature reveals how the definition and classification of water ecosystem services is highly inconsistent (Ojea et al., 2012), which can generate problems such as double counting (Fisher et al., 2009).

The meta-analysis shows that the relationship between the value and type of service is complex and is mediated by the type of beneficiary. Extractive water supply services (involving mostly agricultural and human water consumption) have, in general, relatively high values; although the value of flow-regulating services (in-stream water supply) when the beneficiary is an industrial user (i.e. mostly used for hydropower production) is significantly higher than when used for agricultural and human consumption (but not as high as extractive water supply generally).

There is much less consolidated evidence on the monetary value of damage mitigation and water cultural related benefits in comparison to provisioning services.

Mangroves in South East Asia

Brander et al. (2012) undertook a meta-analysis of 41 studies assessing the value of mangrove ecosystem services around the world to project values for South East Asia.

The range of ecosystem services represented in the collected studies includes provisioning services (fish, fuelwood, materials) and regulating services (coastal protection, flood prevention, water quality). Similarly to the case of tropical forest, the value of cultural ecosystem services is under-represented in the literature.

The type of service significantly affects its value, with water quality and fisheries having a positive and significant effect on this value. Mangrove value is also influenced by the existence of other mangrove forests in the area. This seems to indicate that fragmentation of mangroves and their surroundings (e.g., by road infrastructure) has a negative effect on the value of mangrove.

The median mangrove value in the sample is USD 239 per hectare per year (2007 prices).

Brander et al. 2012 also forecast the value change associated with a projected 2000 – 2050 scenario and estimate an annual value of lost ecosystem services from mangroves in South East Asia, which amounts approximately to USD 2.16 billion in 2050 (2007 prices).

studies provide empirical evidence on the positive effect of forest cover (thus conservation) in reducing the costs of drinking water supply (Ernst et al., 2004; Abildtrup et al., 2013; Fiquepron et al., 2013; Vincent et al., 2016). Forest conservation is also expected to generate positive income and welfare effects by controlling dam sedimentation and increasing hydropower generation (Arias et al., 2011), or by reducing flooding damages for downstream farmers (Kramer et al., 1997). A number of other studies suggest trade-offs between the production of timber and water ecosystem services, implying an opportunity cost (revenues foregone) for landholders, in cases where no compensation schemes are in place (Eriksson et al., 2011; Kucuker and Baskent, 2015; Simonit et al., 2015; Garcia-Prats et al., 2016). A small number of studies are also starting to look at the economic implications of the trade-offs between water quantity and quality associated with forest practices, trying to integrate economic values associated with water ecosystem services into decision support systems (Keles and Baskent, 2011; Kucuker and Baskent, 2015; Mulligan et al., 2015; Garcia-Prats et al., 2016). These studies reveal that internalising water values leads to different optimal forest management decisions than are based on the single maximisation of timber net benefits, which highlights the need for advancing water ecosystem services valuation and integration into decision-making processes.

Further inspection of the literature also demonstrates how most of the existing evidence on the value of (water) ecosystem services provided by forests focuses on limited types of ecosystem services: predominantly provisioning and some of the regulating services; other regulating services and especially cultural ecosystem services, are limited in the monetary valuation literature. There are studies on the recreational value of forests (Chiabai et al., 2011 reviewed some of them) but the link to water ecosystem services is often unspecified which is consistent with the fact that less tangible services are harder to measure and hence tend to be ignored. This represents a critical limitation since the tendency to avoid services that are difficult to measure creates a bias in resultant policy choices. Moreover, it is increasingly argued that water-based ecosystem services provide benefits that go beyond what can be monetised. Even in the realm of human health alone, poor management of water and forest systems has been shown to result in increases in water borne diseases, increasing risk to humans from flooding and coastal inundation, and reducing food security (Corvalan et al., 2005).

5.3.4 Social Consequences and Distributional Considerations

The consequences of changes in forest-based water ecosystem services are not evenly distributed. While aggregate availability of water, as well as its quality, might be reflected in catchment level or system-wide analyses, the spatial distribution of this water, as well as the social and political context within which people have access to or are able to benefit from such services can be highly unequal (Mollinga, 2008; Loftus, 2015). When considering the forest-water relationship in terms of impacts, it is thus important to be mindful of questions of distributional equity, fairness and justice (Sikor et al., 2014), the political economy of water allocation which underpins who gets

Box 5.6

Water allocation in the Murray Darling Basin, Australia: conflicts and consequences

The Murray Darling basin, located in southeastern Australia, covers over 1 million km^2 (14% of Australia's landmass) and contains over 30,000 wetlands. It provides water storage and multiple other ecosystem services across its vast floodplain. Throughout human history the river has played an important role, providing sustenance for Aboriginals and supporting national economic development (Bark et al., 2012). Today some 65% of all irrigated land within Australia lies within its boundaries, accounting for over 95% of total irrigation water withdrawals (Tan et al., 2012).

By the 1990s, threats to the basin's capacity to deliver water for multiple uses across its complex social and political boundaries led to recognition that the river had to be managed holistically, in a more integrated way. As a result, the Murray Darling Basin Authority (MDBA, 2012) was formed. Although at first these institutional developments were positively received at the state and local government levels, many farmers depending on irrigation continued to resist water allocation change. Exacerbated by extreme drought conditions in the first decade of the new millennium, and over-allocation of abstraction licences (Crossman and Overton, 2011), the federal government decided to implement a more stringent plan for water allocation, based on the concept of Sustainable Diversion Limits (SDLs). Key ecosystem functions such as sediment loads, nutrients, carbon exchange, habitat maintenance and connectivity (Falkner et al., 2009) were to be monitored to ensure compliance with the objectives of the plan, and a framework for transferable water rights through a water market were identified (Qureshi et al., 2009), based on water 'buy-backs' being strictly made on the basis of 'willing seller, willing buyer' agreements.

Unexpectedly for the government, when this plan was first released for public consultation in 2011, (MDBA, 2011) panic and chaos ensued among farming communities across the basin. This resulted in demands for a series of additional special studies (Bark et al., 2012; Mooney and Tan, 2012; Jiang and Grafton, 2012), and much debate about what the appropriate level for the SDLs should be (McKay, 2011; MDBA, 2012), causing long delays in the implementation of the plan. Even by early 2018, the debate over ecological water allocations remains unresolved. Making the situation worse, evidence has recently come to light of illegal water transfers being made in many areas of the basin, with national newspaper headlines drawing attention to 'water theft'. Meanwhile, many areas of the floodplain forests of iconic Red Gums continue to decline, conflicts between land and water users remain, and many forest and former wetland areas are consumed by the increasing number of bushfires occurring every year.

how much water, when and where, and to recognise that this is likely not to be equally available to all stakeholders across a landscape. Importantly, environmental flow requirements should be considered in allocation processes. A reflection on access and distribution is provided in Box 5.6.

5.4 Scenario Analysis: Consequences of Change in the Future

Anticipating changes under the 'new normal' is necessary in order to establish likely changes to the forest-water-climate-people system and to determine appropriate measures based on desired objectives. A scenario analysis helps to project into the future.

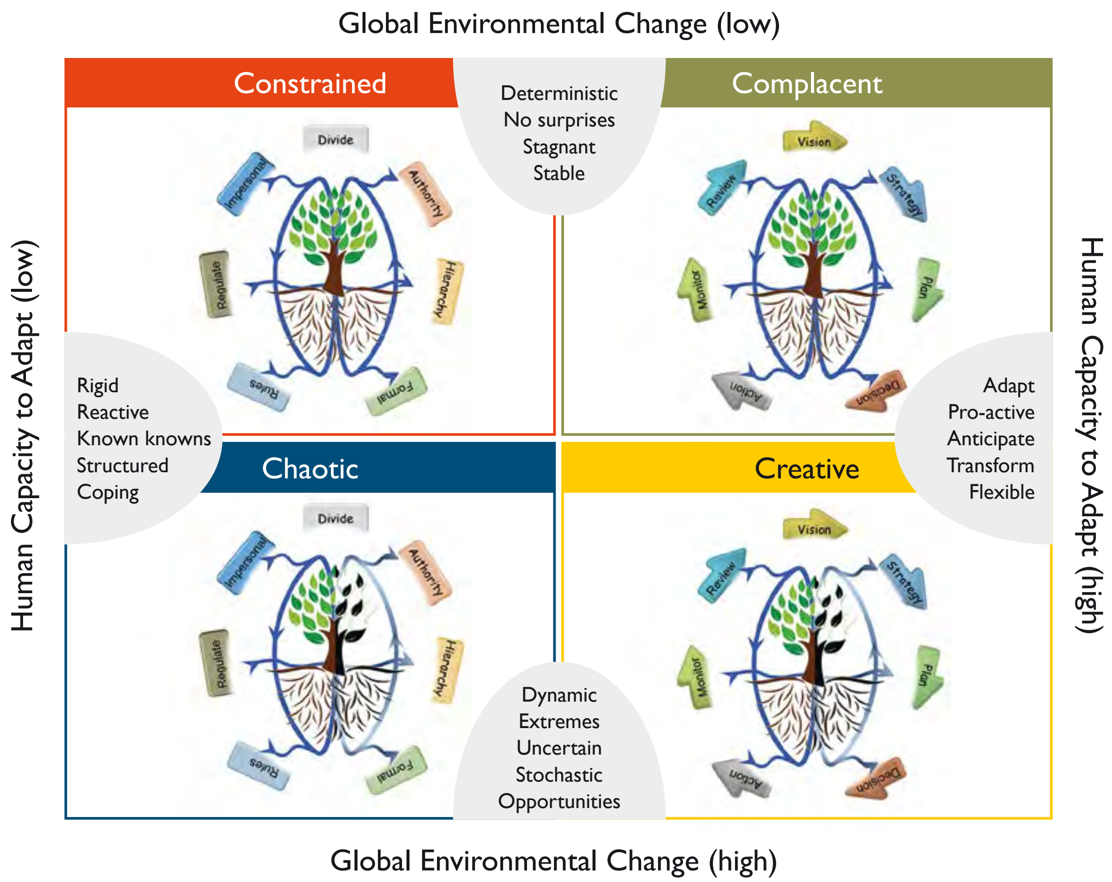
'Normal' generally refers to conditions that are similar to what they have been in the past (Hulme et al., 2009). The 'new normal' describes future conditions that are markedly different from the past (see Chapter 1). An example of such a 'new normal', is the anthropocene as a new geological epoch, where humans predominantly drive planetary changes (Zalasiewicz et al., 2010). The new normal in the context of impacts and consequences for changes in coupled forest-water systems will be characterised by greater complexity and uncertainty and shifts in risk perceptions. Such changes are generally viewed as undesirable, but some changes can also translate into new opportunities. Response options can include preventative measures (to counter undesirable change) and mitigation measures to reduce the impacts of such change or measures to exploit the opportunities brought about by change.

5.4.1 Future Impacts and Consequences

Representations of future possibilities can be useful for long term strategy development, but also to direct actions in the short term to promote a desired future state (Funke et al., 2013). Scenario planning originates in military applications, with Sun Tzu acknowledging the importance of planning in the face of uncertainty 2,400 years ago (Giles, 1910), whereas contemporary applications of scenarios include the RAND Corporation that started to investigate the scientific use of expert opinion in planning for the future in the 1940s (Landeta, 2006) and Royal Dutch Shell that used scenario tools to good effect in the 1970s, leading to a competitive advantage that enabled them to act quickly during the oil price shock of 1973 (Daum, 2001; Wilkinson and Kupers, 2013).

The determinants of change discussed in Chapters 2 and 3 include societal dimensions and environmental dimensions. In this chapter we aggregated the drivers in two

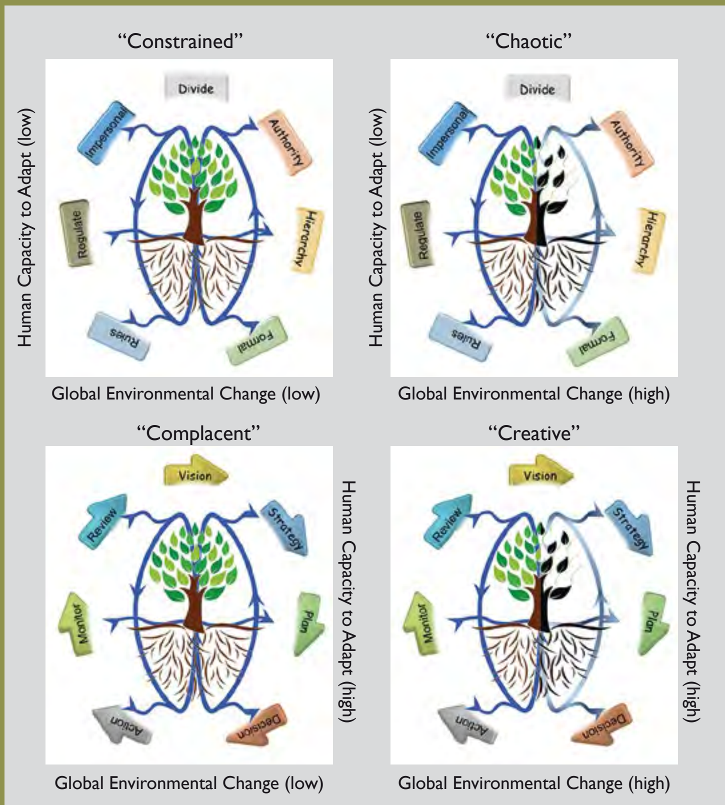
Scenarios for the forest and water domains by 2050 — Figure 5.3



Source: Authors' own elaboration

Box 5.7

Impacts and consequences of coupled forest-water systems in relation to scenario contexts



In a '**Constrained**' future, where environmental change is low and human capacity to adapt is low, there is a stable resource base but a constrained societal capability to service growing demands. The structure and function of the coupled forest-water system is variable, but with little directional change. Consequently, there is high confidence in the goods and services that the system can provide. The utilisation of these goods and services to meet societal development objectives is however constrained by limited innovation and adaptation and societal needs are not met in the future.

A '**Chaotic**' future has high levels of global change and low human capacity to adapt. The structure and function, and associated goods and services, of the coupled forest-water systems change, as do the benefits that society obtain from those services. This can lead to societal losses, compromising livelihoods and affecting human well-being.

A '**Complacent**' future combines low levels of environmental change with high levels of adaptive capacity. The goods and services that are provided at a high level of confidence allow for deliberation on the best portfolios of social and economic development opportunities. The adaptive capacity furthermore allows for learning by doing, thus ongoing adjustments to translate goods and services into development outcomes. However, unexpected impacts such as political dynamics, technological innovations or societal values can lead to unintended consequences, such as inequitable consumption and benefits at different temporal and spatial scales.

A '**Creative**' future combines a high rate of environmental change with a high level of adaptive capacity. This can lead to continuously changing goods and services, and adaptation to the change through ongoing evolution of social and economic activities to harness the dynamic potential. Since the growing demand is met with dwindling resources, incremental improvement is generally not sufficient. Radical and disruptive innovation is needed to meet development aspirations while countering environmental change. This scenario can either spark creative thinking or lead to despair if the challenges are deemed too great to overcome.

higher order drivers: the rate of environmental change and human capacity to adapt, as juxtaposed axes for scenarios related to forest and water by 2050. The resultant scenarios are presented in Figure 5.3. Perspectives on the impacts and consequences of these scenarios are outlined in Box 5.7.

In the scenario analysis presented here, the environmental change components and the relation with demand for these resources are represented by a tree (forest system) and blue arrows (water cycle). Considering future scenarios of environmental change, two options are possible: low levels of change in the forest system, represented by a symmetric tree, or high levels of change, represented by an asymmetric tree. Likewise, in future scenarios where environmental change in water systems is low, the water cycle is shown as a symmetric blue colour, whereas high levels of change in the water systems are indicated by an asymmetric colour of the water cycle.

The human dimensions of change include societal perspectives and solutions. Where there is a low capacity to adapt, the external circle is composed of rectangles, whilst when there is a high capacity to adapt, the circle is composed of arrows.

Under future scenarios with low levels of global environmental change, the coupled forest-water system can be expected to function as a dynamic system with natural variability, but within known ranges of variability. The flow regimes, linkages between climate and vegetation, and forested landscapes will be dynamic, but remain within boundaries that do not cause a major change in the coupled system.

Future scenarios with high levels of global environmental change, however, can cause disruptions in the natural systems and processes. Examples can include shifts in oceanic currents and sea surface temperatures with drastic effects on local climates beyond what can currently be predicted with any confidence or precision.

Variation along the other axis, human ability to adapt, will first of all depend on increased communication and synergy between the different knowledge systems (including local knowledge and values, physical, biological and social science-based knowledge and current policies) that may currently compete. If the various knowledge-to-action chains can be connected, societal adaptation may keep up with the environmental change and avoid the passing of irreversible thresholds.

5.4.2 Governance Responses under Different Scenarios

Under conditions of '*Low human capacity to adapt*', governance systems can be structured and efficient under stable conditions (thus '*Low global environmental change*'). A policy that assumed such a scenario is the US Endangered Species Act, which pursued a return of ecosystems to their 'historical' natural conditions and emphasised restoring habitat for single species, often to the exclusion of other species, but with increasing rates of global environmental change, those systems have been transformed beyond return, precluding more adaptive responses (DeCaro et al., 2017).

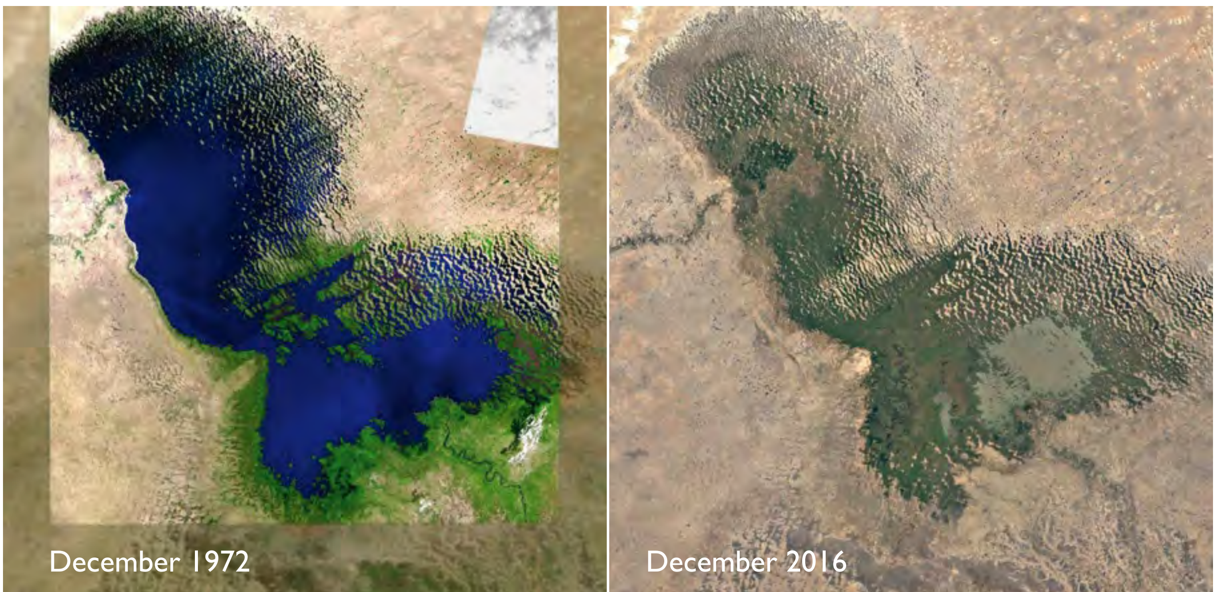
Under conditions of '*High global environmental change*', the same 'robust' systems can be slow to adapt to changes, which can render seemingly good policies less effective. An example of such a situation is Lake Chad, which shrank by 90% over a period of 35 years, which is putting pressure on sustainable food production, wetland habitat conservation, water management in transboundary basins and adaptation to climate change (Zieba et al., 2017). This situation must be taken into account in the formulation of enabling framework policies for managing resources in the Lake Chad area.

A scenario that seems to be a desired future is where '*Low global environmental changes*' are prevalent and where there is '*High human capacity to adapt*'. Although there are fewer uncertainties about environmental conditions in this context, the opportunities brought about by change are also limited. Policy options under these conditions will focus on sustainable practices on the supply side (forests and water) and greater efficiencies in the ever-increasing demand side (social and economic activities). There is a danger of complacency in this scenario, where environmental change may not be immediately apparent, such as the case in southeastern Spain, where intensive groundwater use and mining often exceed replenishment of supplies (Aldaya, 2017).

'*High human capacity to adapt*' is best demonstrated in conditions with '*High global environmental change*'. In such contexts, the ideal policies are framework policies that enable adaptive approaches and are supported by rapid feedback loops and learning systems. Ultimately, adaptive governance consists of a range of interactions between actors, networks, organisations, and institutions emerging in pursuit of a desired state for social-ecological systems (Chaffin et al., 2016).

The drivers of change are relevant at global, regional, national and local spatial scales, however, their manifestation would be different at each scale. Environmental change may be driven by global systems but has significant implications for local conditions. Likewise, the capacity to adapt to change can be facilitated through policies and processes at scale, but also depend on local capacity for action. These dimensions emphasise the need for cooperation across scales to mitigate change and increase adaptive capacity.

The implication of this scenario approach for water and forest interlinkage lies in its reframing of the social response to risk and uncertainties and in viewing policy as not just something within the government domain but also within that of markets and civic movements (Gyawali and Thompson, 2016). Such a triad understanding of power and policy is also what has been described by other schools of thought (e.g., Karl Polanyi (1944) with his concept of exchange, redistribution and reciprocity,



Lake Chad, straddling Cameroon, Chad, Niger and Nigeria, has shrunk by 90% over a period of 35 years
Photo © Mapdata: Google; NASA, U.S. Geological Survey, Landsat/Copernicus

as well as Lukes (1974) with his triad understanding of power). Such a framing will also have implications for management and governance (discussed in Chapters 6 and 7): firstly by defining policy as not just an action by governments but also by markets and civic movements; and secondly by bringing uncertainty and surprise – and the plural social response to them – to the centre stage.

5.5 Data Needs and Knowledge Gaps

- There is still much to learn about the ways eco-hydrological and socioeconomic processes can be integrated into forest and water resources management and planning strategies.
- More recognition of the shortcomings of current knowledge of biophysical processes is needed, along with relationship of these to the generation of ecosystem services and their values.
- We need further integration of biophysical information into the design of valuation scenarios, including new and innovative epistemological approaches for integration which can cope both with biophysical uncertainty and human 'ambiguities' (Byg et al., 2017). i.e. the agenda on valuation should be driven by a better representation of both the biophysical and social complexities (rather than necessarily on instrumental sophistication) (Martin-Ortega et al., 2017).
- Also, new efforts should be directed towards integrating monetary and non-monetary values, and operationalising these and other forms of value into decision making including relational values.
- While there is expertise concerning integrating monetary and non-monetary values at lower geographical scales, challenges remain in scaling up the analyses to regional and global scales.
- There is a need to build more sophisticated forest hydro-economic models based on integrated frameworks, to guide optimal resource allocation between forests (and other land uses) and water ecosystem services. Such models would need a detailed representation of forest functionality and its explicit relationship to watershed-based ecosystem services and their values (Ferraz et al., 2014)

5.6 Conclusions

Linkages between coupled forest-water systems and benefits to people are generally well understood but there are some limitations, specifically across spatial and temporal scales. The ability to attach values to these benefits is often lacking in terms of monetary metrics and even more so for non-monetary metrics.

The lack of a systematised approach to the valuation of water ecosystem services provided by forests hinders their incorporation into mainstream decision-making. Coupled forest-water systems' interactions are characterised by great complexity and uncertainty across space and time, in which trade-offs and synergies of goods and services are governed by complex environmental and management factors and interactions. Those environmental and management interactions are magnified when linked to the complex socio-economics and political boundaries given the multiple human well-being dimensions that can be affected by forest-water related ecosystem services. That leads to a recognition that complex socio-ecological forest and water interactions need to be managed holistically and in a more integrated way.

Changes to the underlying structure and function of coupled forest-water systems will affect available goods and services and consequent development options. While these linkages are conceptually well-understood, we need to improve our ability to characterise the relationships to support choices about management and policy options. Under future scenarios with low levels of global change, the coupled forest-water system can be expected to function as a dynamic system with natural variability, but within known ranges of variability.

The 'new normal' in the context of impacts and consequences for changes in coupled forest-water systems is characterised by greater complexity and uncertainty and shifts in risk perceptions. Such changes are generally viewed as undesirable, but some changes can also translate into new opportunities. However, consequence of changes in forest-based water services are not evenly distributed, affecting unequally people's rights and responsibility. Social justice and institutional arrangements need to be examined within the particular political and historical settings.

Responses under different future scenarios incorporate state, market and civic domains. For the coupled complex system to evolve towards sustainability, it is necessary for all these voices (including those of women and other marginalised groups) to be heard and responded to in a spirit of constructive engagement.

Current knowledge suggests that there is well established evidence on the fact that changes in the structure and functions of forests result in changes in the delivery of water ecosystem services, and these have consequences for the benefits people can obtain from forests. However, substantial levels of uncertainty remain in elaborating the details of the direction and magnitude of these relationships, but methods for improving our understanding of these consequences are rapidly developing. These methods are improving at the local/lower levels (e.g., catchment or lower levels), which means that the evidence they provide is quite solid, although still limited to specific places where data and monitoring systems are in place. However, there is more work to be done in terms of expanding this understanding to 'data scarce' locations. Much more needs to be done in terms of understanding and bringing this up to broader and global scales.

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

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6.1 Introduction

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

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

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

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

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

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

6.2 Management at Catchment Scale

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

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

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

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

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

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

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

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

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

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

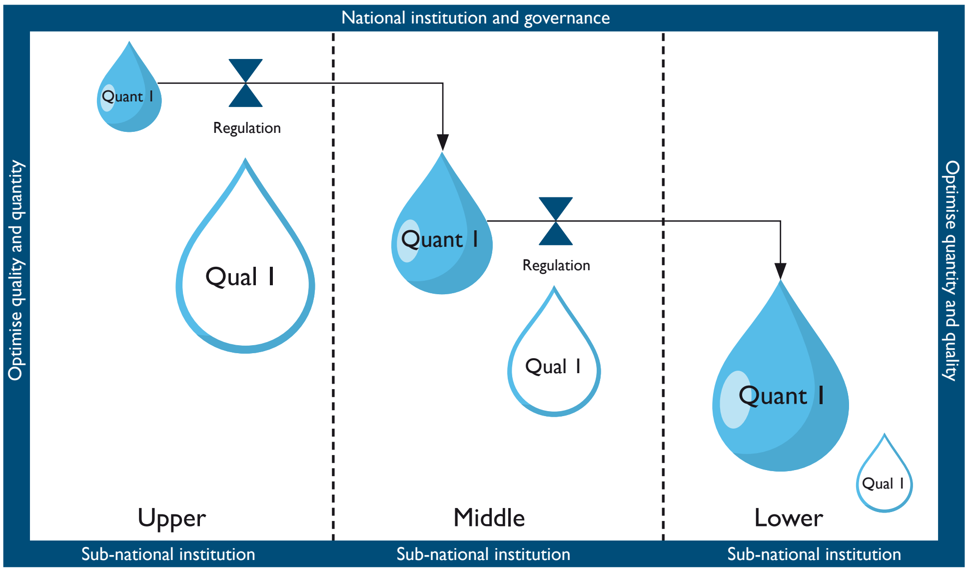
The timing of water flow is important to proper aquatic zone structure and function. Increases in annual water flow may not have beneficial impacts on aquatic populations if there is a reduction in seasonal water flow despite an overall annual increase. For example, protection of salmon populations in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

Riparian vegetation and landscape in Mongolia
Photo © Alexander Buck

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

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

Figure 6.1



Source: Daniel Murdiyarso (author's own elaboration)

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

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

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

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

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

6.3 New Management Options in the Context of the New Normal

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

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

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

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

Box 6.1

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Box 6.2

Impact of forest type on spring water quality and quantity

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

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

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

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

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

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

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

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

6.4 Socio-Institutional Options at Micro-Scales

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

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

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

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

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

Photo © Andre Purret

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

6.5 Socio-Economic Instruments and Incentives

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

Box 6.3

Connecting gender, water and forests

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

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

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

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

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

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

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

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

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

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

Box 6.4

Payment schemes for ecosystem based water services provided by forests in Latin America

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

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

Box 6.5

Key facts regarding compliance and additionality monitoring findings in developing countries

Compliance monitoring (conditionality)

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

Additionality monitoring

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

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

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

6.6.1 Identifying Critical Water Zones

A number of critical water zones may be identified across any landscape in which trees and forests exist. The identification of these critical water zones across the landscape is a crucial first step if water-sensitive land use management practices are to be implemented, and watershed services delivered (Groffman et al., 2003; Postel and Thompson, 2005). Exactly what constitutes a critical/sensitive water zone, and how best to identify and delineate these, may differ from country to country. However, the zones that are most commonly considered critical in terms of forest/water relationships include water source areas and riparian/wetland areas, as well as appropriate buffer zones around these (Nava-Lopez et al., 2016; Zheng et al., 2016). The importance of managing and protecting these critical water zones because of their contributions toward delivery of water of sufficient quantity and quality for downstream users, has been recognised internationally and mapped accordingly (Dudley and Stolton, 2003; Viviroli et al., 2007). This is particularly pertinent in countries characterised by highly variable climate and rainfall, which usually translates into uneven distribution of water resources and often a case of a small fraction of the country producing a disproportionately large amount of usable water. Box 6.6 provides an example of how the recharge zones for springs in the Hindu Kush-Himalaya (HKH) region are a focus of attention, recognising the importance of these critical water zones for the lives of millions of (especially poorer) households in the region.

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

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

Photo © Jitendra Bajracharya/ICIMOD

Box **6.6**

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

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

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

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

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

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

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

6.6.2 Mitigating Risk to Critical Water Zones

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

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

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

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

6.6.3 Stakeholder Engagement and Decision-Making around Critical Water Zones

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

Box 6.7

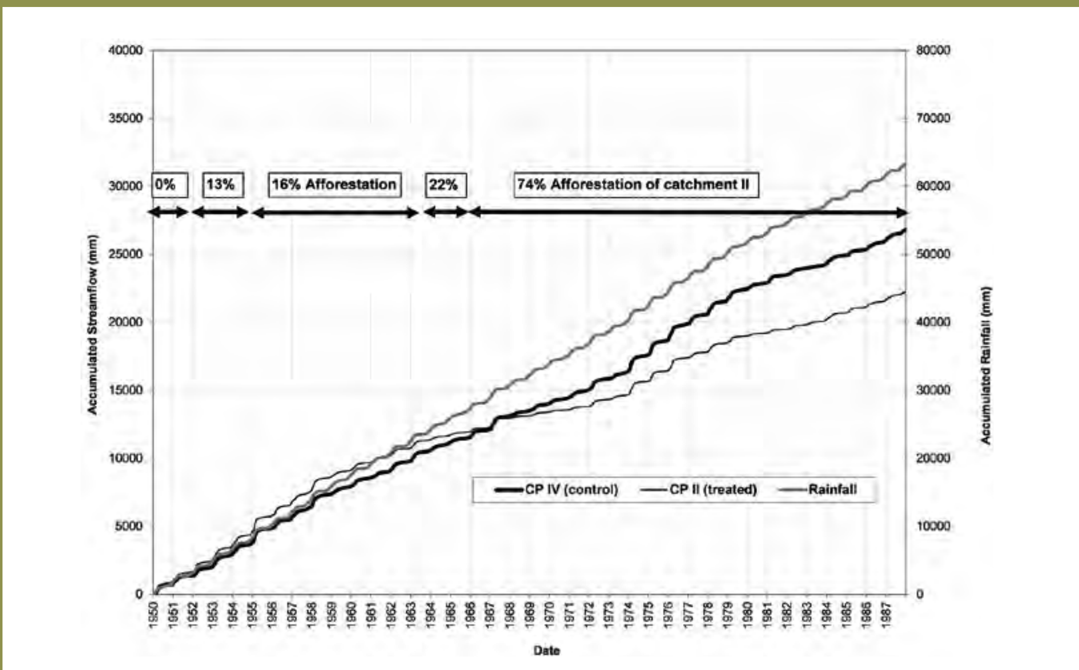
Figure 6.2

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

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

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

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



Source: Mark Gush (author's own elaboration)

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

6.7 Knowledge Gaps and Data Needs

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

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

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

Photo © Pixabay: Pexels

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

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

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

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

In addition, this chapter has highlighted that there are a number of ways in which reciprocal relationships across forest-water landscapes are managed in multi-stakeholder decision settings. While there is a growing emphasis within some policy, academic and donor literature on the importance of mediating these relationships through market, or quasi-market structures, there is a need to recognise that there are alternative ways to organise these social and institutional settings which build on mutual commitment and reciprocity, but do not necessarily rely on the logic of markets and incentives. There is a need for more systematic evidence on these plural institutional forms, and what makes them work in specific settings, to expand the toolkit of interventions beyond the current focus on payments and markets.

6.8 Conclusions

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

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

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

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

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

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

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

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

Given the vital role water plays even in facilitating the continuous sequestration of carbon in standing forests, a lack of understanding of landscape-scale effects amongst the hydrological and forest science communities and policymakers, is of increasing concern as it raises the risk of policy failure in managing forest resources for water quality and quantity.

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

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

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Chapter 7
Governance Options for Addressing Changing Forest-Water Relations

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7.1 The Problem of Governance – Knowledge, Scale, Institutional Structure and the Technology of Governance?

From a systems perspective, governance represents a key driver when it comes to the potential for addressing rapid environmental, climate, social and even technological change. As our knowledge of forest-water interactions and their potential to improve human welfare expands, new opportunities emerge to optimise the strategic use of natural resources in ways that may bring multiple spinoff benefits to those who depend on these resources for their livelihood and prosperity.

Even without considering the constraints of the 'new normal' and the challenges imposed by climate change, land use practices could be modified in ways that can potentially optimise natural resource availability across space and time. On the other hand, the increasing threat posed by both climate change and the rise of the 'new normal' further intensifies the need to better understand forest-water interactions, and to raise our proficiency at puttingthem to good use.

If the management of forests for water is genuinely to be considered, then a number of aspects need to be addressed before the principal set of priorities can be adequately and reasonably reordered:

1) First, there needs to be some relative agreement that the forest-water relationship should be prioritised over the more common forest-related goals of producing timber and/or sequestering carbon. Despite the comparatively uncontroversial notion that forested watersheds can help provide clean drinking water (see e.g., Box 7.1), such strategies are far less frequently employed than might be possible. Likewise, despite the uncontroversial notion that forests depend on water for their survival, this logical reordering of priorities appears to be less straightforward than it seems. The increasing number of forestation projects (defined as a generic term for projects aiming to increase tree cover regardless of baselines, species or methods used) that have failed to adequately consider the water demands of newly introduced foliage suggest there is a clear need to convince practitioners and communities that increasing forest cover is not necessarily good under all circumstances. Considerable care must be taken, for example, in the choice of species that are well-adapted to local circumstances (see for example the discussions of 'potential natural vegetation' (PNV) in (Maes et al., 2009, 2011; Wahren et al., 2012), as well as the PNV data collection project (Ramankutty et al., 2010; see also Little et al., 2009; Aranda et al., 2012).
2) Second, attention must be paid to the scale, scope and structure of the political institutions governing forest-water interactions. Many of the newer scientific insights regarding forest-water interactions are potentially observable from a much broader geographic and spatial perspective, leading to concern in particular about the spatial organisation of land use practices across hydrologic space (Ellison et al., 2017; Keys et al., 2017). As these authors demonstrate, this has implications for the related governance structure. The general mismatch between natural ecosystem scales and legal jurisdictions where both up- and downstream as well as up- and downwind forest-water relationships are concerned, ultimately requires a radical rethink of how to manage and govern forest-water interactions, and how to address some of the imbalances that can occur as a result of the failure to consider, in particular, up- and downwind forest-water relationships (see also e.g., Dirmeyer et al., 2009; Wang-Erlandsson et al., 2017). Forest-water relationships that do not fit neatly into existing political-institutional and decision-making frameworks are often ignored.
3) Third, social-ecological systems such as the forest-water-climate-people system suffer from multi-scalar challenges, including scale mismatches that affect the ability of the social system to address the challenges presented by the ecological system (Cash et al., 2006). The scalar mismatch between goals and means has plagued many aspects of natural resource governance (Holling, 1986). There are, however, many relevant and important exceptions to this rule, for example, South African forest taxation or the management of forested watersheds as water resources. The general trend has perhaps been toward increased awareness of, and attention to, the management of forests for water. But water governance institutions generally tend to focus on the local or catchment scales and are considered separately from forest governance. Moreover, forests are generally managed either at the scale of the forest stand, based on private forest ownership, or at the regional or national scale, generally speaking, irrespective of water governance concerns.

The relative primacy of concerns over water often means that forests and forest-water interactions are not adequately integrated into the water management concept.

On the Nam Ou river, Luang Prabang, Laos. Many local people depend on water – both for economic and social reasons
Photo © Peter Tarasiewicz

The reasons for this remain unclear. People often have a closer relationship with water than forests, and forests have often been defined on the basis of the exclusion of local people and restrictions on their land use. In lower watersheds and especially in delta regions where high concentrations of people live, water management has little explicit relationship with forests and trees. The conceptual relationship is perhaps strongest in middle/upper watersheds, with conflicts in accessible locations where logging and conversion to other land uses have historically been most attractive. Where attempts to tackle the forest-water system have occurred, a conventional focus on the partitioning of water resources across catchment scales has typically led to a focus on the up- and downstream management and uses of water.

To increase awareness of the importance of forests for water, the United Nations' Sustainable Development Goals (SDG) framework as well as the United Nations Forum on Forests' attempt to incorporate the SDGs into its own set of guidelines (the United Nations Forest Instrument (UNFI) and the UN Strategic Plan for Forests (UNSPF) for the period 2017-2030), have helped to frame the general debate about optimising environmental relationships and act as important agenda-setting tools. Moreover, the SDG agenda is well placed in the international arena, since all countries are encouraged to consider and potentially mobilise environmental resources in ways that can help improve human welfare. At the same time, the explicit links across the multiple forest-water interactions and their potential usefulness in the natural resource management context still need to be meted out and appropriately allocated. This requires both sufficient knowledge about the benefits of these forest-water interactions, as well as the potential restructuring and reform of the social governance institutions that must put these in place.

Livelihoods and the interests of individuals and communities are frequently intimately intertwined with forests and/or water, resulting in powerful and important interests and demands influencing decision-making on the use and management of these resources (Dewi et al., 2017; van Noordwijk, 2017; Watson et al., 2018). Thus, a wide range of socio-economic and political interests intersect with an increasingly complex set of forest-water interactions. For effective governance, these need to be optimised in suitable ways.

This chapter addresses the question of forest-water governance from the systems, willingness, ability and capacity to act perspective, as it applies both to natural resource governance in general, as well as to the project of forest-water governance in particular. Thus, we consider governance from a systems perspective (7.2), look for expressions of the political will to act on the forest-water agenda (7.3), consider the ability to act based on the nature and structure of existing governance institutions (7.4), and finally, consider the capacity to act based on whether the requisite knowledge exists, as well as the availability of appropriate models for action (7.5). Section 7.6 highlights persistent research gaps, while 7.7 concludes.

Box 7.1

'Forever wild' for water supply in the Adirondack Forest Preserve of 1894

Conservation of forests has been a central tenet of managing the drinking water supply of New York City for over 150 years. During the latter half of the 1800s forest multiple-use strategies in the headwaters of the Hudson River attempted to allow for timber harvest, while protecting the water supply, wildlife and recreation. An influential publication concerning this decision was 'Man and Nature' (Marsh, 1864) which propounded the value of forests in protecting water resources. Frustration with the 'balance' that allowed for too much cutting, exacerbated by forest fires and a drought, led to an unprecedented measure to protect forests when the state constitution of New York State was drafted in 1894. The state legislature required that all state-owned land (about half of the total area) in the 2.5 million-hectare Adirondack Forest Park was to be 'forever wild'. The decision is an excellent example of the power that ideas about forest-water relations can have for policy (Michaels et al., 1999). More recently, new measures to guide forest management with a primary focus on protecting drinking water supplies have been implemented in other forest areas of New York State. Particulate and pathogen concentrations were reaching levels where expensive water treatment plants would be required. Instead, forest management measures provided a more cost-effective way of controlling particulates and preserving the water supply (NRC, 2000).

7.2 The Challenge for Governance – A Systems Perspective

Political institutional features such as democracy, transparency, competitive party systems, open media, etc. all tend to be positively related with indicators of the quality of governance, so it is likely that frameworks generated from these contexts would be more effective (Weaver and Rockman, 1993; Persson et al., 2003; Buchholz et al., 2008; Mills et al., 2008; Rothstein, 2011).

The following factors have been identified with respect to the overall quality of governance and potentially, natural resource governance:

- **International agenda-setting/treaty building:** Placing new ideas and issues at the centre of international negotiations and agenda-setting represents one of the first important steps to devising meaningful solutions to important global problems. This not only requires a sufficient institutional framework, but requires the commitment of more internationally-minded actors. The current SDG framework within the United Nations is a prime example, as is the UNFF's parallel focus on integrating the SDG agenda.
- **The evolving need for new institutional frameworks:** Given that institutions typically represent the interests of those within them, if the institutional framework is not large enough to have complete purview over the relevant eco-hydrologic relationships, some relationships may well take precedence over

others. For example, while up- and downstream interests and concerns are more commonly represented, up- and downwind interests and concerns have not even begun to enter the political and institutional vocabulary.

- **Democracy, decentralisation and polycentric governance:** Institutions that can look both upward (to higher-level governance institutions) and downward (to more local-level governance institutions and interests), without ignoring political will and interests at all other levels of governance are more likely to be able to arrive at policy outcomes adapted to broader communities of interests. The necessity of considering a broader spectrum of interests and adapting these to relevant policy outcomes is one central motivation for re-thinking the institutional features underpinning the quality of natural resource governance. In this sense, democratically-driven, participatory and polycentric governance frameworks with multi-centred authority, are potentially better suited to addressing the problems of scalar mismatch and the spatial dislocations of (potentially) competing interests.
- **Strategies for overcoming entrenched interests:** The effort to provide meaningful solutions regarding natural resource governance, is frequently either slowed or completely stalled by the interference of powerful and entrenched special interests. Scenario analyses (see Chapter 5) may provide one potential strategy for finding new alternatives to old and largely unsolved problems. This approach has the advantage of creating buy-in to commonly devised policy options through the apparatus of participatory and strategic brainstorming.
- **Actors versus institutions and the necessity of leadership:** Though there does not seem to be any perfect strategy for finding good leadership, there is no replacement for those few individuals who are willing to champion important ideas and goals. Good leadership often seems accidental and is rarely planned. Institutional features such as good governance and the presence of good skill-building educational institutions may nonetheless support the likely emergence of such leadership. And these institutions may themselves be more likely under more polycentric systems.

Institutionally-driven solutions are clearly no panacea and cannot guarantee positive, natural resource governance solutions. In this regard, they may represent an important, but insufficient condition for success. Governments may, for any number of reasons, opt for less than optimal natural resource governance solutions. Economic interests and security concerns are among the many factors that can easily converge to derail an otherwise positively-minded executive or legislative branch of government (e.g., Altenburg and Lütkenhorst, 2015). Moreover, political systems are frequently weighted toward more powerful individuals and groups, or those for whom the costs of collective action are either lower, or the benefits more highly rewarded (Olson, 2003).

Even with firmly entrenched democratic institutions, there is no guarantee that environmental issues will be adequately addressed. Governments require the presence of actors with an interest in environmental protection and sound natural resource governance to engage in appropriate action (e.g., Olson, 1993). The development of an eco-centric foundation within the recently announced five-year plan in China (Ouyang et al., 2016) is a positive example of the progress made in accepting the importance of the environment for human well-being, despite the fundamental lack of more democratically-oriented or polycentric institutions. Democratic political systems can fail in their environmental responsibilities and are entirely capable of choosing leaders who have no interest in, or knowledge of, environmental issues and concerns. In contrast, even highly centralised and autocratic systems, when inhabited and motivated by well-meaning actors, can potentially arrive at optimal solutions far more rapidly than democratic systems that are typically based on lengthy decision-making processes.

Although the concept of a universal model of 'good governance' has been roundly criticised (e.g., Masson-Vincent, 2008), the principles of accountability, legitimacy and transparency (World Bank, 2009; PROFOR & FAO, 2011) have in the past been called upon to set the standard for ensuring sustainable forest management. Such principles tend to be more strongly defended in systems that are democratic and based, for the most part, on the principles of participatory governance.

The relative advantages of polycentric forms of governance – marked essentially by frameworks that are more open and responsive to signals from multiple levels and directions, and that recognise multiple centres of power – are gradually being recognised (Ostrom, 2010a). Generally speaking, there seems to be relatively broad support for the idea that the more governments are polycentric in character, the more likely they will be able to deliver quality governance (Ostrom, 2010a, 2010b; Gao and Bryan, 2017). This recognition builds upon experience from multi-level governance frameworks such as those in the European Union and in some more federal systems (e.g., Hooghe and Marks, 2003; Gillard et al., 2017). And the emphasis on polycentric forms of multi-level governance has also found support in the forest governance literature (see in particular Mwangi and Wardell, 2012, 2013). To cite Andersson and Ostrom (2008), "the complexity of many natural resources requires sophisticated governance systems capable of recognizing the multiscale aspects of natural resource governance and of seeking to determine optimal policy outcomes, despite the presence of countervailing incentives".

Presidential systems with strong veto powers provide significant authority and power to single individuals. Likewise, majoritarian party systems (based on single member electoral district systems) tend to thin out the ranks of political competition and reduce the potential for opposition. In contrast, institutions which support concepts of 'shared governance' may prove less susceptible to the whims of individual rulers. Parliamentary systems, in particular those that are governed by multi-party systems, tend to divide power and authority across a broader set of individuals, in part through the mechanism of coalition governments. Moreover, power and authority in multi-party parliamentary systems are continuously subject to review and potential recall through parliamentary procedures that allow for the interim removal of leaders who require parliamentary support for their survival in office. In contrast, presidential systems, tend to enjoy fixed terms and leave comparatively few options for the removal of standing presidents.

7.3 Political Will and the Forest-Water Agenda

Primarily as a result of climate change, forest-related policy objectives have significantly shifted toward the management of forests for carbon. To date, the traditional paradigm has been to manage forests for their ability to provide biomass, for their multi-functional uses, and/or for their ability to sequester carbon.

7.3.1 International Agreements and Programmes

The December 2015 Paris Agreement signed by the members of the 23rd Conference of the Parties (COP) under the United Nations Framework Convention on Climate Change (UNFCCC) led to a broad range of countries deciding to include forests into what are now called Nationally Determined Contributions (NDCs). To date, a total of 73% of the 189+ countries to submit intended NDCs have included Land Use, Land Use Change and Forestry (LULUCF) in their mitigation (and/or adaptation) plan, and forests are expected to contribute approximately 25% of the total emission reductions by 2030 (Grassi et al., 2017). The principal emphasis of the Paris Agreement remains on carbon; concerns about the availability of water and the potential impacts, both positive and negative, of forest-water interactions on the hydrologic cycle are absent from this agreement. The focus on the carbon sequestration potential of forests that resulted from previous UNFCCC discussions under the Kyoto Protocol led to a similar emphasis without, however, incorporating a similarly forceful declaration on the importance of water. Thus, the fact that so many countries are now beginning to pay more attention to the potential role of forests in the climate change mitigation framework presents both an opportunity and a challenge for water as it could potentially lead to unexpected and unintended outcomes.

Flooded neighbourhood in the US after Hurricane Harvey in 2017

Photo © iStock: Karl Spencer

Water concerns have typically been of secondary importance. At the same time, the increasing scarcity of, and rising demand for, water may be shifting the balance toward increasing concerns about water (Vörösmarty et al., 2010; Mekonnen and Hoekstra, 2016). Climate change has exacerbated, and will continue to further exacerbate, these concerns through rising temperatures, changes in precipitation patterns and amounts, the increasing likelihood of droughts and the increasing occurrence of less frequent but more intense rainfall events (Fischer and Knutti, 2015), as well as the potential flooding these imply.

7.3.2 Water and Forest Goals Side by Side

By and large, forest-water interactions have been almost entirely ignored in the management of global freshwater resources (Ellison, 2010; Ellison et al., 2012, 2017; Vörösmarty et al., 2015; Mekonnen and Hoekstra, 2016). On the other hand, there are many emerging fora in which these issues are increasingly being discussed and pushed onto the international and also national and local agendas (e.g., Ellison, 2010; Creed et al., 2016; Ellison et al., 2017).

Emphasis on increasing carbon capture as part of global climate policies, especially in dry areas (often avoiding direct competition for land with local populations in more hydro-climatically endowed areas), has resulted in a direct trade-off between blue water production and carbon sequestration in reforested areas (Jackson et al., 2005; Benyon et al., 2006; Trabucco et al. 2008; Filoso et al., 2017; Garcia-Chevesich et al., 2017). Numerous forestation projects have failed to consider adequately the water demands of newly introduced foliage, or to use species that are well-adapted to local conditions (Little et al., 2009). All too often, fast-growing species have been used without thinking about the relative impacts on the locally available water supply (e.g., Jackson et al., 2005; Benyon et al., 2006; Trabucco et al., 2008; Garcia-Chevesich et al 2017; Filoso et al., 2017). Lessons from these projects have helped to initiate and further promote concerns about the impacts of managing forests only for carbon (Jackson et al., 2005; Trabucco et al., 2008; Filoso et al., 2017). More often than not, knowledge of the forest-water relationship is inadequate, has not even been considered, or fails to be adequately contextualised, in favour of generalisations.

While such experiences have challenged the dominant forests-for-carbon paradigm, it is above all the improved understanding of positive and beneficial forest-water interactions that have led to a call for an explicit shift in the focus of the management of forests for water (Ellison et al., 2012, 2017; van Noordwijk et al., 2014; Ilstedt et al., 2016; Syktus and McAlpine, 2016). Many actors, public and private, support forestation strategies in order to restore the world's forests, but few have turned their focus towards an integrated view of the potential benefits of forest and water interactions. Recently, WeForest together with the Global Partnership on Forest Landscape Restoration (GPFLR), have attempted to shift the focus toward forests and water, and are currently involved in efforts to develop a Forest Landscape Restoration (FLR) set of principles that would help to encourage donors and recipient countries to place more of an emphasis on these important interactions.

Though mainstream approaches to forest-water interactions have over the past decades focused on the fact that trees and forests 'use' water (Bosch and Hewlett, 1982; Farley et al., 2005; Jackson et al., 2005; Vose et al., 2011; Filoso et al., 2017), this literature has never really attempted to determine what happens to the atmospheric moisture that is produced by trees and forests through the process of evapotranspiration. A major step in the evolution of thinking on forest-water interactions is to complete the logical and conceptual shift from an almost exclusive focus on demand-side, catchment focused thinking, to one that incorporates the supply-side, up- and downwind aspects of forest-water interactions (van der Ent et al., 2010; Ellison et al., 2012, 2017; Keys et al., 2016; van Noordwijk et al., 2014; Wang-Erlandsson et al., 2017).

The concept of ecosystem services and the underlying view that forests and the water they process and regulate provide invaluable returns to human civilisation, is ultimately a more recent phenomenon, arising primarily at the very end of the 20th century and becoming more prominent in the 21st century (see Chapter 5). International support for national and local actions has been at the 'motivational' (rather than the regulations or incentives) level. Milestones in the international recognition of the forest-water issues at stake include: the 2002 Shiga Declaration on Forests and Water (http://www.rinya.maff.go.jp/faw2002/shiga.html), the Millennium Ecosystems Assessment (MEA, 2005), various meetings of the Ministerial Conference on the Protection of Forest in Europe (renamed Forest Europe), in particular the 2007 'Warsaw Resolution 2 – Forests and Water' have begun to affect thinking on forest-water issues (Calder et al., 2007; Ellison, 2010; Creed et al., 2016). The FAO, for example, has initiated comparatively intensive discussions on forests and water with the creation of a 'Forest & Water Action Plan' announced at the 2015 FAO World Forestry Congress in Durban, South Africa (Ellison et al., 2017). The FAO's current efforts are focused on the development of a Forest and Water Monitoring Framework ('FAO Forest-Water Monitoring Framework, A Year Later'). Some NGOs are likewise working on similar agendas. The 'Gold Standard' certification body for forest/climate investments (https://www.goldstandard.org/) has also recently undertaken initial efforts towards integrating forest and water issues into their reforestation agenda, though it remains unclear what form this might take.

7.3.3 Sustainable Development Goals

The SDGs express commitments from all UN Member States to tackle the various challenges of sustainable development in a coherent way. The 17 SDGs, adopted by the UN General Assembly in September 2015 (UN, 2015), with 169 associated targets, are aimed at balancing the three dimensions of sustainable development (economic, social and environmental) in an integrated and indivisible way.

Given the urgency of challenges that face us in the Anthropocene, the United Nations' SDGs offer an opportunity to revisit the case for cooperation across different sectors, development priorities and across the water-forest-climate nexus (Brondizio et al., 2016; Lima et al., 2017). A seven-point scale has been proposed to describe interactions between goals: cancelling, counteracting, constraining, consistent, enabling, reinforcing and indivisible (Nilsson et al., 2016). Where interactions among SDGs are primarily negative (cancelling to constraining), trade-offs need to be understood and managed; where interactions are primarily positive (enabling to indivisible), synergies can be achieved. The SDGs represent an important milestone towards a global social policy (Deacon, 2016), even though the SDG document as such was found to fail in improving the architecture of global social governance, thereby reverting back to an era of strengthening national sovereignty that reflects the current 'mood' in many countries.

The SDGs feature forests and water multiple times and indeed forests could be said to be linked to almost all of the SDGs in one way or another. However, the SDGs continue to treat forests and water separately, thus reflecting the strong sectoral pre-determination of policymaking on forests and water. Whereas the 17 goals are listed in the resolution, their interrelationships are not explicitly defined (other than acknowledging that they are indivisible). In exploring the role of policy and governance in promoting development outcomes, the SDGs can be organised into functional groups (Figure 7.1).

Dependencies between the functional groups indicate causality. For instance better basic services will promote development outcomes, whereas improved equity is supported by access to natural resources. And improved equity can also lead to improvements in natural resources. Referring to the resource perspective on forests and water, SDG 15 is to "*Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss*". Target 15.1 specifically addresses forests and water to "*… ensure the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services, in particular forests, wetlands, mountains and drylands …*", whereas Target 15.2 calls for "*… implementation of sustainable management of all types of forests, halt deforestation, restore degraded forests and substantially increase afforestation and reforestation globally*". The other ten targets under SDG 15 address related aspects of life on land including mountain ecosystems, degradation, benefit sharing, poaching, invasive species, integrated planning and financial resources. Whereas the goals and

Sustainable Development Goals' functional groups and dependencies Figure 7.1

Source: Authors' own elaboration

associated targets may have independent merit, it is also useful to understand which other goals will aid in achieving SDG 15, but also, to what extent advances under SDG 15 will support other goals.

Improvements in equity (SDGs 4, 5 and 10) can create conditions for more equitable utilisation of forest and water resources, for instance through increased knowledge, resources and alternatives, thus having a positive impact on natural resources. Likewise, improvements in the provision of basic services (SDGs 6, 7, and 12) can reduce the impact of unsustainable consumption and waste products. The goals related to institutions (SDGs 8, 9, 11 and 16) can support the achievement of natural resource goals through effective policies, processes and practices.

Progress towards achieving the natural resource-related goals (SDGs 13, 14 and 15) can in turn support the provision of basic services, provide conditions for equitable development and provide a sustainable basis for institutions to translate resources to development outcomes (SDGs 1, 2 and 3). The partnerships defined in SDG 17 underpin all the goals.

To understand the role of the SDGs in the context of models of governance and policy objectives, we need to understand the dynamics of governance at the global, national and local levels. Weiss and Wilkinson (2014) state that many of the most intractable contemporary problems involve the overreach of trans-national non-state actors and that addressing them successfully requires actions that are not unilateral, bilateral, or even multilateral, but rather global, given that "everything is globalised – that is, everything except politics". The policy, authority, and resources necessary for tackling such problems remain vested in individual states rather than collectively in universal institutions. The SDGs tread this precarious line between national sovereignty and international intent. The means of implementation of the UN Resolution (UN, 2015) emphasise linkages to other international agreements and implementation through national policies and processes. Thus, whereas the intent and commitment are provided at a global scale, the emphasis on implementation is at the sovereign national level. Since forest and water systems span national boundaries, the SDGs (particularly SDGs 6, 12 and 15) provide a valuable means to support national action and cross-national cooperation for regional or global benefit.

In the framework of the water-forest-climate nexus, three SDGs are particularly relevant: SDG 6 on water, SDG 13 on climate and SDG 15 on terrestrial ecosystems. The SDGs, by their very nature, are framed in the context of human-wellbeing, which can directly be associated with the ecosystem services framework. For example, the role of coastal trees in protecting low-lying cities from storm surges is both part of the climate-forest-water nexus and corresponds to the regulating service of forests and coastal wetlands, whilst contributing to SDGs 13 and 15. Table 7.1 highlights the links between functions provided by the forest-water system and ecosystem services, whilst linking them to three of the SDGs. Relevance to other SDGs, particularly at the level of specific targets, can also be identified, however we consider the three most relevant SDGs here for illustrative purposes.

The United Nations Forum on Forests (UNFF) has likewise begun to integrate the SDG framework into its overall forest policy guidelines. In particular, the United Nations Forest Instrument (UNFI) and the UN Strategic Plan for Forests (UNSPF) for the period 2017-2030 and beyond, represent important steps along the path toward sustainable management of the world's trees and forests from a more water-driven perspective. In particular, Article V of the UNFI and Global Forest Goal 6 of the UNSPF open pathways for the integration of both currently and newly recognised forest-water interactions in the sustainable forest management framework. These frameworks, however, require further elaboration and concerted efforts in order to bring about the successful integration of the forest-water paradigm into the forest management framework.

Both the science and the science-policy interface still require considerable effort in order to be able to fully integrate forest-water interactions into the SDG, UNFF and other forest and water management frameworks. Without substantially improved knowledge and awareness of how forest and water interactions can be put to good use, more optimal outcomes are not very likely.

The relative success of initiatives such as the UNFCCC's 2015 Paris Agreement, as well as the Convention on Long-Range Transboundary Air Pollution (Box 7.2), on the other hand, suggest the general will to act is present and can be mobilised on a grand scale, in particular in cases where humanity's well-being is threatened. At the same time, the relative slowness of the UNFCCC's response to the climate challenge further suggests that such action is not easy to bring about and can require considerable expenditure in terms of resources, time and effort.

7.4 Governance as Driver and the Ability to Act – Creating Systems Potential

While scientifically it may be clear why improved links between water and forests make good resource management sense (Nutley et al., 2007), there is little acceptance of this in political circles at any level of governance (Pielke, 2007).

Table 7.1 **Ecosystem services, forest-water system functions and SDGs**

SDG	Ecosystem Service	Ecosystem function of forest-water system (see Chapter 2)
SDG 6 – water	Provision of reliable and clean water	W1 - Water transmission W3 - Gradual release of stored water supporting dry-season flows W4 - Maintaining water quality (relative to that of rainfall) W9 - Ecological rainfall infrastructure and biological rainfall generation, including atmospheric moisture recycling
SDG 13 – climate	Climate change mitigation, and adaptation	W2 - Buffering peak flows W5 - Stability of slopes, absence of landslides W7 - Microclimate effects on air humidity, temperature and air quality W8 - Coastal protection from storm surges, tsunamis W9 - Ecological rainfall infrastructure, biological rainfall generation, including atmospheric moisture recycling
SDG 15 – terrestrial ecosystems	Ecosystem services associated with biodiversity from terrestrial ecosystems	W5 - Stability of slopes, absence of landslides W6 - Tolerable intensities of net soil loss from slopes by erosion W9 - Ecological rainfall infrastructure and biological rainfall generation, including atmospheric moisture recycling

Box 7.2

Convention on Long-Range Transboundary Air Pollution

When looking for models of international cooperation to protect ecosystem services, one of the signature successes is the Convention on Long-Range Transboundary Air Pollution. Signed in 1979 under the auspices of the United Nations Economic Commission for Europe, the treaty itself is straightforward in that the parties (now 51 countries), simply recognise air pollution as a threat that should be reduced, without any specific commitments. But the eight protocols that have been negotiated within the framework of the convention have not only set up specific goals but created a basis for remarkable reductions of air pollutants including heavy metals, volatile organic compounds and oxidising sulphur. Two lessons from this convention of relevance to governance of the forest-water system are:

1. the methods used, i.e. "*exchanges of information, consultation, research and monitoring*". Building a scientific basis for decisions, including the collection of key data, and a forum for discussing science to work out issues has been a key part of the convention's success.
2. the focus on long-distance "*air pollution whose physical origin is situated … under the national jurisdiction of one State and which has adverse effects in … another State at such a distance that it is not generally possible to distinguish the contribution of individual emission sources or groups of sources*". This has parallels to the issues vexing the discussion of forest and water where it is unclear where the water put back into the atmosphere by forests in one place will actually come down.

(N.B. Both of the quotes in the bullet points come from the Convention, which can be accessed at:

http://www.unece.org/fileadmin/DAM/env/lrtap/full%20text/1979.CLRTAP.e.pdf. See also Strahan and Douglass, 2018).

Men drawing water from Itare River – one of the 'water towers' in Kenya

Photo © Sande Murunga/CIFOR

In many countries, the governance and management of both water and forests in a practical sense are often seen as low priority among government officials (Wallace et al., 2003). Frequently this is a legacy of past governance arrangements even dating back to colonial times in many places, and until this (im)-balance of power within and between government agencies is addressed, it is unlikely that there will be significant change in resource allocation to support more effective governance within the water and forest sectors (Biermann et al., 2009; Devisscher et al., 2016). Even within the water sector itself, a majority of countries fail to integrate those responsible for water resources and provision with those staff engaged in waste water management. In both sectors, however, forest-water interactions could have an important role to play.

The lack of attention paid to forest and water issues is reflected, for example, in the way that data is collected on illegal logging and water withdrawals. While the problem of illegal logging is well documented (e.g., Kleinschmit et al., 2016), the widespread practice of illegal water withdrawals and connections to municipal water systems is less publicised. In the water sector, this means that official water resource plans may be ineffective from the outset, with practical difficulties resulting for water utilities and other agencies who are faced with the problem of 'unaccounted' water use. Regarding the problem of illegal logging and other unofficial access to forest resources, this again gives rise to inaccurate data resulting in an increased likelihood of policy failure when attempts are made to integrate the sectors.

At the local community and household scale, access to water is essential yet inequitably distributed around the world (Sullivan, 2002). In many areas, lack of access to water for domestic use and food production is the result of poor governance arrangements. The improvement of water provision has much potential to reduce poverty, as labour availability of household members will be increased (Sullivan et al., 2003). Similarly, access to healthy forest systems provide multiple benefits for households, including increased food security, especially in times of economic stress (Sullivan, 2003).

Although most forests are found on territorial land governed by a range of customary institutions and rights (Peluso, 1992), official ownership falls to governments in over 70% of the world's forests (RRI, 2014). Yet local institutions structure villagers' attitudes, social relationships and even technology in such a way as to ensure the sustainability of forest management and to secure collaboration in managing forest notably, for water. Forest decentralisation has therefore become a key indicator for 'quality of governance', which has promoted both local participation as well as forest recovery worldwide (Agrawal et al., 2008; Xu and Ribot, 2004; Rothstein 2011). For example, in China two-thirds of forestlands are collectively-owned by local communities. The Collective Forest Reform has triggered tree planting and increased forest cover, therefore contributing to ecosystem functioning (Hua et al., 2018). In Indonesia, the hopes of customary communities have recently been bolstered by Constitutional Court assurances that they have the right to control customary forest (Myers et al., 2017). In the payments for ecosystem services (PES) framework, community-based models have been among the most successful at promoting forest cover (Min-Venditti et al., 2017).

However, claims to customarily managed forests will likely provide little control over the rivers that are crucial for local livelihoods, with forest and mining concessions able to increase sediment loads and decrease water quality at will. Rural communities around the globe are highly dependent on forest resources, but do not always have secure access to the forestlands on which their knowledge, institutions and practices are based (Scherr et al., 2003). Responsibilities of stakeholders are not always clearly defined to ensure fair and locally controlled decision-making processes at ecoregional and watershed levels (Cohen and McCarthy, 2015).

As reviewed in Chapter 2, 'rights to water' and 'rights to forest' have evolved in various parts of the world in ways that reflect the local importance of collective action for water quality and flood protection. Subsequent state institutions claimed forests primarily as a source of income for private actors (often connected to elites) and the state, with water-related concerns forming an addendum. Locally-developed ways of managing the forest-water-agriculture interface have gained recognition as traditional ecological knowledge (see Chapter 2).

The real question raised by these observations is how best to bring the knowledge, interests and rights of local communities into a forest-water governance framework, without at the same time endangering the delicate balance that must be established across potentially competing scalar dimensions, whether these encompass up- and downwind, or up- and downstream interests, or both. Faced with the mismatch of scales across social and ecological systems, the concept of landscape governance was introduced in the early 2000s. Landscape governance emphasises the multi-scalar and multi-stakeholder nature of environmental decision-making (Görg, 2007; Beunen and Opdam, 2011; van Oosten, 2013; Ros-Tonen et al., 2014; Dawson et al., 2017). It reflects the recognition that forests and water are part of a social-ecological system (SES) (Ostrom, 2007; Ostrom, 2009), and acknowledges the dynamic and multi-scalar nature of both systems.

7.4.1 National Level Frameworks

Historically the primary rationale for government involvement in forests and water was national security. A shortage of masts for shipbuilding caused the British Navy to commission the first published English language study on forests (Evelyn, 1664), while keeping river deltas navigable was a primary concern in water management (van der Brugge et al., 2005; Grigg, 2005). Beyond that, the two policy domains diverged.

Forests and water have been historically developed as separate policy domains (Gibson et al., 2000; Saleth and Dinar, 2004; Arts and Buizer, 2009), with the possible exception of upper watersheds where slope stability is a common concern. Both policy domains have dealt with local as well as national policy challenges, including transport and security issues, but often in different ways and through institutions that have little incentive to work together (Ostrom et al., 2007).

Environmental issues were invisible to many, especially in policy-making institutions, until such institutions as environmental ministries were introduced, largely in the 1980s and later, although these were often underresourced. In many countries there is little connection between the legislative framework for forests and that for water, though some countries such as the UK and other European countries, have nonetheless managed to develop forest and water guidelines. Moreover, each is most commonly addressed by different ministries and also managed at different institutional levels. Water governance has historically distinguished between waters used as transport infrastructure, measures for flood control, irrigation, provisioning of drinking water and wastewater recycling. Such diverse issues are rarely handled by any single ministry. Most water management is addressed at lower levels of administrative authority. The European Union's Water Framework Directive (WFD – Directive 2000/60/EC of the European Parliament and of the Council) has, for better or worse, shifted lower level administrative management in many countries from the local level to higher level subnational regional authorities. On the other hand, these regional authorities have no hydrologic or forest-related jurisdictional definition.

Forests, on the other hand, tend to fall far more frequently within the authority of an individual ministry, most typically the Ministry of Agriculture, though occasionally they fall within the authority of a Ministry of the Environment, or a combined Ministry of Agriculture and the Environment. In Ethiopia, for example, forests fall within the jurisdiction of the Ministry of the Environment, Forest and Climate Change but water is the responsibility of the Ministry of Water, Irrigation and Electricity. In Austria, on the other hand, responsibility for both forests and water have been incorporated, along with other natural resources, into the new Ministry for Sustainability and Tourism created in January 2018. Canada also exhibits a similar composition bringing together natural resource management into a single ministry (Natural Resources Canada).

Perhaps the most important reason for a lack of integration between forest and water is that the dominant view of the impact of forests on water resources has remained focused primarily on the catchment and the demand-side functions that most water resource management agencies are required to fulfil. Thus, the predominant view has tended to be that forests use water and remove it from the hydrologic cycle. In this sense, forests are typically managed either for their economic benefits (harvested wood products and fuel supply), for their benefits as a watershed purification system (see e.g., Box 7.1 on the Adirondack Forest Reserve), or, as has been more and more common across different countries from the first to the second half of the 20th century, for their benefits as recreational and symbolic natural resources.

Box 7.3

Challenges of managing water and forest interactions at the mega-catchment scale: an example from the Nile Basin

The Nile River is the longest river in the world with a basin area of 318 million hectares covering about 10% of Africa. A fast-growing population of almost 300 million people depend on the Nile waters for their livelihoods and sustenance. Yet it is one of the most water scarce river basins in the world, and high pressures from rapid population growth and related expansion in agricultural demand risk sharpening transboundary conflicts over water. Climate and land use change impacts exacerbate these deep-seated tensions (Swain, 2011). The high spatial and temporal variability of rainfall across the basin results in highly variable water availability within the different sub-catchments and, as a result, complex institutional arrangements are needed if water is to be shared equitably between the riparian states. To support the institutional development needed to manage surface and groundwater in such a complex situation, the Nile Basin Initiative (NBI) was established with significant support from the World Bank, donor organisations, and from the riparian countries themselves.

Under the auspices of the NBI, efforts have been made to quantify some of the ways benefits have been shared between the riparian states. This has mostly been achieved through cooperative efforts in agriculture, energy generation, water management, irrigation schemes, and efforts in climate adaptation. In terms of energy generation, only 20% of the total basin hydropower generation potential of 33,024 MW has been developed, with 6,833 MW mainly generated in Egypt, Kenya and Sudan (NBI, 2012). Estimates suggest, however, that the combined GDP of the basin countries would increase by USD 15.59 billion if this potential could be realised (NBI, 2014; World Energy Council, 2013).

Cooperative efforts in watershed management across the basin could also result in increases in the value of benefits from agriculture. The introduction of trees in shelterbelts could protect valuable cropland, and in the villages of Argi, Abkar and Afaad, a 40 km strip of tree-planting could generate a net benefit of USD 2.2 million (NBI, 2007). On a broader scale, soil and water conservation could translate to an increase in crop value of USD 5.49 billion per annum (World Bank, 2009; NELSAP, 2012), and increased regional trade in agricultural produce could potentially generate an increase of USD 9.78 billion to the basin as a whole (NBI, 2014).

Agriculture is by far the most significant user of water in almost every country (see e.g., Hoekstra and Mekonnen, 2012), and access to agricultural water is often influenced by distorted power relations or corruption. Countries across the world have built large dams to support agricultural water use (as well as hydropower production), frequently causing massive population displacement and creating serious damage to forest ecosystems both during and after the construction phase. Furthermore, these dams may be financially supported by capital loans from international institutions. While the major beneficiaries of these dams are often large scale commercial farmers, the repayments of this capital may often have to be generated by the nation's taxpayers (Sullivan, 2006). A further problem arising from dam construction in forest areas is the increased incidence of vector borne diseases associated with land clearing, and pooling of water in rutted surfaces where heavy equipment is used for forest operations (Alves et al., 2002).

In order to be able to better address many forest-water interactions – for example the management of forested watersheds for clean drinking water, or flood moderation by better managing the extent of forest cover – it may be enough to improve governance structures at the national and sub-national levels. Increasing the relative degree of institutional and policy convergence across forest-water interactions by, for example, creating hybrid ministries to address integrated natural resource governance represents perhaps one of the more compelling models to emerge in recent years. However, when catchment-level forest-water interactions begin to merge into landscape level forest-water interactions, national-level institutional innovations may not be sufficient.

7.4.2 International Level Frameworks

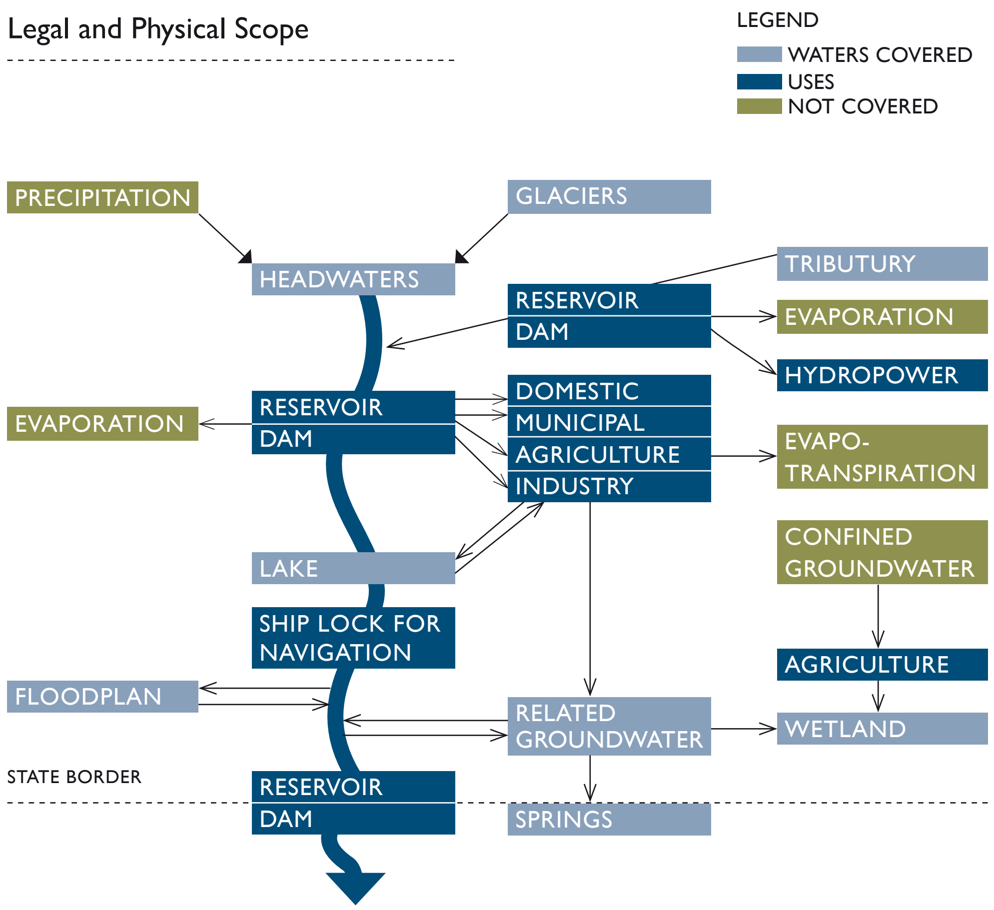
Most river basin agreements, along with institutional and state-level actors other than those representing local and regional basin-defined surface water flows, ignore the full water cycle. Likewise, the international legal framework that attempts to establish appropriate boundaries for what is covered under international water basin sharing arrangements is insufficient. As illustrated in Figure 7.2, the UN Water Convention, which intends to provide such an international legal framework, ignores the role and importance of evapotranspiration, regardless of whether evapotranspiration derives from intra- or extra-basin flows of atmospheric moisture. More generally, recognition of these types of forest water interactions has been slow to materialise (Dirmeyer et al 2009; van der Ent et al., 2010; Keys et al., 2012, 2017; Ellison et al., 2012, 2017).

Some of the water balance components in Figure 7.2 could be major limiting factors to future livelihoods and societal development, in particular the failure to measure and assess the impact of evapotranspiration, both at the local and the cross-catchment level.

A more recent dynamic view of the spatial dimension of the hydrologic landscape that moves beyond the framework of the catchment, raises the complexity of governance of the system to another level. When it comes to up- and downwind governance arrangements, there does not seem to be a single international integrated water management framework that has thus far managed to go beyond the inclusion of the riparian countries bordering the catchment in question, or that has managed to include countries that are the principal sources of the evapotranspiration that falls in a given basin as precipitation. Improved understanding of the consequences of the spatial organisation of land use

Waters covered (and not covered) under the UN Water Convention Framework

Figure 7.2



Source: Rieu-Clarke et al., 2012

practices and the role these play in the production and total available amounts of atmospheric moisture is crucial to our ability to make better use of this option.

We are only aware of one international agreement that recognises and attempts to constrain the potential for countries to interfere in the atmospheric hydrologic cycles of other countries and that is the Convention on the Prohibition of Military or Any Other Hostile Use of Environmental Modification Techniques (the Environmental Modification Convention, or ENMOD for short). Signed in 1977, it was primarily designed to prohibit countries from interfering in the weather of other countries under conditions of war (the initial complaint involved the US' use of cloud seeding to increase rainfall in specific target areas during the Vietnam war). The actual convention is not limited to acts of war and incorporates all relevant environmental modifications that can have a 'hostile' impact on environmental outcomes in other countries. The Convention on Biological Diversity, developed in 1992 and which came into force in 1993, also bans some forms of weather modification, or geoengineering. Finally, the focus on the 'long-distance effects' of pollutants in the Convention on Long-Range Transboundary Air Pollution (see Box 7.2), likewise provides a potential framework for future discussion.

7.4.3 From Catchment to Landscape – On the Configuration of Regional and Transboundary Institutions

Recent publications (Dirmeyer et al., 2009; Keys et al., 2017; Ellison et al., 2017) highlight the failure to consider up- and downwind sources of atmospheric moisture, in particular in arrangements that attempt – sometimes very explicitly – to regulate the amounts of water used by individual countries along a river basin, as a cause for concern. As demonstrated, in particular, by the case of the West African Rainforest and Ethiopian Highland atmospheric teleconnection (see Box 7.4), the availability of waters in the Nile River basin are potentially influenced by changes in land use practice in the Tropic forest belt across the West African Rainforest and the Congo Basin. This is all the truer in situations where high rates of deforestation threaten to alter important land-atmosphere interactions and the supply of atmospheric moisture (Nobre, 2014; Lawrence and Vandecar, 2015).

Regional and transboundary commissions (Box 7.5) have been established to deal with water governance in some of the more important transboundary basins. However, even at the catchment scale, these integrated water basin management frameworks face challenges. For example, in the case of the Nile Basin Initiative (NBI) most of the more important agreements are currently signed separately, either between the major downstream countries (Egypt and Sudan), or between the principal upstream countries (Burundi, DR Congo, Ethiopia, Kenya, Rwanda, Tanzania and Uganda). Though negotiations continue, attempts to bring these two sets of countries together have thus far failed to yield more encompassing agreements that would permit an adequate reconciliation of potentially competing demands over water rights and access (e.g., Salman, 2017; Yihdego, 2017 and also Boxes 7.3 and 7.4 on the transboundary Nile Basin arrangement).

Despite the fact that catchment-level transboundary governance institutions still require significant effort to successfully govern the entire precipitationshed, significant reform of the existing Nile Basin Initiative would be necessary to encompass both the catchment countries and the precipitationshed countries, which include the West African Rainforest and the Congo basin areas. Such a broader governance perspective may well be necessary to successfully manage up- and downwind flows of atmospheric moisture, in particular in the context of persistent and progressive climate change, but also, more generally, in the context of rapid population growth, rising food demand, increasing agricultural production and progressive deforestation.

Box 7.4

West African Rainforest teleconnections to an African water tower

About 85% of the surface water reaching Egypt originates from less than 10% of the Nile River Basin's total area: the Ethiopian Highlands. Much of the precipitation falling on these highlands originates as atmospheric moisture transported from the Indian and Atlantic Oceans as well as from the West African Rainforest (WARF; Viste and Sorteberg, 2013). Though concerns remain about how accurately these sources can be apportioned, evapotranspiration from the WARF provides an important contribution to rainfall in the Ethiopian Highlands and Blue Nile Basin areas. The WARF also influences the weather patterns bringing atmospheric moisture to the highlands. Changing land use in the WARF, especially deforestation, and associated changes in atmospheric transport patterns and regional climate will influence future rainfall patterns over the Ethiopian Highlands. This has major ramifications for subsistence farming in the highlands and regional food security, as well as for livelihoods further downstream along the Nile. Transboundary negotiations over water resources in this international basin ignore the importance of the WARF. Negotiations about water and food security in the Nile Basin should ideally move beyond transboundary discussions to include transregional governance, with an eye to the sources of the precipitation that provide the lion's share of the Nile waters beyond the basin. Ellison et al. (2017) and Keys et al. (2017) are two of the first papers to think through the implications, and enormous challenges, of managing such teleconnections. *Source: Gebrehiwot et al., 2018*

Box 7.5

Transboundary river basin management

At all geospatial scales, transboundary rivers and forests provide challenges for management.
Progress on transboundary water governance at the global scale has been slow. Even though the 1997 UN Convention on the Law of the Non-Navigational Uses of International Watercourses entered into force in 2014, there is no guarantee that the majority of the world's transboundary riparian states will adhere to it. More recently, the UN Water Convention has been seen as being more effective (UNECE, 2013), but there is still no global agreement on how transboundary rivers can effectively be managed for the equitable sharing of benefits from the river system (including groundwater). The Greater Mekong catchment, for example, contains six countries, all with very different governance approaches to both forest and water management. In spite of the fact that a Mekong River Basin Commission was set up three decades ago, full integration of how water is managed across the basin has yet to be achieved. As a result, land degradation and deforestation rates have been dramatic, with far-reaching consequences for the whole region. Between 1990 and 2015, some 5% of forest cover has been lost across the basin, with a corresponding loss of ecosystem services.
In response to this, initiatives are now being undertaken to address this governance challenge through the formation of 'Voices for Mekong Forests', a multinational collaborative effort by non-state actors. In this EU-funded project, the interests of 85 million forest dependent people, (including some 30 million indigenous people), are being addressed, in specific transboundary forests in Lao PDR, Myanmar, Thailand and Vietnam (Dahal et al., 2011; RECOFTC, 2017). This is to be achieved through the establishment of a regional 'Forest Governance Monitoring System', and capacity development for regional non-state actors to enable them to play a more meaningful role in forest governance in the Greater Mekong Basin.

International and transboundary aquifers likewise pose very similar problems of international management and coordination (see e.g., Gleeson et al., 2012). Moreover, the explicit role of forests in promoting recharge, or the role of deforestation in explaining aquifer retreat and loss, have, at best, been inadequately explored. The work of Ilstedt et al. (2016), however, suggests these issues deserve more attention.

Blue Nile falls in Tis Abay, Ethiopia
Photo © iStock: Joel Carillet

7.4.4 Actors versus Institutions and the Problem of Agency

It is opportune to consider which factors are most likely to support and strengthen the likelihood that actors will act in the interest of the general public and broad communities of interest, as well as to support and promote more innovative knowledge generation systems. Such factors are perhaps best explained by the existence of strong civil society organisations and effective educational systems but are also potentially the result of more polycentric institutions that favour 'shared governance' over reliance on single individuals and/or political parties.

Similar sets of questions can be directed at the behaviour of the private sector. The private sector has no immediate public mandate and has the explicit goal of defending economic interest, the profit-motive and the ideal of personal gain. Thus, since the goals of corporate entities are primarily profit-driven, they do not have any strong inclination to serve either the public interest, or the interests of sustainable natural resource governance except insofar as they rely on natural resources for their business. The role of the private sector , however, is increasingly central in the governance of natural resources.

Exploring the factors that drive corporations to internalise the externality costs of ecosystem damage and destruction may provide important insights into potential opportunities for mobilizing the corporate sector into positive action on natural resource governance. In many cases, this can happen because corporate actors may

Box
7.6

The responsible corporation

Corporations that are directly or indirectly dependent on forest-based commodities have a significant role to play in climate security through responsible action in their operations and supply chains. If not addressed, businesses face significant social, environmental and economic risks that will impact the reputation, the operations and ultimately the expenditure of their business (FAO, 2017). These risks unfold due to deforestation impacting on ecosystem infrastructure and services, causing biodiversity and habitat loss, greenhouse gas emissions, disruption to water cycles, soil erosion and social conflict (CDP, 2016, 2017). The Carbon Disclosure Project provides a platform for businesses to publicly disclose their efforts to reduce their impacts relating to carbon issues. Another mechanism is corporate social responsibility (CSR) strategies, which include clean development in support of the United Nations Framework Convention on Climate Change (Aggarwal, 2014).

Initiatives such as the New York Declaration on Forests, the Consumer Goods Forum and the Banking and Environmental Initiative, are drivers for zero-net deforestation through reforestation. Such initiatives – and those of certification and procurement standards such as the Forest Stewardship Council (FSC), the Roundtable on Sustainable Palm Oil (RSPO), Naturland, and the Rainforest Alliance Certified Coffee Farms – attempt to eliminate deforestation from supply chains.

The Paris Agreement on Climate Change and the Sustainable Development Goals (SDGs) are fairly new international mechanisms that are further driving businesses to commit to zero deforestation. While such mechanisms are driving governments to regulate unsustainable practices, it is the investors that are uniquely positioned to influence change by shifting financial commitment away from high-risk unsustainable business practices (CDP, 2017; FAO, 2017). Investors are increasingly favouring companies with policies and supply chains that decouple commodity production from forest impacts, which also affects supply chains and producers.

Companies as diverse as Dell and IKEA have planted millions of trees as part of their corporate social responsibility programme. While the notion of planting trees is extremely appealing, it may not always be as valuable as it appears. Tree planting is one of the most contested issues in the climate policy debate (Carton and Andersson, 2017). Offset projects have been linked to land grabbing; the displacement of rural communities; the unequal distribution of, and access to resources; a particular propensity for corruption; and a range of deleterious environmental side effects (Böhm and Dabhi, 2009; Leach and Scoones, 2015). In some instances carbon offsetting projects have been criticised for displacing the burden of mitigation to some of the world's poorest communities while giving the richest countries – and those most responsible for climate change – the opportunity to avoid taking action themselves (Bumpus and Liverman, 2008).

actually depend to varying degrees on the provision of specific ecosystem services, or because they may have an interest in presenting a positive public profile (see Box 7.6). Moreover, at a global scale, private sector entities are increasingly held accountable for actions along the total value chain (Mithöfer et al., 2017). For many tropical commodities 'zero deforestation' pledges have become popular, but how these are defined and implemented remains to be seen (Pasiecznik and Savenije, 2017).

7.4.5 Multi-Level Governance, Polycentricity and Multi-Scalar Governance

In the last couple of decades, the central authority of environmental governance has migrated from its focus central governments to multiple geographical scales (from international to local) and now also encompasses a broader diversity of actors (from local communities to large multinational companies; Box 7.6). Four recent trends in environmental governance have been highlighted: decentralisation, globalisation, the increasing role of market and agent-focused instruments and cross-scale environmental governance (Lemos and Agrawal, 2006). We might add to this the phenomenon of cross-sectoral environmental governance in institutional frameworks that begin to create the potential for building upon interactions across important natural resource frameworks, in particular forests and water.

The most appropriate level for addressing environmental resource governance issues remains somewhat obscured in controversy (e.g., Ostrom, 2009). Many point to the relative advantages of the international level of governance as a framework for sending appropriate signals (e.g., through the setting of 'norms' or the establishing of treaties) to regional and national level governments (Frieden et al., 2016). The fact that related issues – such as the up- and downstream management of forest-water interactions – have, to some degree, already been addressed in international conventions may suggest the international governance pathway represents one possible strategy for incorporating up- and downwind land-atmosphere interactions into some kind of international agreement.

On the other hand, there are frequent calls for decentralisation, and for returning to more local levels of governance as a way of generating closer attachments to local needs, interests and expertise (see e.g., Colfer and Capistrano, 2005). The downside of larger scale governance at the regional, national, transboundary and transregional levels, is that local interests, needs and knowledge are frequently overlooked and usurped by power seeking interests at these larger and often more distant spatial scales. Participatory governance and increasing decentralisation represent two strategies that have been invoked in an attempt to encompass and incorporate greater involvement from the local level, ensure the recognition of local interest and rights, as well as to engender greater legitimacy for policy-making.

The downside of increasing expectations regarding decentralisation and even local autonomy are that the management of multi-scalar needs for both up- and downstream, as well as up- and downwind interests and concerns with respect to forests and water require institutional frameworks that are capable of coordinating across disparate groups that are spatially and geographically separated, sometimes by long distances (see e.g., the discussion of long-distance teleconnections in van der Ent et al., 2010 and van Noordwijk et al., 2014).

Tensions between more centralised and more localised governance frameworks are not new. They have long troubled the smooth functioning of social, economic and political systems. And they have only been exacerbated with the increasingly rapid emergence of globalisation and the diverse set of international governance frameworks to emerge alongside national level governance. In this regard, developing strategies that can successfully reach across these domains seems more important than delegating exclusive authority to one level of governance over the other.

Increasing and/or achieving reciprocity across different levels of governance, from the local to the national level (and ideally all the way up to the international level) is one of the principal goals of polycentric governance, which is based on greater degrees of power-sharing and participatory decision-making across multiple levels of governance (Ostrom, 2010a, 2010b). Political and institutional decision-making frameworks that makes it possible for groups to interact and coordinate their interests, without at the same time imposing excessive power either from the top-down or the bottom-up is likely to be better suited to managing both the desire for decentralisation, on the one hand, and the necessity of coordinating multi-scalar forest and water interests across spatially and geographically distinct regions.

The ideals of 'participatory governance' rest upon a similar set of principles. General guidelines for participatory governance models are widely available (see e.g., Fischer, 2010, or the work of the International Observatory on Participatory Democracy, https://oidp.net/en/about.php). These models emphasise and promote the advantages of inclusiveness in decision-making processes. The concept of polycentricity may however go one step further, since it opens up more questions about the locus of final decision-making authority and may extend more flexibility and reciprocity across the individual components of the polycentric system. But the concepts of reciprocity and general inclusiveness in the discussion and coordination of the issues of the day, in this case natural resource governance, are common to both.

Polycentric institutions of shared governance are also likely to reinforce the selection of other institutional and civil society features based on the ideals of polycentrism. Ostrom makes this argument herself when she writes; *"Polycentric systems tend to enhance innovation, learning, adaptation, trustworthiness, levels of cooperation of participants, and the achievement of more effective, equitable, and sustainable outcomes at multiple scales, even though no institutional arrangement can totally eliminate opportunism with respect to the provision and production of collective goods"* (Ostrom, 2010a).

These models provide a strong foundation for thinking about how to improve interactive reciprocity across different levels of government and society. States and national governments, however, lie in-between the local and international levels of governance and are typically vested with the right to act. Moreover, states possess all the appropriate trappings of modern governance (executive, legislature, judiciary) (Scheffer et al., 2009). Thus, whether or not such strategies are chosen will depend, not on idealised models of governance, but rather on the balance of interests and the evolution of political coalitions at the national level. Clearly not all states or national governments can or are willing to move in the direction of more polycentric forms of governance – witness for example the many calls for subsidiarity, even in the context of European governance. But to the extent this is possible, and is supported by broad political coalitions, it may provide the foundations for more balanced natural resource governance outcomes.

7.5 Governance and the Capacity to Act – Reforming Governance Systems

7.5.1 Knowledge of Environmental Systems

As illustrated through this assessment, our knowledge of the ways in which environmental systems, including forest-water interactions, function, is reasonably well advanced, despite the fact that not all aspects of these interactions are all that well accepted. However, the extent to which we have progressed with the integration of forest-water interactions in the general policy framework is far more limited. And continued disagreement regarding some aspects of forest-water interactions has not simplified this process.

It is more important , therefore, to turn our attention to relevant policy frameworks that can potentially be used for setting some of these goals into action.

7.5.2 Models for Action

Many measures can be undertaken without a significant amount of institutional reform. Thus, for example, the promotion of forested watersheds for the provision of clean drinking water, or the reforestation of flood prone landscapes. Text boxes 7.7 and 7.8 provide other meaningful examples of measures that individual countries have undertaken without the need for significant institutional reform.

On the other hand, for other concerns, more significant institutional reform may be required. Thus, for example, the merger of ministries that integrate natural resources into a single institution (or ministry) represents a far more significant reform that requires significant legislative and/or executive effort and preparation. On the other hand, the advantages that may arise out of such mergers may well be worth that time and effort. It will thus be interesting to follow the experience of those countries that have undertaken such such strategic shifts in behaviour.

At the same time, it is important to recognise that not every country is prepared, nor has the political will to undertake such transitions. Certainly, the transition to more polycentric forms of governance, or to democracy, represent even more considerable evolutions that not all countries can adequately manage. And, as the eco-compensation model in China illustrates (see e.g., Ouyang et al., 2016; Leshan et al., 2018), the transition to democracy, let alone to more polycentric forms of governance, may not necessarily hold the only key to successful environmental and natural resource governance. In this regard, first environmental principles and adequate knowledge of environmental systems can potentiatlly trump the adequacy of governance institutions. But, on the other hand, arbitrary rule, dependence on the will and whim of the rulers, may leave such systems prone to future failure.

7.5.2.1 Instruments and Incentives for Forest-Water Governance

One of the suggested models for integrating the interests of different and potentially competing groups is represented by the market-based instruments (MBIs) and PES models discussed in more detail in Chapters 5 and 6. These strategies generally illustrate a set of principles concerning the potential governance of forests (and water) that may be useful in beginning to define a pathway for achieving reciprocity across multiple governance layers, as well as regions and differentiated spatial locations. What is uniquely interesting about these arrangements is that they allow for some degree of local self-governance and management, within a larger, multi-scalar and geographically dispersed cooperative and coordinated framework.

At the same time, however, these models are also being contested for their potential risk of inducing nature commodification (Gómez-Baggethun, 2014; see also the discussion in Box 7.6) and contributing to changes in values or mind-sets relating to environmental protection, changing conservation logic "*from moral obligation or community norms towards conservation for profit*" (Rode et al., 2015). Whether or not this is a bad thing, remains to be seen. On the one hand, without valuation, it is much simpler to usurp the provisioning power of ecosystem services for singular interests and purposes. On the other, with valuation, it may be easier to guide this provisioning power of ecosystems more in the direction of services in the interest of public and human welfare. Without such supporting framework, the transition away from defending purely economic interests may not always be possible.

As noted in Chapter 6, the typical market-based instrument and PES models involve performance-based payments that generally tend to be 'conditional' on the delivery of ecosystem services or on the actions that are supposed to deliver those services. These payments are also expected to provide 'additionality', i.e. go beyond what would be delivered in the absence of the scheme. Governments generally agree to organise the provision of these services because they would not otherwise occur in market systems. These strategies remain market-based, however, in the sense that stakeholders are paid for the contractual fulfilment they agree to provide (see e.g., Martin-Ortega et al., 2013; Porras and Asquith, 2018).

Box 7.7

Working for Water (WfW) in South Africa - An example of an innovative multiple benefits approach

South Africa and, indeed, the African continent more broadly, has a long history of attempts to deal with problems directly and indirectly related to invasive alien species (see, for example, IPBES 2018). The case of the Working for Water (WfW) programme in South Africa provides us with a useful example of a management approach that has tried (with acknowledged limitations) to focus not simply on one objective, but to take a positive synergies approach and yield benefits in a range of areas. Established in 1995, and currently managed by the Department of Environmental Affairs, WfW has worked on clearing alien invasive species with the intention of improving ecosystem services, including water provision, while also focusing on job creation and the broader objectives of land management.

A 'WfW' team at work in Limpopo Province, South Africa. Photo © Jane Furse

Van Wilgen et al. (2013) found that the programme had reduced invasion with regard to some species, but not all, finding that invasions had become more of a problem in many biomes. By 2013, WfW had spent approximately USD 457 million on the control of alien invasive plant species (interestingly, two invasive species combined account for just over a third of the expenditure). Given the mixed success, a more focused approach was recommended, with more funding redirected to support biological control, where success rates have been higher. In this way, WfW provides an imperfect, but useful example of a management approach that attempts to yield results across a range of sectors, focusing on alignment where possible. For more information, see also Marais and Mlilo (2018)..

Box 7.8

'Thanks to the Forest, We have Water' youth perspectives on community-forest-water linkages

The Future of Forest Work and Communities project engages forest youth from around the world to share insights and ideas about community, territory, rural versus urban life, forest values, and forest work and governance. Multi-day 'visioning' workshops have been held, or will soon be held, in Bolivia, Canada, Ecuador, Guatemala, Indonesia, Mexico, Mozambique, Nepal, Peru, the Philippines, Tanzania and Uganda.

Water was not a theme that explicitly informed workshop activities, yet water has repeatedly been raised by participants as a key issue. In four of eight workshops, clean drinking water was among the most important benefits of village life, with 'access to clean water' a major reason why youth may choose to stay in their communities. While access to clean water was a major pull factor for communities, water scarcity and water contamination were among the key drawbacks associated with city life. Youth also perceived a clear connection between water availability and forest stewardship. When asked why forests were important, youth at every workshop talked about the role forests play in 'providing' or 'purifying' water. In Poplar River First Nation, Canada, a youth stated it was *"important for everyone in the world to have forests … for water, oxygen… we want to support them [the forests] for our kids, for the future"*. In Intag, Ecuador, water was the main reason for restoring its cloud forest – *"Why plant trees? Because they give us WATER!"* When youth discussed forest work opportunities, water remained centre stage. In five workshops, youth developed project ideas based on locally-sourced water, including community water bottling plants (Bolivia, Mexico and Nepal), irrigation infrastructure (Bolivia) and water purification systems (Canada).

While many consider life outside of their communities, this work is showing just how connected these young people are to territory and the forests they still call home. Water plays a fundamental and increasingly important role in these place-based relationships. As actors work to improve community-based forest management, community-based applications of REDD+, and other PES projects, it is vital that they understand such perspectives. After all, it is these young people who will shape local community capacities to lead future forest strategies.

(See: http://pilot-projects.org/projects/project/the-future-or-forest-work-and-communities)

In this basic model, several features appear to be key:

1) Some role for the higher-level assessment and recognition of ecosystem gaps is necessary. Without government intervention, these gaps are presumably less likely to be recognised and action less likely to be undertaken.
2) These strategies typically involve more or less formal contracts between governments and various stakeholders for services rendered.
3) As long as the ecosystem or related services are provided (performance-based strategy), payments are typically made to the providers of these services.
4) In many cases the provision of ES depends on the maintenance or adoption of certain land use/management frameworks can potentially deliver nature-based services important for general human welfare. This is common in the case of water-related services, as the performance (output in terms of the actual service:

water yield or improving quality is not monitored, but land use/management changes are).

Though most PES models are based on some degree of knowledge about forest-water interactions, this knowledge is at best imperfect and often competing views about the viability of forests for promoting water availability are prevalent. The vast majority of PES for water services provided by forest are established to address up- and downstream dynamics at the catchment level (Martin-Ortega et al., 2013). Inadequate attention is paid to the up- and downwind framing of forest-water interactions (where most supply-side, precipitation-recycling is likely to have its principal impact).

Many of the initial signals for the establishment of agenda-setting principles and potential projects designed to mobilise and promote ecosystem services often have an initial spark as government plans. International agenda-setting on the goals of integrating forest and water interactions into the general climate change adaptation framework is important because of the signal it sends to national governments, as well as the many stakeholder organisations. At the same time, it is becoming increasingly important to recognise the value of monitoring and assessing the outcomes and viability of these ecosystem-based strategies (see e.g., Taffarello et al., 2017). The Forest and Water Programme at the FAO is also currently working on the development of such a Forest and Water Monitoring Framework.

Real, performance-based systems are hard to achieve (van Noordwijk et al., 2012) as they require high quality and fine-grained data on carefully selected metrics (van Noordwijk et al., 2016; Lusiana et al., 2017). While progress is being made to disentangle the combined effects of climate variability and change, and land use change on streamflow in specific landscapes (Ma et al., 2014), most ‘performance-based’ schemes will for the foreseeable future rely on ‘land use proxies’ for the desired ‘ecosystem services’.

At the same time, many of these performance-based schemes exhibit positive outcomes. Min-Venditti et al. (2017) highlight the fact that both PES (88% of cases) and community-based management strategies (81% of cases) have had strong positive impacts on increasing forest cover, in particular in Mexico and Costa Rica. Whether PES systems have proven capable of addressing questions of scalar mismatch, however, has generally not been assessed. The Min-Venditti et al. (2017) study, for example, does not consider the impacts of such strategies on forest-water relationships – though clearly such a research programme could provide new terrain for the analysis of PES and reforestation programmes more generally.

7.6 Research Gaps and Future Priorities

The transition to a forest and water management framework that manages to successfully integrate forest and water interactions and, in addition to up- and downstream relationships, is able to encompass up- and downwind forest-water relationships, is necessary, but is likely to be the cause of some conflict and controversy. The challenges of increasing water scarcity and progressive and persistent climate change, not to mention additional contextual factors related to rapid population growth, etc., require us to identify strategies that can help facilitate adaptation to, and ultimately mitigation of, climate change through mechanisms that will help to preserve existing forest cover and perhaps even go beyond.

Institutionally-driven decision-making frameworks that are large enough in their membership and representation to physically encompass the geographic spread of such ecological relationships are far more likely to be able to address up- and downwind relationships. A focus on the catchment is inadequate, since this framework has typically led institutions and countries to ignore both the downwind impacts of local action, as well as the potential upwind contributions to the local water regime. In order to bring these relationships into the general discussion of forest-water and hydrologic relationships, there is a need to extend the geographic coverage of such institutional negotiation and decision-making frameworks.

The relative importance of finding ways to further encourage integration of the larger scale hydrospace perspective into the general framework of policy output and decision-making on forest and water issues can no longer be ignored. The livelihoods of millions of people may well depend on how well individual countries and larger regions are able to manage these larger scale relationships. However, there is still much to be learned. In particular, it would be helpful to greatly improve our knowledge of when and where additional forest cover can help intensify the hydrologic cycle. Though we think of this general relationship as a universal principle, there are likely to be important differences across biomes that have not been adequately considered.

A shift toward policy objectives that increasingly incorporate the knowledge-base provided by the current literature on forest-water interactions can significantly impact human welfare. Thus, benefit sharing and uneven distributional impacts, both in the water and forestry sectors, as well as across geographic landscapes, have to be carefully examined if new strategies are to be developed towards greater cross-sectoral and multi-scalar, cross-regional harmonisation.

The ability of governments and more international decision-making frameworks to adapt to these emerging concerns may well depend on their ability to devise appropriate discussion and decision-making structures and/or institutions. This may involve the elaboration of institutions capable of addressing forest and water issues simultaneously and in concert (as opposed to in separate institutional ‘silos’), or it may involve the elaboration of negotiation frameworks that are capable of spanning not only the catchment, but also the precipitationshed.

To the best of our knowledge, no existing PES schemes or governance frameworks reflect the emerging broader understanding of forest-water dynamics. A next step in the MBI and PES discussion would be to try and classify existing forest-water strategies into different categories and to assess their effectiveness based on where they fit within this general framework, i.e. whether they are designed to address only the catchment or attempt to mobilise larger-scale (beyond-the-basin) visions of the hydrologic landscape, similarly to what has been achieved with international/global carbon-credit and REDD+ schemes. Another potential next step may be to begin proposing PES and MBI schemes based on the supply-side model.

7.7 Conclusions

A mismatch exists across ecological and administrative scales, generating challenges for the management of transboundary forest and water resource systems. A further mismatch occurs within national scale governance contexts, especially in federal governance systems where responsibilities for forests and water are typically shared between central ministries and administrative bodies as well as provincial and municipal level counterparts nested within an administrative hierarchy.

Not all countries are in a position to optimise their existing institutional and political frameworks. In this regard, the failure to arrive at more optimal solutions may be dictated by the inadequacies of the existing political and institutional frameworks. In such systems, the only recourse may be strong social and civil action in order to overcome persistent barriers to successful natural resource governance. As Andersson and Ostrom (2008) note:

> *"there is no guarantee that such [polycentric] systems will find the combination of rules at diverse levels that are optimal for any particular environment. In fact, one should expect that all governance systems will be operating at less-than-optimal levels given the immense difficulty of fine-tuning any complex, multi-tiered system"* (p.78).

The governance of, and co-investment in, water and forests as resources can be improved to reduce the identified hydro-vulnerability in the context of all SDGs, and the persistent and growing threats arising from climate change. Failure to place water at the centre of discussions on forest – climate interactions and diverse forestation strategies, will have important negative impacts on policy effectiveness and ultimately on the provision of water.

Governance frameworks play a key role in the potential optimisation of natural resource management. Moving from an emphasis on decentralisation to one that addresses flexibility and balanced interaction across multiple levels of governance (polycentrism) is more likely to ensure outcomes that are able to address concerns central to the management of larger scale landscapes (as opposed to catchments). People must be respected as integral components of the forest-water interface, and policies to strengthen that interface must engage with them at all levels to ensure success. The challenge for polycentric governance is to balance top-down and bottom-up forces.

Models that increase the degree of shared governance and move away from dependence on single individuals or majorities may be more successful at providing positive natural resource governance. Likewise, such models may provide opportunities for reconciling interests in decentralisation and relative local autonomy (subsidiarity) with the simultaneous need for more regional and cross-national coordination of policy goals.

Cloud forests in Rincón de la Vieja National Park in Costa Rica
Photo © iStock: PobladuraFCG

Market-based instruments in environmental management are part of new public-private partnerships involving non-state actors taking responsibility for resource governance. Moreover, this type of institutional structure presents opportunities for the coordination of up- and downwind, as well as up- and downstream interests and concerns. The framing of rights and obligations, however, remains a sensitive issue.

Institutional frameworks that have been set up to address transboundary concerns need to be re-constituted and reformed to be able to address both up- and downwind, as well as up- and downstream forest-water relationships. This is further likely to extend the geographic purview of such institutional frameworks due to the requirement of bringing together locations that are the providers of atmospheric moisture, with basins where that atmospheric moisture contributes to potential rainfall.

International governance plays a highly important, symbolic and substantive role by creating norms (such as the SDGs), and providing fora in which these norms can be discussed, negotiated and agreed upon. National level governance can also be radically improved, in particular, by beginning to bring together competing sectors of the economy into national level institutional frameworks that encourage cooperation and negotiation across the broader scope of forest and water interactions.

Strategies that can assist governments and NGO actors to move beyond the dominance of entrenched interests are important for shifting policy goals away from more profit-oriented and toward more sustainability-oriented strategies, policy building and policy learning. Market-based instruments and PES schemes may provide one, though certainly not the only, model for moving forward.

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Chapter 8
Forest, Trees and Water on a Changing Planet: How Contemporary Science Can Inform Policy and Practice

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Trees and people both need water. With a growing global population and continued forest loss and degradation – a key question becomes: are trees and people competitors or friends? The relationship between forests, trees and water is an issue of considerable complexity and uncertainty, but of high priority for both people and the environment. In the face of such challenges, the next generation of policymakers and decision-makers will have to consider climate-forest-water-people interactions in a more holistic way. Water may be the key to unlocking policies that flow from a local understanding to actions at global scales.

In the forestry community, it is still largely assumed that only forest authorities are in a position to provide the water required by society. Yet, the combined effects of climate change and climatic variability, modification of forests and increasing demand for water suggest that more explicit attention should be directed at managing trade-offs between forests, water and people. Managing these trade-offs is particularly important in multifunctional landscapes that include forests and trees.

This GFEP assessment focused on three key questions:

1. *"Do forests matter?":* To what degree, where and for whom, is the ongoing change in forests and trees outside forests increasing (or decreasing) human vulnerability by exacerbating (or alleviating) the negative effects of climate variability and change on water resources?
2. *"Who is responsible and what should be done?":* What can national and international governance systems and co-investment in global commitments do in response to changes in water security?
3. *"How can progress be made and measured?":* How can the UN SDG framework of Agenda 2030 be used to increase the coherence and coordination of national responses in relation to forests and water across sectors and from local to national and international scales?

This report provides a global assessment based on relevant scientific evidence and established and emerging scientific concepts. There has been substantial progress in the past decade in the understanding of more narrowly delineated subsystems within the forest-water system. However, the GFEP Panel on Forests and Water recognised that comprehensive answers to the above three questions would vary depending on the region of focus and involvement of regional stakeholders and would require time and resources well beyond the scope of this report.

Our conclusions and their implications (Table 8.1) are intended to inform relevant international policy processes such as the 2030 Agenda for Sustainable Development and related SDGs. The Bowtie Risk Management Assessment Tool inspired the structure of this GFEP assessment report – with individual steps that linked: 1) determinants of change in the forest-water relationship, and drivers of forest and land use change (Chapter 3) to 2) pressures on ecosystem structure and 3) changes in ecosystem functions (Chapter 4), which 4) affect ecosystem services and the people benefitting from them, now and in the future (Chapter 5), 5) leading to a range of prevention controls to reduce pressures or mitigation controls to reduce or adapt to impacts (Chapters 6 and 7).

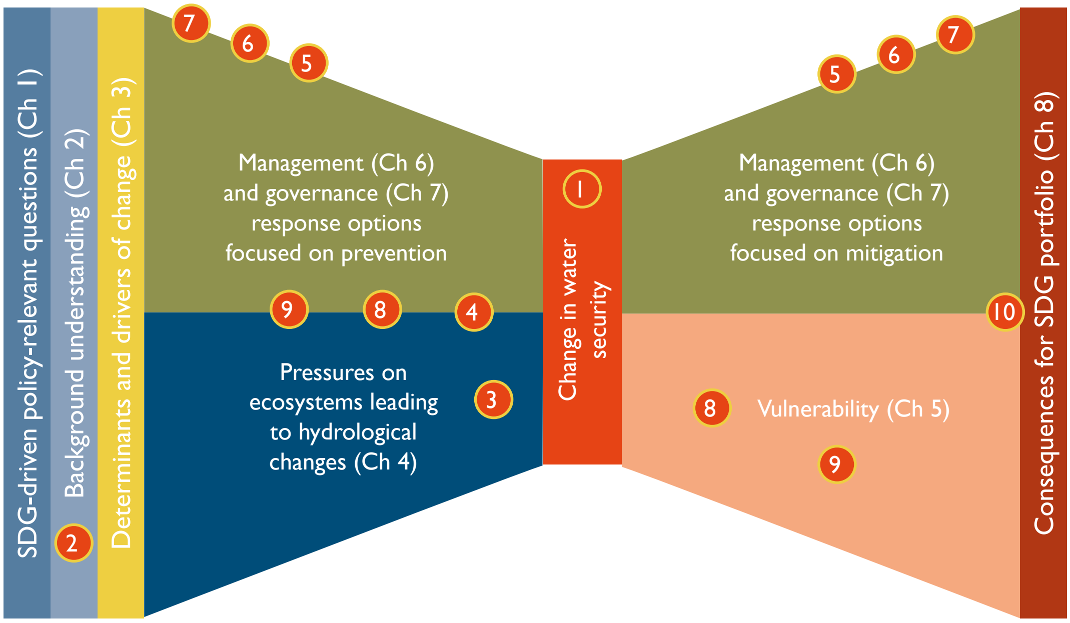
In Figure 8.1 we link the conclusions and implications back to these individual steps.

Conclusions and their implications for decision-makers Table 8.1

Conclusions	Implications
1. Water is central to all 17 SDGs and ambitions.	Governments and other stakeholders that want to achieve the SDGs need to understand the centrality of water and its relations with social, environmental and economic outcomes. Increasingly, it is recognised that SDGs cannot be dealt with individually.
2. A systems approach to climate-forest-water-people relations that integrates hydrological processes and their interactions at all scales is needed.	Limited public understanding of complex ecosystem interactions prevents rational decision-making and can lead to unintended consequences.
3. Forests, especially natural forests, contribute to the resilience of water supply for humans in the face of global change.	Investments in the preservation of existing native forests are needed as part of a multiple disaster prevention strategy, as well as to improve resilience in the face of increasing risk.
4. Forests can be managed for resilience of water supplies to enable adaptation to change if locally relevant data and resources are available.	Investments in data collection and interpretation are essential to support evidence-based risk management planning and adaptation.
5. Multiple water-related objectives across the portfolio of SDGs present new challenges for policymakers and managers of forests and landscapes with partial tree cover.	New institutional responses are needed to tackle multiple water-related objectives across the portfolio of SDGs, taking a multiple benefits approach.
6. International and regional institutional and governance frameworks can play a key role in optimising climate-forest-water management.	New or improved levels of collective action and coordination are needed, including those that coordinate across sectors and across spatial scales.
7. A clear policy gap in climate-forest-water relations exists, waiting to be filled.	Forest-water relations deserve at least as much policy attention, from local to global scales, as forest-carbon relations.
8. Regulations and rights-based approaches to climate-forest-water relations provide an essential foundation for innovation in forest-water governance.	Incentive-based mechanisms present opportunities for coordination of interests and concerns in climate-forest-water management but must respect the rights of local, indigenous and other vulnerable communities.
9. To successfully achieve SDGs, social and environmental justice, along with equity targets, must be integrated into climate-forest-water policies and management strategies.	Already marginalised and vulnerable communities should not be exposed to further risks; opportunities to improve community health and well-being need to be explored when developing forest-water adaptive management strategies.
10. The global nature of the current assessment limited the scope to be quantitative and geographically explicit.	More quantitative regional-scale case studies that include atmospheric relations, surface and groundwater flows are needed that can be extrapolated to other areas with different social and economic conditions.

The ten concluding points mapped onto the Bowtie Risk Management Assessment Tool used in structuring this GFEP report

Figure **8.1**



Source: Authors' own elaboration

1. Water is central to all 17 SDGs and ambitions. Governments and other stakeholders that want to achieve SDGs need to understand the centrality of water and its relations with social, environmental and economic outcomes. Increasingly, it is recognised that SDGs cannot be dealt with individually.

Water is central to the United Nations' (UN) 17 Sustainable Development Goals (SDGs) and to global prosperity as a whole. Eight SDGs require an increased supply of safe, secure and reliable water. Six SDGs address social justice and equity, and their attainment will reduce injustice and inequity in access to forests and water. The remaining three SDGs build and maintain an ecological infrastructure that support the other 14 SDGs by adapting to climate change and securing the integrity of the terrestrial and aquatic parts of the planetary system. It is increasingly clear that the SDGs cannot be dealt with individually. Instead, a multiple benefits approach is necessary, and this is particularly important in climate-forest-water-people interactions that comprise the focus of this assessment.

Water scarcity will inevitably increase in the future, as climate variability and change generate uncertainties in water supply, while a growing human population increases demand for water. Forests and forested landscapes regulate the provision of water and water-related ecosystem services. The majority of the estimated four billion people facing insufficient access to clean water live in areas with low forest cover, and most of them depend on engineered infrastructure that redistributes water across watershed boundaries. Preservation of existing native forests and better-informed management of planted forests, are especially critical in areas with low forest and tree cover. Effective decision-making mechanisms that help to resolve transboundary water conflict and promote shared benefits of water-sharing are necessary.

2. A systems approach to climate-forest-water-people relations that integrates hydrological processes and their interactions at all scales is needed. Limited public understanding of complex ecosystem interactions prevents rational decision-making and can lead to unintended consequences.

A century of science has taught us that forests process water and this water becomes a source for people downstream. Governments and other stakeholders need to work together on global water governance to promote resilient and reliable upstream-downstream and upwind-downwind water supplies. Water is a local as well as a global resource and changing water supplies have cascading effects that no longer respect political and national boundaries. Climate change and climatic variability increase the hydrological uncertainty of the delivery of forest-water related ecosystem services, and, hence, the realisation and distribution of benefits that people derive from them.

3. Forests, especially natural forests, contribute to the resilience of water supply for humans in the face of global change. Investments in the preservation of existing native forests are needed as part of a multiple disaster prevention strategy, as well as to improve resilience in the face of increasing risk.

Natural forests improve resilience of water supply in the face of disturbance and climate change and climatic variability. Changes – both natural and anthropogenic – in natural forests may be undermining this resilience that cannot be fully replaced by tree planting efforts. Climate change and climatic variability and their impacts on natural forest health are reducing the already challenged capacity of forests to secure predictable water flows. Hence, preservation of existing native forests should be a priority in the face of changing climate and associated increased probability of extreme weather events.

4. Forests can be managed for resilience of water supplies to enable adaptation to change if locally relevant data and resources are available. Investments in data collection and interpretation are essential to support evidence-based risk management planning and adaptation.

Hydrological effects of forest disturbance, forest conversion and forestation can be understood through the changes in four 'ecosystem structure' descriptors of forests: leaf area index, effective soil cover, soil macroporosity (infiltration rate) and rooting depth. The first can be managed by influencing stand density, the others may primarily be managed (given inherent soil properties) through tree species selection.

Generally, increased forest cover can be expected to have positive effects at local scales (including micro- and meso-climatic effects on temperature and wind speeds), reduced water yields at landscape scales in non-tropical regions, and positive effects downwind in some places at some times. Furthermore, the recovery of aboveground benefits is feasible within a few years, but recovery of belowground benefits (i.e., infiltration and recharge) is often a slower process, counted in decades rather than years. The type of forest cover that is feasible may be constrained by water availability, especially where targets are to be met by planting rather than by natural regeneration. Trade-offs exist between the magnitude and regularity of water flows and associated water quality. These trade-offs depend on the type, density and distribution of tree cover, and require location-specific assessment.

Additional research is needed to better understand the relative magnitude of the effects of climate change and climatic variability, the effects of changes in forest cover, and their interactions on seasonal and annual water yields.

5. Multiple water-related objectives across the portfolio of SDGs present new challenges for policymakers and managers of forests and landscapes with partial tree cover. New institutional responses are needed to tackle multiple water-related objectives across the portfolio of SDGs, taking a multiple benefits approach.

While a first group of SDGs (especially 1, 2, 6 and 7) implies increased demand for clean, regularly flowing water, a second group of SDGs (especially 5, 10, 12 and 16) implies a change in power-sharing that allows multi-stakeholder involvement, thus increasing the need for transparency and equity in decision-making. The third group of SDGs (13, 14 and 15) establishes targets for resource conservation and restoration that require location-specific scenarios in order to be relevant for local stakeholders rather than relying on generic expectations that all types of forest cover are good for all hydrological functions. Overall, the potential success in avoiding the trespassing of planetary boundaries critically depends on an increase in human adaptive capacity and the ability to transcend existing conflicts; as well as an ability to take a multiple benefits approach and realise positive synergies in addressing SDGs. Information beyond what is currently available is needed to optimise downstream and downwind water availability for the multiple objectives across the portfolio of SDGs.

6. International and regional institutional and governance frameworks can play a key role in optimising climate-forest-water management. New or improved levels of collective action and coordination are needed, including those that coordinate across sectors and across spatial scales.

International governance can play both a symbolic and a substantive role by creating norms (such as the SDGs), by providing fora in which norms can be discussed, negotiated and agreed upon, and by providing opportunities for assessing progress. Strategies that can assist governments and other policy and management entities to move beyond the dominance of entrenched interests and paradigms, including the ability to take a cross-sectoral approach, are important for shifting policy goals away from more profit-oriented toward more sustainability-oriented strategies, policy-building and policy-learning. Furthermore, governance systems with increased polycentrism (characterised by increased reliance on multiple centres of power and multiple levels of decision-making) may provide opportunities for reconciling interests in the decentralisation of decision-making with needs for national and international coordination of policy objectives. Greater reliance on the ideals of participatory and shared governance, as supported, in particular, by the model of polycentrism, may facilitate improved management of top-down and bottom-up forces, as well as to practically realise multi-level adaptive governance.

7. A clear policy gap in climate-forest-water relations exists, waiting to be filled. Forest-water relations deserve at least as much policy attention, from local to global scales, as forest-carbon relations.

The role of forests in current climate policy is defined by targets to reduce net greenhouse gas emissions and increase carbon storage. However, ill-defined local-scale efforts to increase carbon storage may reduce local water availability. It is essential to place water at the centre of discussions of forest-climate interactions in areas of water scarcity because carbon-centred forestation strategies will have important consequences on water resources.

8. Regulations and rights-based approaches to climate-forest-water relations provide an essential foundation for innovation in forest-water governance. Incentive-based mechanisms present opportunities for coordination of interests and concerns in climate-forest-water management but must respect the rights of local, indigenous and other vulnerable communities.

Market-based instruments are increasingly used as strategies to involve non-state actors in taking on the responsibilities of resource governance. Existing and potential future commitments to achieve deforestation-free product and value chains present opportunities for the coordination of up- and downwind, as well as up- and downstream interests and concerns. Such private-public partnerships are well-aligned with the idea of increasing shared governance and polycentrism but must maintain and enhance commitments to the rights of the most vulnerable groups.

9. To successfully achieve SDGs, social and environmental justice, along with equity targets, must be integrated into climate-forest-water policies and management strategies. Already marginalised and vulnerable communities should not be exposed to further risks; opportunities to improve community health and well-being need to be explored when developing forest-water adaptive management strategies.

Changes to the coupled climate-forest-water system will affect the delivery of related ecosystem goods and services and consequent development options. Impacts and consequences of these changes will not be evenly distributed geographically, socially or economically. Any new institutional arrangement should be sensitive to distributional concerns, as well as to social and environmental justice and equity. In particular, the rights of marginalised and vulnerable communities must be protected.

10. The global nature of the current assessment limited the scope to be quantitative and geographically explicit. More quantitative regional-scale case studies that include atmospheric relations, surface and groundwater flows are needed that can be extrapolated to other areas with different social and economic conditions.

A global assessment such as this one could not provide sufficient geographic specification of risks to forest-water relations and management options to reduce these risks. A series of regional/continental assessments, with broad involvement of all relevant scientific disciplines and sources of knowledge is needed to complement and extend the current global GFEP assessment.

Major knowledge and data gaps needed to be filled to inform these regional/continental assessments include the following:

- Specific characteristics of both native and managed forests (e.g., tree species, ages, densities, etc.) that contribute to sustained season and annual water yield, by geographic region.
- Specific locations of forested areas which are most important as sources of water to ecosystems and to downwind and downstream water users.
- Range of variability of forest water quantity and quality as a function of climate change and climatic variability across geographic regions.
- Comparison of changes in water quantity and quality across different land uses.
- Knowledge of how forests and the water that comes from these forests are perceived and valued by local people.

Appendix I
Glossary of Terms and Definitions

Above-ground biomass (AGB): All biomass of living vegetation, both woody and herbaceous, above the soil including stems, stumps, branches, bark, seeds, and foliage (FAO, 2004; IPCC, 2006).

Adaptation (in relation to climate change impacts): Adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities (Seppälä et al., 2009).

Adaptive capacity (in relation to climate change impacts): The ability of a system to adjust to climate change (including climate variability and extremes) to moderate potential damages, to take advantage of opportunities, or to cope with the consequences (IPCC, 2007).

Adaptive management: A dynamic approach to forest management in which the effects of treatments and decisions are continually monitored and used, along with research results, to modify management on a continuing basis to ensure that objectives are being met (IUFRO, 2005).

Afforestation: Establishment of forest through planting and/or deliberate seeding on land that, until then, was not classified as forest (FAO, 2010). According to the definition used by the UNFCCC, afforestation can take place on land that has not been covered by forest for at least 50 years. See also *Agroforestation*, *Forestation* and *Reforestation*.

Agroforestation: For the purpose of this report, defined as "Increase in area under *agroforest*".

Agroforest: Farmer-managed tree-based vegetation, which can include the management of remnant, planted and spontaneously established trees (Ordonez et al. 2014).

Agroforestry: Land use at the interface of agriculture and forestry (van Noordwijk, et al., 2016).

Atmospheric residence time: For the purpose of this report, defined as "Average time a water molecule spends in the atmosphere between *evapotranspiration* and *precipitation*".

Baseflow: For the purpose of this report, defined as "The complement of *peakflow* in the *streamflow* pattern as experienced at a specific point of observation, not responding directly to incoming *precipitation* (or snowmelt)".

Basin: Area having a common outlet for its surface runoff; land area contributing (blue) water to a river (WMO, 2012).

Below-ground biomass (BGB): All biomass of live roots. (FAO, 2004; IPCC, 2006).

Biodiversity [Biological Diversity]: The variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems (CBD, 1992).

Biomass: Live organic material both above-ground and below-ground, e.g. trees, crops, grasses, roots. Biomass includes the pool definition for above- and below-ground biomass (adapted from: IPCC, 2003; FAO, 2004).

Bioprecipitation: A feedback cycle linking Earth history, ecosystem dynamics and land use through biological ice nucleators in the atmosphere (Morris et al., 2014).

Blue water: Water in streams, rivers, lakes, reservoirs or groundwater stocks and flows (and as such the primary interest of hydrological science) (Sood et al., 2014).

Carbon cycle: The term used to describe the flow of carbon (in various forms, e.g. as carbon dioxide) through the atmosphere, ocean, terrestrial biosphere and lithosphere (IPCC, 2007).

Carbon sequestration: The process of increasing the carbon content of a reservoir/pool other than the atmosphere (IPCC, 2007).

Carbon sink: Any process, activity, or mechanism that removes a greenhouse gas, an aerosol, or a precursor of a greenhouse gas or aerosol from the atmosphere (IPCC, 2007).

Carbon: In this assessment report, except when referring to specific [carbon] stocks and fluxes, 'carbon' refers to the net balance of CO_2 and non-CO_2 greenhouse gas emissions and removals.

Deforestation: For the purposes of this report, defined as "Change from a forested to a non-forested state, which depending on forest definition, relates to both institutional and tree cover dimensions".

Ecological resilience: The ability of a system to absorb impacts before a threshold is reached where the system changes into a different state (Gunderson, 2000).

Ecosystem (or ecological) functions: For the purposes of this report, ecosystem 'functions' are synonymous with 'processes' and refer to all of the physical, chemical and biological actions performed by organisms within ecosystems. Some of these functions are ecosystem services, including production, pollination, nutrient cycling (e.g., decomposition, N_2-fixation) and carbon storage (MA, 2005) that directly benefit humans. Other examples include photosynthesis, predation, scavenging and herbivory.

***Ecosystem restoration*:** The process of managing or assisting the recovery of an ecosystem that has been degraded, damaged or destroyed as a means of sustaining *ecosystem resilience* and conserving *biodiversity* (CBD, 2016).

Ecosystem services: Benefits people obtain from functioning ecosystems. These include i) provisioning services such as food, water, timber, and fibre; (ii) regulating services that affect climate, floods, disease, wastes, and water quality;(iii) cultural services that provide recreational, aesthetic, and spiritual benefits; and (iv) supporting services such as soil formation, photosynthesis, and nutrient cycling (MA, 2005).

Emission: The release of greenhouse gases and/or their precursors into the atmosphere over a specified area and period of time (IPCC, 2003).

Evaporation: For the purposes of this report, defined as "Phase shift from fluid to gas form of water, not involving active control by plants; it includes evaporation at the soil surface plus water intercepted in waterfilms on aboveground plant parts".

Evapotranspiration: *Evaporation* plus *Transpiration*

Flow regime: For the purposes of this report, defined as "temporal pattern in the quantity of *streamflow* (or river discharge)".

Forest: Land with trees under specified management authority; common definitions combine biophysical aspects of tree cover ("Land spanning more than 0.5 hectares with trees higher than 5 metres and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ") with institutional aspects (excluding trees that are considered to be agricultural and/or land that is predominantly under agricultural or urban land use). It also includes areas temporarily unstocked, e.g. after clearfelling or disturbance, that are expected (without time limit) to revert back to tree cover above the stated threshold (FAO, 2004).

Forestation: In this report used as collective term for increase in forest area through *afforestation*, *reforestation* and *agroforestation*.

Forest conversion: For the purposes of this report, defined as "Clearance of natural forests for other land uses, such as plantations, agriculture, pasture for cattle settlements, mining and infrastructure/urban development. This process is usually irreversible."

Forest degradation: For the purposes of this report, defined as "The reduction of the capacity of a forest to provide (specified) goods and services".

Forest ecosystem: A forest ecosystem can be defined at a range of scales. It is a dynamic complex of plant, animal and micro-organism communities and their abiotic environment interacting as a functional unit, where trees are a key component of the system. Humans, with their cultural, economic and environmental needs are an integral part of many forest ecosystems. (http://www.cbd.int/forest/definitions.shtml [Accessed on 31 May 2018].

Forest fragmentation: Any process that results in the conversion of formerly continuous forest into patches of forest separated by non-forested lands http://www.cbd.int/forest/definitions.shtml [Accessed on 31 May 2018].

Forest management: The processes of planning and implementing practices for the stewardship and use of forests and other wooded land aimed at achieving specific environmental, economic, social and/or cultural objectives. Includes management at all scales such as normative, strategic, tactical and operational level management (FAO, 2004).

Forest services: *Ecosystem services* derived from forests.

Forests and tree-based systems: For the purposes of this report, this includes the spectrum from management of forests to optimise yields of wild foods and fodder, to shifting cultivation, to the broad spectrum of agroforestry practices and to single-species tree crop management.

Governance: Refers to the formation and stewardship of the formal and informal rules that regulate the public realm, the arena in which state as well as economic and societal actors interact to make decisions (Hydén and Mease, 2004).

Green water: Water derived from *precipitation* that is returned to the atmosphere via *evapotranspiration*; it can include *blue water* used for irrigation downstream of where precipitation occurred (Sood et al., 2014).

Greenhouse gas: Gaseous constituents of the atmosphere, both natural and anthropogenic, that absorb and emit radiation at specific wavelengths within the spectrum of infrared radiation emitted by the Earth's surface, the atmosphere, and clouds. This property causes the greenhouse effect. Water vapour (H_2O), carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4) and ozone (O_3) are the primary greenhouse gases in the Earth's atmosphere. Due to its short atmospheric residence time, however, water vapour is not included in greenhouse gas accounting. As well as CO_2, N_2O and CH_4, the Kyoto Protocol deals with the greenhouse gases sulphur hexafluoride (SF_6), hydrofluorocarbons (HFCs) and perfluorocarbons (PFCs) (IPCC, 2007).

Grey water: Water returned to rivers or other surface water bodies, with quality affected by previous human use (Hoekstra and Mekonnen, 2012).

Groundwater: For the purposes of this report, defined as "Water in the saturated zone of a soil profile".

Groundwater-derived streamflow: *Streamflow* that has passed through groundwater stocks before reaching a river; likely associated with, but not identical to *baseflow*.

Hydraulic conductivity: For the purposes of this report, defined as "Ease with which water can move through pore spaces, depending on porosity, pore continuity and the degree of saturation".

Hydroclimate: In this report used as part of climate directly related to the *hydrological cycle*, especially *precipitation* and *evapotranspiration*, its energy-balance determinants and consequences.

Hydrological cycle: Transfers of water between atmosphere, land and water, involving *precipitation*, terrestrial water use and *evapotranspiration* (WMO, 2012).

Hydrological Response Unit (HRU): The smallest spatial unit of a model or water resources planning exercise, within which all similar land uses, soils, slopes and hydrological processes are lumped according to user-defined thresholds (Arnold et al. 2012).

Hydrological space: Opportunity to modify water flows for specified human objectives (Ellison et al. 2017).

Interception: Water retained on aboveground plant tissues after (in a specified time interval) a precipitation event (WMO, 2012).

Land degradation: Reduction or loss in arid, semiarid and dry sub-humid areas of the biological or economic productivity and complexity of rainfed cropland, irrigated cropland, or range, pasture, forest and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns, such as: (i) soil erosion caused by wind and/or water; (ii) deterioration of the physical, chemical and biological or economic properties of soil; and (iii) long-term loss of natural vegetation (UNCCD, 1994).

Landscape: Drawing on ecosystem definitions, we define a landscape as an area delineated by an actor for a specific set of objectives (Gignoux et al., 2011). It constitutes an arena in which entities, including humans, interact according to rules (physical, biological, and social) that determine their relationships (Sayer et al., 2013).

Leaf Area Index (LAI): In this report used as sum of leaf area above a unit of land.

Leakage: In this report, 'leakage' refers to direct *emissions* elsewhere caused by the emission reduction in a project/programme area, e.g., protection of a forest area in one location leading to emissions caused by *deforestation* in other locations.

Litter layer: For the purposes of this report, defined as "Plant-based necromass covering the soil surface".

Long cycle: For the purposes of this report, defined as "Water cycle that involves precipitation over land of water with oceanic origin".

Managed forests: For the purposes of this report, managed forests are those whose structure, and the diversity and density of edible plant and animal species, have been modified by various management practices to improve their nutritional, economic and biodiversity values for people.

Mitigation (climate): An anthropogenic intervention to reduce the anthropogenic forcing of the climate system; it includes strategies to reduce greenhouse gas sources and emissions and enhancing greenhouse gas sinks (IPCC, 2007).

Monocultures: For the purposes of this report, defined as "Forms of agriculture or plantation forestry aimed at producing a single crop (typically considering anything else growing in the same plot to be a weed)".

Native species: Species, which naturally exists at a given location or in a particular ecosystem, i.e. it has not been moved there by humans http://www.cbd.int/forest/definitions.shtml [Accessed on 31 May 2018].

Natural capital: Stock of natural resources, which includes the earth's crust and its minable minerals and energy reserves, soils, water, air, atmosphere, climate and all living organisms.

Natural forest: Forest stands composed predominantly of native tree species established naturally [i.e., through natural regeneration]. This can include assisted natural regeneration, excluding stands that are visibly offspring/descendants of planted trees (CPF, 2005). See also *Primary forest*, *Naturally regenerated forest*, *Secondary forest*.

Naturally regenerated forest: Forest predominantly composed of trees established through natural regeneration (FAO 2010). See also *Primary forest*, *Secondary forest*.

Net primary production (NPP): Net primary production is the rate of photosynthesis minus the rate of respiration of primary producers (autotrophic respiration).

Overland flow: For the purposes of this report, defined as "Water (derived from *precipitation* or snowmelt) reaching streams without infiltration into soils".

Peakflow: complement to *baseflow* in the *streamflow* pattern as experienced at a specific point of observation, responding directly incoming *precipitation* (or snowmelt) (WMO, 2012).

Planetary boundaries: Limits to self-regeneration of planetary resources and global ecosystems in response to human resource use and modification (Rockström et al. 2009).

Plantation forests: Planted forests that have been established and are (intensively) managed for commercial production of wood and non-wood forest products, or to provide a specific environmental service (e.g. erosion control, landslide stabilisation, windbreaks, etc.) (Carle and Holmgren, 2003; FAO, 2010).

Precipitation: For the purposes of this report, defined as "Transfer of atmospheric moisture to the planet's surface, following a phase shift from gas (water vapour) to fluid or solid forms".

Precipitationshed: Part of the planet's surface (whether ocean or land area) that is the source of a specified fraction (e.g. 95%) of the atmospheric moisture leading to *precipitation* in a location (e.g. a country of watershed) of interest; calculations can be based on annual or seasonal precipitation (Keys et al., 2016).

Primary forest: Naturally regenerated forest of *native species*, where there are no clearly visible indications of human activities [including commercial logging] and the ecological processes are not significantly disturbed (FAO, 2010).

Rainbow water: Atmospheric moisture, derived from oceanic or terrestrial sources, that forms the basis for *precipitation* (van Noordwijk et al., 2016b).

Reforestation: Re-establishment of forest through planting and/or deliberate seeding on land classified as forest (FAO, 2012). According to the definition used by the UNFCCC, reforestation can occur on land that was forested but that has been converted to non-forested land.

Resilience: The ability of a social or ecological system to absorb disturbances while retaining the same basic structure and ways of functioning, the capacity for self-organisation, and the capacity to adapt to stress and change (IPCC, 2007).

Restoration: see *Ecosystem restoration*

Runoff: For the purposes of this report, defined as "Part of *precipitation* reaching a stream at a specific point of observation; it can include both overland and soil-based flow pathways".

Secondary forest: forests regenerating largely through natural processes after significant removal or disturbance of the original forest vegetation by human or natural causes at a single point in time or over an extended period, and displaying a major difference in forest structure and/or canopy species composition with respect to pristine primary forests (FAO, 2004).

Sedimentation: For the purposes of this report, defined as "Deposition of soil particles carried in water".

Short cycle: For the purposes of this report, defined as "Water cycle over land based on *precipitation* of atmospheric moisture derived from terrestrial *evapotranspiration*".

Snowmelt: Phase shift from solid (snow or ice) to fluid (water) (WMO, 2012).

Social-ecological systems: Systems in which the interaction between social and ecological subsystems is explicitly acknowledge and subject to feedbacks (Ostrom, 2009).

Soil carbon: Organic carbon in mineral and organic soils (including peat) to a specified depth chosen by the country and applied consistently through the time series. Live fine roots of less than 2 mm (or other value chosen by the country as diameter limit for below-ground biomass) are included with soil organic matter where they cannot be distinguished from it empirically (IPCC, 2006).

Soil macroporosity: For the purposes of this report, defined as "Volume of large pores (voids) between soil particles, per unit soil volume".

Soil organic matter: Includes organic carbon in mineral soils to a specified depth chosen by the country and applied consistently through the time series. Live and dead fine roots and dead organic matter within the soil, that are less than the minimum diameter limit (suggested 2mm) for roots and dead organic matter, are included with soil organic matter where they cannot be distinguished from it empirically (IPCC, 2006).

Streamflow: pattern in the fluctuation of water transported in a stream (or river) as experienced at a specific point of observation, with *peakflow* and *baseflow* as components and a range of operational flow separation methods (WMO, 2012).

Sustainable development goals (SDGs): A set of 17 UN-approved goals that define targets, ways of monitoring and means of implementation to improve human wellbeing and reduce negative environmental impacts and feedbacks (UN, 2015).

Tenure: Systems of tenure define and regulate how people, communities and others gain access to land, fisheries and forests. These tenure systems determine who can use which resources, for how long, and under what conditions. The systems may be based on written policies and laws, as well as on unwritten customs and practices (FAO, 2012).

Traditional (ecological) knowledge: A cumulative body of knowledge, practice and belief, handed down through generations by cultural transmission and evolving by adaptive processes, about the relationship between living beings (including humans) with one another and with their forest environment (Berkes, 1999).

Transpiration: For the purposes of this report, defined as "Phase shift of water to (gas-phase) water vapour inside plant organs followed by release to the atmosphere, controlled by stomatal closure".

Virtual water: For the purposes of this report, defined as "Water consumed in the production of agricultural and forestry products traded internationally".

Water balance: For the purposes of this report, defined as "Summation over defined spatial and temporal (e.g. daily or annual) scale of incoming (precipitation (rainfall, snow a.o.), irrigation) and outgoing water by pathway (back to atmosphere, surface water, groundwater), accounting for changes in internal storage".

Water footprint: Accounting of (green) water use associated with production of agricultural/forestry products, and/or a specified human consumption pattern, and/or the *grey water** footprint of water affected in terms of quality (Hoekstra and Mekonnen, 2012).

Water quality: For the purposes of this report, defined as "Physical (incl. temperature, sediment loads), chemical (incl. pH, nutrient and dissolved organic carbon concentrations) and biological (incl. biological oxygen demand, bacterial concentrations, biota with known tolerance levels) properties of water that are relevant for specific uses (e.g. aquatic biota or use as drinking water)".

Water tower: For the purposes of this report, defined as "Land area that through its elevation within a watershed and precipitation contributes to river flow (with quantitative criteria e.g. as specified in Dewi et al. 2017)".

Watershed: For the purposes of this report, defined as "Land area contributing *(blue) water* to a specified river (synonym of basin*) and/or the boundary that separates water flowing to multiple rivers".

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1st meeting of the GFEP Expert Panel on Forests and Water in Cambridge, UK
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