1 The World's Mountains in the Anthropocene

Udo Schickhoff, Maria Bobrowski, Suraj Mal, Niels Schwab, and R.B. Singh

Abstract

This review summarizes current understanding of drivers for change and of the impact of accelerating global changes on mountains, encompassing effects of climate change and globalization. Mountain regions with complex human–environment systems are known to exhibit a distinct vulnerability to the current fundamental shift in the Earth System driven by human activities. We examine indicators of the mountain cryosphere and hydrosphere, of mountain biodiversity, and of land use and land cover patterns, and show that mountain environments in the Anthropocene are changing on all continents at an unprecedented rate. Rates of climate warming in the world's mountains substantially exceed the global mean, with dramatic effects on cryosphere, hydrosphere, and biosphere. Current climatic changes result in significantly declining snow-covered areas, widespread decreases in area, length, and volume of glaciers and related hydrological changes, and widespread permafrost degradation. Complex adaptations of mountain biota to novel constellations of bioclimatic and other site conditions are reflected in upslope migration and range shifts, treeline dynamics, invasion of non-native species, phenological shifts, and changes in primary production. Changes in mountain biodiversity are associated with modified structure, species composition, and functioning of alpine ecosystems, and compromise ecosystem services. Human systems have been negatively impacted by recent environmental changes, with both inhabitants of mountain regions as well as people living in surrounding lowlands being affected. Simultaneously, accelerating processes of economic globalization cause adaptation strategies in mountain communities as expressed clearly in changing land use systems and mobility patterns, and in increasing marginalization of peripheral mountains and highlands. The current state of the world's mountains clearly indicates that global efforts to date have been insufficient to make significant progress towards implementing the Sustainable Development Goals of the 2030 Agenda for Sustainable Development, adopted by all United Nations member states.

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1.1 Introduction

It has been known since Alexander von Humboldt (1769–1859) that the decrease of temperature with increasing elevation in mountains induces vertical climate alterations which are reflected in all climate-dependent landscape elements, especially in the altitudinal zonation of vegetation and land use. From the results of the pioneers and key exponents of geographical high mountain research such as von Humboldt, Carl Troll (1899–1975), and Bruno Messerli (1931–2019) a picture of the natural setting of high mountains and of the interwoven geoecological and human-geographical factor complexes emerged, which has undergone major changes in recent decades. Over the past decades, mountain regions have been subjected to above-average climate warming and significant land use changes. Contemporary climate change and modified land use intensities and land use systems have tremendous effects on mountain landscapes so that the pioneers of high mountain research would hardly recognize certain landscapes on a visit today. These effects are the core theme of this book; they are explored in the following chapters which include compelling examples from around the world.

The significance of mountains for the Earth system (Fig. 1.1) and for a considerable part of the human population is often not rated highly enough. Mountain ecosystems have evolved on every continent, characterized by the complexity of their topography associated with steep environmental gradients, i.e. distinct variations of climatic, edaphic, and other environmental factors over short distances (Schickhoff 2011). Mountains and highlands cover nearly 25% of the terrestrial surface of the Earth (Romeo et al. 2015), 11% of the global land surface are higher than 2000 m above sea level (a.s.l.) (Kapos et al. 2000). Based on topographic ruggedness of the Earth's surface, Körner et al. (2017) calculated an area of 12.5% of the land surface covered by mountains (excluding Antarctica) of which 24% comprise alpine and nival belts. Elias (2020) and Testolin et al. (2020) quantified a comparable land area covered by alpine biomes. As a result of the physiography and diverse topography—major mountain ranges rise prominently above their surroundings—mountains exert a great influence on energy and moisture fluxes and on local and regional airflow patterns up to the large-scale atmospheric circulation. Their influence on airflows, temperature and humidity extends far beyond their geographic boundaries and may be felt for hundreds and thousands of kilometers (Bach and Price 2013).

Mountains provide ecosystem goods and services to more than half of humanity, thus they are of critical importance to people in almost every country of the world (Ives et al. 1997; Schickhoff 2011; Byers et al. 2013). Approximately 13% of the human population derives their life-support directly from mountains (Price 1998; Romeo et al. 2015), including diverse communities of distinct ethno-linguistic and cultural identity. Mountains are essential resource regions for the supply of water, energy, grazing lands, forest and agricultural products, and mineral resources. Many plant and animal species are endemic to mountain regions which are characterized by increased biodiversity relative to the surrounding lowlands (biodiversity hotspots). Mountains are also centres of ethnic, religious and cultural diversity, provide ample opportunities for recreation and tourism, and are of spiritual significance. Water supply is usually considered the key function of mountains for humanity since all of the world's major rivers have their headwaters in mountains, and huge quantities of freshwater are stored as snow and ice as well as in lakes and reservoirs and gradually released to the lowlands. Mountains are often called 'water towers' of the Earth owing to the key role they play for supplying water to billions of people in lowlands used for drinking, domestic use, irrigation,

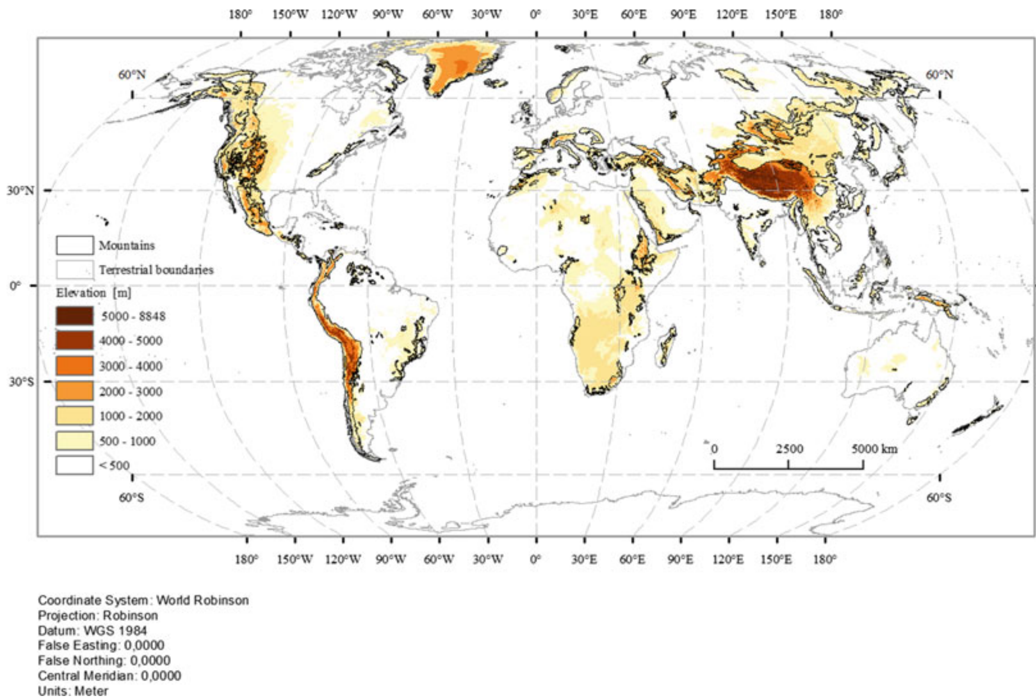


Fig. 1.1 Mountains of the world (background image from https://dds.cr.usgs.gov/srtm/version2_1/SRTM30/)

hydropower, industry, and transportation (Körner et al. 2005; Viviroli et al. 2007; Schickhoff 2011; Byers et al. 2013). Water supply from mountains is essential for life in semiarid and arid regions where the proportion of water generated at higher elevations may be more than 95% as in the basin of the Aral Sea (Messerli 1999). Even in humid regions, 60–80% of the total freshwater available is provided by mountain watersheds. Hydropower from these watersheds provides about one-fifth of the world's total electricity supply (Byers et al. 2013). Water supply from mountains forms the basis for ensuring availability and sustainable management of water and sanitation for billions of people (Goal 6 of the UN Sustainable Development Goals). Integrated water resources management as a global framework covering policies, institutions, management instruments and financing for the comprehensive and collaborative management of water resources has still been implemented at a low level (UN 2020).

Mountains show above-average species richness and comprise many unique biomes that are globally significant as core areas of biodiversity. A quarter of all terrestrial biodiversity is situated in mountains (Körner et al. 2017). Over evolutionary time scales, mountains also have generated high levels of diversity through in situ adaptations and diversification (Badgley et al. 2017; Hoorn et al. 2018). The global hotspots of species diversity, areas with increased levels of species richness and high proportions of endemic species, are predominantly mountainous regions. The particular species richness is related to the topographic complexity and associated high levels of geodiversity, i.e. the small-scale diversity of habitats and site conditions resulting from steep climatic and ecological gradients in fragmented and topographically diverse terrain. The compression of climatic life zones along vertical gradients, spatial isolation, combined with effective reproduction systems, as well as moderate disturbance influences additionally contribute to small-scale extraordinarily high levels of biodiversity. Tropical and subtropical mountain regions in particular are home to highly diverse and species-rich ecosystems constituting the global centres of vascular plant diversity (Körner 2002; Barthlott et al. 2005, 2007). Species diversity includes the most important food staples such as potatoes, maize, wheat, rice, beans or barley which had been domesticated in mountain regions (Brush 1998). Promoting sustainable use of terrestrial ecosystems, reversing land degradation, and halting biodiversity loss are major targets at the heart of Goal 15 of the UN Sustainable Development Goals which need to be supported in particular in mountain regions (UN 2020).

The resource function of mountain regions also contributes substantially to their global significance (Schickhoff 2011). For instance, mountain forests account for more than a quarter of the area of global closed forests (Kapos et al. 2000). They provide diverse goods and services to millions of people including provisioning services (both timber and non-timber forest products such as fuelwood, fodder, grazing resources, medicinal plants, and mushrooms), regulating and supporting as well as cultural services (Price and Butt 2000; Price et al. 2011; Gratzer and Keeton 2017). Mountain forests play a critical role for mountain dwellers and valley communities regarding protection against natural hazards such as landslides, rockfalls, avalanches, and floods as well as for reducing soil erosion and maintaining hydrological cycles. Mountain forests also represent a major carbon sink, and carbon sequestration in those forests is of increasing significance in climate change mitigation. The past two decades have seen a significant increase globally in the extraction of mineral resources from mountains; mines in mountains are the major current source of many of the world's strategic non-ferrous and precious metals (Fox 1997; Jacka 2018), contributing to the fast increasing global material footprint. As mountain regions continue on a path of using natural resources unsustainably, the successful transition to sustainable consumption and production patterns is more essential than it has ever been before (addressed by Goal 12 of the UN Sustainable Development Goals) (UN 2020).

The global significance of mountain regions can only be fully grasped if the focus is on mountain dwellers. Between 2000 and 2012, the global mountain population increased from 789 to 915 million people, and will further increase in the next decades (Romeo et al. 2015). Most mountain populations are nowadays integrated, to varying degrees, economically, socially and politically with lowland communities and the wider world (Funnell and Parish 2001). Nevertheless, mountains are still home to many indigenous peoples, encompassing an amazing diversity of human cultures and communities. For example, 100 different ethnic/caste groups were identified in the 2001 census in the mountainous state of Nepal (Sharma 2008), and more than 700 languages are spoken in mountainous regions of New Guinea (Stepp et al. 2005). This cultural diversity contributes to the attractiveness of mountains that have become key tourism destinations in many parts of the world. The significance of mountains as centres for recreation, adventure, scenic beauty or interaction with local people will increase in coming decades as tourism is the world's largest and fastest growing industry. The large influx of tourists to mountain regions is not without conflicts due to the impacts on fragile high altitude environments and the special spiritual and cultural significance mountains have in many cultures (Price and Kohler 2013; Hamilton 2015).

Mountain ecosystems represent some of the few remaining wilderness areas of the globe, and encompass some of the most intriguing habitats in terms of the particular fascination of high mountain landscapes, with regard to high biodiversity levels and resident biota's special adaptation to the harsh physical environment, as well as in terms of the extraordinary cultural diversity and the sophisticated and complex resource utilization strategies that mountain dwellers have developed over many generations. Mountain ecosystems on the other hand are exceptionally fragile, susceptible to global environmental changes, and less resilient since longer periods of time may be needed for recovery from damage or excessive stress. As elsewhere on the globe, climatic changes and land use changes are the major drivers which are increasingly threatening the integrity of mountain ecosystems, affecting their capacity to provide goods and services.

Mountain regions around the world provide increasing evidence of ongoing impacts of land use change and of climate change on physical and biological systems. High elevation environments with steep relief, complex topography, cryospheric systems (snow, glaciers, permafrost), the compression of ecological vertical gradients and specific human–environmental subsystems are in general considered to be among the most sensitive terrestrial systems to reflect effects of climatic variations and consequences of changes in land use (Huber et al. 2005; Körner et al. 2005; Grabherr et al. 2010; Löffler et al. 2011; Schickhoff 2011, 2016a, b; Gottfried et al. 2012; Grover et al. 2015; Schickhoff et al. 2016a; Pauli and Halloy 2019; Hock et al. 2019; Schickhoff and Mal 2020). Observed changes of glaciers, snow cover, permafrost, hydrological conditions, and of the complex altitudinal zonation of vegetation and fauna indicate a distinct vulnerability, mountains are considered to be at the forefront of climate change impacts (Pihl et al. 2019). Mountain plants and animals, in particular endemic species, are often adapted to relatively narrow ranges of temperature and precipitation, even minor climatic changes can have significant effects (Körner 2003; Grabherr et al. 2010). If the water supply from High Asia is significantly reduced by retreating glaciers, more than half of Asia's population would be adversely affected (cf. Körner et al. 2005; Viviroli et al. 2007). More than a billion people in Asia live in the watersheds of rivers that have their sources in mountains. With regard to physical systems, current global warming has already left distinct traces in the cryosphere and hydrosphere of the world's mountains. It is also a powerful stressor on alpine biota, inducing shifts in phenology, species distributions, community structure as well as other ecosystem changes. As the climate crisis continues unabated, in particular in mountain regions, and as pervasive and catastrophic effects have become obvious, taking urgent action to combat climate change and its impacts and accelerating the transitions needed to achieve the Paris Agreement is the order of the day (Goal 13 of the UN Sustainable Development Goals) (UN 2020).

In many mountain ranges, ongoing alterations of montane and alpine land use systems caused by widespread socio-economic transformation processes are the major underlying driver of the transition of mountain landscapes. From a global perspective, changes in land use affect mountain forests and their ecosystem services in particular. In recent decades, two opposing trends have become apparent in the area covered by forests in mountain regions reflecting general global trends in forest cover: In many countries of the Global South forest cover is further declining, whereas a gradual expansion can be observed in industrialized countries (Schickhoff 2011, 2016b). For both montane and alpine life zones, it needs to be highlighted that the fragility of these high elevation environments poses a tremendous challenge for sustainable land use and natural resource management.

This chapter provides a global overview of the current state of knowledge on the effects of climate change and land use change on mountain landscapes. Presenting examples from major mountain systems around the world, the current knowledge is summarized with respect to climatic changes, impacts on physical systems (changes of snow cover, glaciers, permafrost, and related hydrological processes), biotic responses (phenological shifts, species migrations, range extensions, treeline dynamics, shifts in species composition), and effects of modified land use systems. Understanding how structures and functions of mountain ecosystems are affected by environmental change is a focal point for the mountain research agenda, in particular with regard to the abundance of ecosystem services and the multifunctionality of mountains (cf. Egan and Price 2017; Palomo 2017). At the same time, understanding the effects of environmental change on mountain ecosystems is of vital importance for adaptation planning, both for mountain people and for billions living in lowlands, in order to mitigate implications of climate and land use changes and to enhance the adaptive capacity of mountain socio-ecological systems in response to anticipated future changes. The international recognition of the importance of mountain environments and mountain peoples has increased over recent decades, however, the local and global awareness for the essential role mountain systems play in the geo-biosphere needs to be further supported and increased. Milestones of international efforts to establish mountains as a research priority, to support intergovernmental and nongovernmental processes of advocacy for mountains, and to support sustainable mountain development in general include the establishment of the UNESCO-MAB (Man and Biosphere) project on 'Impact of Human Activities on Mountain and Tundra Ecosystems' in 1973, the United Nations Conference on Environment and Development (UNCED) held in Rio de Janeiro in 1992 (inclusion of a mountain chapter into Agenda 21), the establishment of both the Mountain Forum (a global network of intergovernmental, nongovernmental, scientific, and private-sector organizations and individuals) in 1995 and the Mountain Research Initiative (a global scientific promotion and coordination effort towards strengthening the dialogue between science and policy) in 2002, the International Year of Mountains 2002, and the UN resolution 'Sustainable Mountain Development' in 2010 (Messerli 2012; Price and Kohler 2013; Kohler et al. 2015). Advances in international efforts to increase awareness for the importance of mountain research and development has stimulated scientific interest, reflected in a number of recent pioneering national and global research initiatives such as the GLORIA (Global Observation Research Initiative in Alpine Environments) programme, the scientific collaboration network of the WGMS (World Glacier Monitoring Service) or the Global Terrestrial Network for Permafrost (GTN-P). We strongly endorse further awareness-raising by producing and disseminating mountain-related education and research materials. All efforts towards sustainable mountain development should ideally be embedded in the 2030 Agenda for Sustainable Development, an urgent call for action substantiated by the 17 Sustainable Development Goals (UN 2020).

1.2 Recent Climate Change and Its Effects in Major Mountain Systems of the World

1.2.1 Climatic Changes

1.2.1.1 General Overview

Greenhouse gas emissions which continue to increase are the dominant factor in the observed persistent warming trend for the global mean surface temperature over recent decades and in recent years, with the last five-year period (2015–2019) and the last ten-year period (2010–2019) being the warmest of any equivalent period on record, and with 2015, 2016, 2017, 2018, and 2019 being the five warmest individual years (WMO 2019). July 2019 was the hottest month on record globally. Global warming is currently estimated to be 1.1 °C above pre-industrial values (1850–1900) and 0.2 °C warmer than 2011–2015, with the high latitudes of the Northern Hemisphere, in particular the northern Asian sector, showing the largest increase in mean temperature (Hoegh-Guldberg et al. 2018; WMO 2019). Here, the polar amplification leads to warming rates of more than 2 °C per 50 years, while warming trends and increasing temperature extremes have been generally observed in major mountain systems of the world over the past century (IPCC 2014). Temperature trends in most mountain regions substantially exceed the global mean over recent decades (Fig. 1.2), albeit with distinct patterns of spatial and seasonal differentiations, in particular in terms of vertical gradients. A current warming rate of 0.3–0.4 °C per decade is observed in most mountain regions of the world including western North America, the European Alps, and High Mountain Asia. This rate is significantly higher than the global mean and accelerating (cf. IPCC 2018; Hock et al. 2019; WMO 2019).

A widespread phenomenon is the amplification of warming rates at higher elevations, to be attributed mainly to changes in albedo and downward thermal radiation (Rangwala et al. 2013; Pepin et al. 2015; Hasson et al. 2016; Palazzi et al. 2019). At local and regional scales,

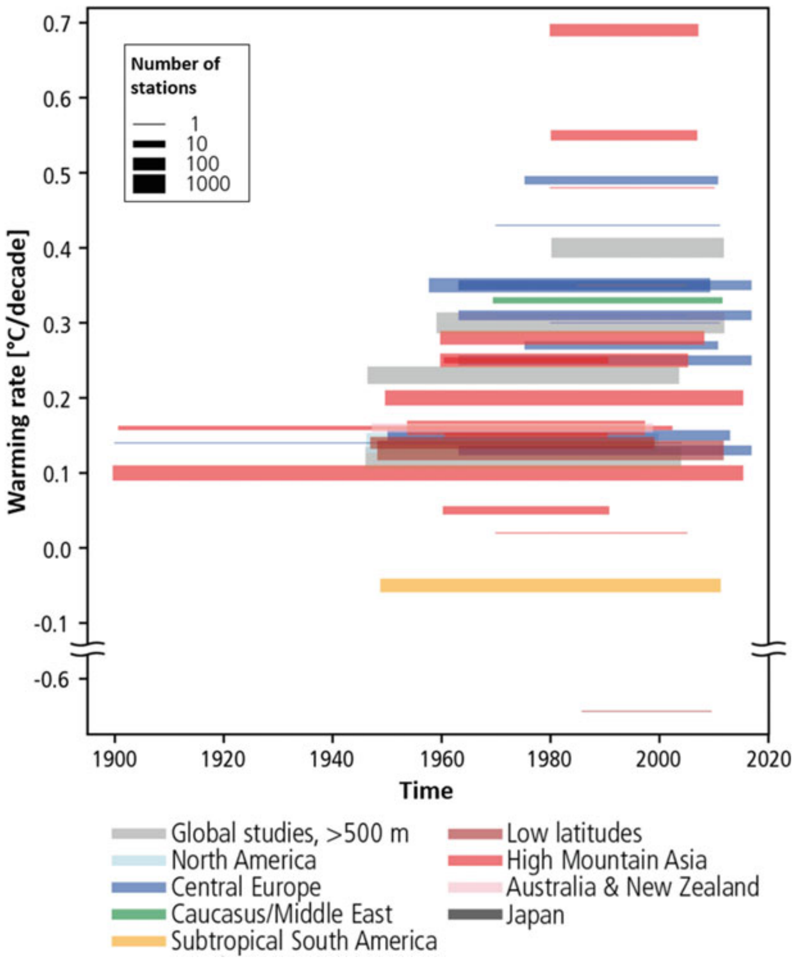


Fig. 1.2 Mean annual surface air temperature in mountain regions; each line refers to a warming rate from one of 40 studies based on multiple observation stations, with line thickness indicating the number of observation stations used. (modified from Hock et al. 2019)

however, evidence for elevation-dependent warming is sometimes contradictory. Obviously, trends in air temperature vary with elevation, but not in a consistent manner. Variations result from the effects of region, season, and selected temperature indicators (cf. Hock et al. 2019). The amplification of warming at higher elevations will increase with higher greenhouse gas emission scenarios, subjecting high elevation environments to comparatively more distinct changes in habitat conditions than lower elevations (Schickhoff et al. 2016a). Regardless of the underlying climate scenario, surface air temperature in mountain regions is projected to further increase at an average rate of at least 0.3 °C per decade until the mid-21st century (IPCC 2018), irreversibly affecting mountain ecosystems and their biodiversity, and impairing their capacity to provide key ecosystem services. This emphasizes the necessity of achieving the climate action target of the UN Sustainable Development Goals (UN 2020).

Compared to temperature changes, precipitation trends in mountain systems of the world are much more heterogeneous. Observations of annual precipitation often do not show significant increases or decreases over the past decades, while snowfall exhibits a more or less consistently decreasing trend, in particular at lower elevations (Hock et al. 2019). All greenhouse gas emission scenarios project a further decrease of snowfall at lower elevations throughout the twenty-first century, thus the rain/snow partitioning will be continuously affected. In contrast, projections of annual precipitation for the next decades show increases in the order of 5–20% for many mountain regions in South and East Asia, East Africa, and temperate Europe; only some mountain regions (the Mediterranean, Southern Andes) will experience a decrease in annual precipitation (Hock et al. 2019). The frequency and intensity of extreme precipitation events is projected to increase in many mountain regions.

1.2.1.2 Regional Overview

Asia[1] and Australasia

Temperature trends in the vast Hindu Kush Himalaya (HKH) are quite representative for many of the extensive mountain systems of Asia. The HKH has experienced warming from 1901 to 1940, cooling from 1940 to 1970, and a strong amplification of warming rates to 0.2 °C per decade over the period 1951–2014 (Fig. 1.3) (Ren et al. 2017; Krishnan et al. 2019a). Without any doubt, the warming trend has accelerated in the past two decades and in recent years (Diodato et al. 2011; Kattel and Yao 2013; Gerlitz et al. 2014; Hasson et al. 2016). At higher elevations, mean annual and mean annual maximum temperatures have been increasing at rates between 0.6 and c. 1 °C per decade over the past 40 years (Shrestha et al. 1999; Liu et al. 2006, 2009; Bhutiyani et al. 2007, 2010; Shrestha and Aryal 2011; Yang et al. 2011). Winter season temperature trends have been generally higher than those of other seasons (Hasson et al. 2016). Extreme warm days and nights show an increasing trend of occurrence in the past decades (nights by 2.54 days per decade), while occurrences of cold days and nights have declined (Hijioka et al. 2014; Krishnan et al. 2019a). In addition to the significant warming the HKH has seen in the past, the climate is projected to change more dramatically in the coming decades, with warming to be at least 0.3 °C higher, and in the NW Himalaya and Karakoram at least 0.7 °C higher than the targeted 1.5 °C as a global mean (Dimri et al. 2018; Krishnan et al. 2019a). Across Asia, the strongest warming of hot extremes is projected to occur in western and central Asia (Hoegh-Guldberg et al. 2018).

Significant and accelerated warming rates were observed over the entire Tibetan Plateau (Hasson et al. 2016; You et al. 2016, 2017; Ren et al. 2017). Yan and Liu (2014) reported a considerably increased warming trend in mean annual temperature of 0.32 °C per decade between 1961 and 2012, overcompensating the global warming slowdown period of 1998–2013 (cf. Ji and Yuan 2020). Current warming rates in Tibet are much higher than previously estimated (cf. Liu and Chen 2000), for the period 1992–2017 a warming rate of 0.47 °C per decade was assessed (Li et al. 2019). Significant warming of winter and annual temperatures are consistently reported from the West and Central Himalaya in India. Over the northwestern subregion, winter temperature has shown an elevated rate of increase (1.4 °C/100 years) compared to the monsoon temperature (0.6 °C/100 years) during the period from 1866 to 2006 (Bhutiyani 2015, 2016). Higher winter season mean temperature trends of up to 2.0 °C were detected for the period 1985–2008 (Bhutiyani et al. 2007, 2010; Shekhar et al. 2010; Dimri and Dash 2012; Singh D et al. 2015; Kumar et al. 2018). Seasonal maximum and minimum temperatures have increased by 2.8 and 1.0 °C, respectively; they show an increasing trend over the Pir Panjal, Shamshawari and Greater Himalayan ranges (Shekhar et al. 2010). Significantly increasing winter, monsoon and annual temperatures are reported from most stations, with the magnitude of warming being higher during recent decades compared to the century average (Bhutiyani et al. 2010; Singh and Kumar 2014; Shafiq et al. 2019; Negi et al. 2020). In Uttarakhand, temperature records of the past 100 years show a notable warming trend, particularly prominent during the last decade and at higher elevations (Mishra 2014; Singh RB et al. 2016).

A recent comprehensive evaluation of temperature trends across Nepal over the period 1980–2016 showed widespread significant warming which is higher for maximum temperature (0.4 °C per decade) than for minimum

[1]The information on mountain systems in Asia compiled in this paper is expanded and updated from Schickhoff & Mal (2020).

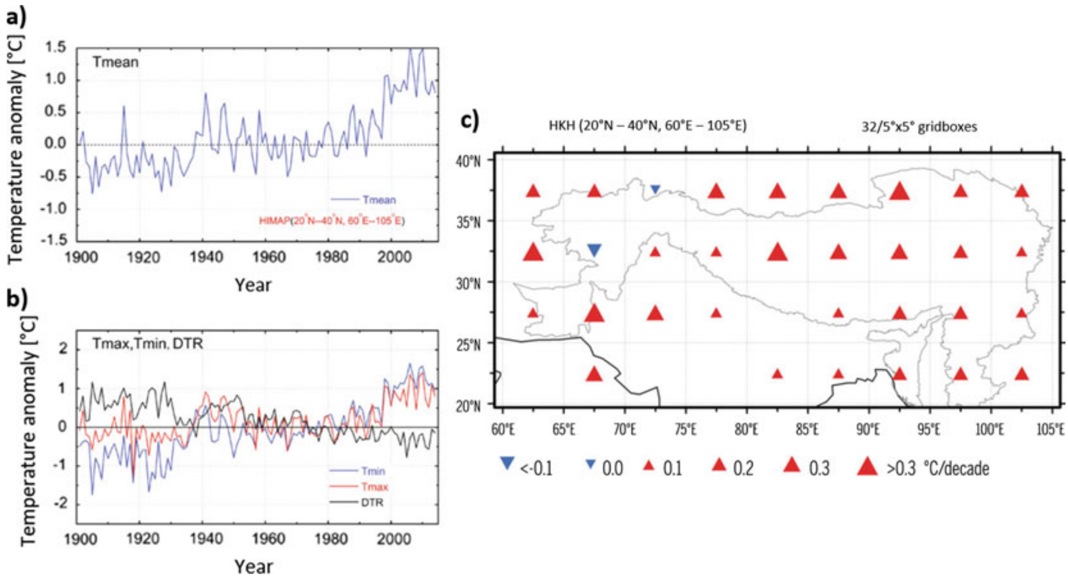


Fig. 1.3 Annual mean temperature anomaly series (°C) for the HKH region between 1901 and 2014 relative to 1961–90 mean values (**a**: Tmean; **b**: Tmax, Tmin, and DTR); **c**: Grid-averaged trends of annual mean temperature in the HKH region since 1901. (modified from Krishnan et al. 2019a)

temperature (0.2 °C per decade), higher in the mountainous region than in valleys and lowlands, and higher in the pre-monsoon season than in the rest of the year (Karki et al. 2019). Shrestha et al. (2019) reported more or less equal magnitudes of warming, with a more pronounced rate of increase after 2005 (see also Dahal et al. 2019). Current mean annual temperature warming rates in Sikkim and Bhutan amount to 0.3–0.4 °C per decade (cf. Hoy et al. 2016; Goswami et al. 2018; Patle et al. 2019), comparable to current warming trends in the eastern Himalaya (Arunachal Pradesh, India) (cf. Yang et al. 2013; Bhagawati et al. 2017). In the western HKH, annual mean temperatures showed a slight increase in recent decades, whereas summer temperatures are slightly decreasing or show rather small magnitude of trends at many climate stations in the Karakoram (Fowler and Archer 2006; Khattak et al. 2011; Bocchiola and Diolaiuti 2013; Raza et al. 2015; Hasson et al. 2017; Waqas and Athar 2019; Latif et al. 2020). In winter and summer, the Karakoram has been near the boundary between large-scale cyclonic and anti-cyclonic trends over recent decades, while the Central Himalaya has been under the influence of an anti-cyclonic trend (Norris et al. 2019). Deviations from the general HKH climate warming pattern are linked to the Karakoram glacier anomaly (see 2.2; Forsythe et al. 2017).

Patterns of elevation-dependent warming have been widely observed in the HKH and in particular on the Tibetan Plateau and surrounding regions (Hasson et al. 2016; Karki et al. 2019; Krishnan et al. 2019a; Dimri et al. 2020). Maximum warming rates have been assessed between 4000 and 5000 m a.s.l., locally even at higher elevations (cf. Gao et al. 2018; Pepin et al. 2019; Rangwala et al. 2020). High resolution temperature trends over the Himalaya for the period since the 1980s show a clear elevational gradient in the pre-monsoon season with maximum values of up to 1.2 °C per decade at higher elevations (Gerlitz et al. 2014; Schickhoff et al. 2015). Thakuri et al. (2019) confirmed elevation-dependent warming based on stations up to 2600 m a.s.l. in Nepal. Higher warming rates at intermediate elevations were reported by Negi et al. (2020) for the NW Himalaya.

Trends in annual precipitation are difficult to derive considering the widespread non-availability of long-term observations and distinct variabilities prevalent in different subregions and seasons (Schickhoff et al. 2016a). Over the last 100-plus years, the trend of annual precipitation in the entire HKH is characterized by a slight decrease (Fig. 1.4) (Ren and Shrestha 2017; Ren et al. 2017; Krishnan et al. 2019a). The marginal reduction in annual precipitation (with concurrent interdecadal variability) over quite a large part of the Indian subcontinent is consistent with a weakening tendency of Indian summer monsoon precipitation, associated with a weakening land-sea thermal gradient, a decline in the number of monsoon depressions and an increase in the number of monsoon break days (Krishnamurthy and Ajayamohan 2010; Kulkarni 2012; Lacombe and McCartney 2014; Roxy and Chaithra 2018; Singh D et al. 2019; Basu et al. 2020). Nevertheless, all global and regional climate models and scenarios project an increase in both the mean and extreme precipitation of the Indian summer monsoon in the twenty-first century, largely due to increased moisture flux from ocean to land (Christensen et al. 2013; Krishnan and Sanjay 2017). Observations in subregions of the HKH over recent decades show either slightly decreasing or slightly increasing trends, but trends are rarely significant. Generally increasing trends for winter precipitation, originating from western disturbances, and positive trends at many stations for summer precipitation (predominantly monsoonal) have been observed in the Karakoram over recent decades (Khattak et al. 2011; Palazzi et al. 2013; Hasson et al. 2017). Increasing trends of winter precipitation at the majority of stations in the NW, W, and Central Himalaya in India are overcompensated by decreasing summer (monsoonal) precipitation rates since the 1960s, resulting in prevailing negative trends of annual precipitation (Sontakke et al. 2008; Bhutiyani et al. 2010; Singh and Mal 2014; Bhutiyani 2016; Shafiq et al. 2019). Decreasing trends of annual precipitation were also observed in Far West Nepal (Wang et al. 2013; Pokharel et al. 2019), while the major remaining parts of Nepal experienced a positive trend of annual precipitation, in particular of monsoonal precipitation, in the period 1979–2016, notably in the years after 2000 (Shrestha et al. 2019; see also Panthi et al. 2015 for the Kali Gandaki River Basin). Further east (Sikkim, Bhutan, Arunachal Pradesh, eastern Himalaya) no significant longer-term trends or slightly positive trends, if any, are observed (Qin et al. 2010; Li et al. 2011; Jain et al. 2013; Hoy and Katel 2019). Annual precipitation on the Tibetan Plateau has slightly increased since the 1960s, although respective trends are not uniform across the entire Plateau region (Hasson et al. 2016; You et al. 2017).

A clear shift in temporal characteristics of precipitation variation has been assessed after 1990 with greater interannual variability and more frequent intense precipitation events and less frequent light precipitation events (Krishnan et al. 2019a). Higher-elevation areas, in particular the Tibetan Plateau, have witnessed a significant increase in annual mean daily precipitation intensity (Ren et al. 2017; Zhan et al. 2017),

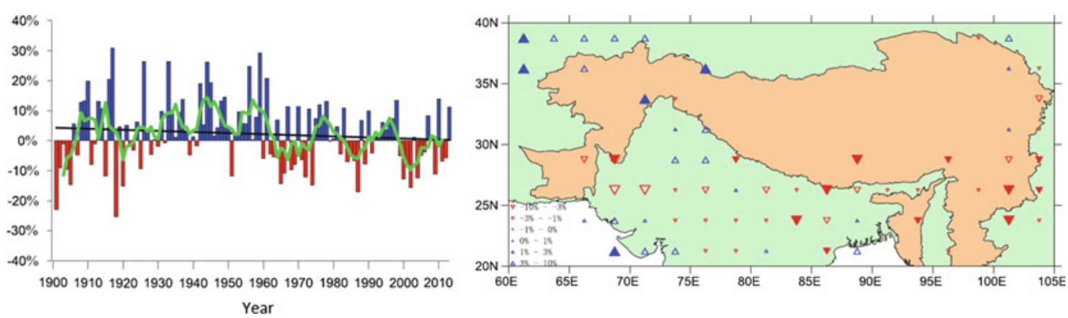


Fig. 1.4 Regional average annual precipitation percentage anomaly (PPA) during 1901–2014 in the HKH region (green line: five-year moving average; black line: linear trend) and spatial distribution of linear trends. (Modified from Krishnan et al. 2019a)

subjecting alpine life zones to additional stress. Over the western Himalaya, Priya et al. (2017) and Krishnan et al. (2019b) identified a rising trend of synoptic-scale western disturbance activity and related precipitation extremes during the recent few decades. For some parts of Nepal, a significant increase of high intensity precipitation extremes was observed during 1970–2012, and at the same time, the number of rainy days is significantly decreasing over the whole of Nepal while the number of consecutive dry days is significantly increasing (Karki et al. 2017).

Significant warming has also characterized surface air temperature trends in other Asian and Australasian mountain systems. Observations in E and NE Asia (China, Taiwan, Korea, Japan) indicate an abrupt increase of summer mean surface air temperature since the mid-1990s (Dong B et al. 2016), with extreme summertime droughts having increased in frequency, severity and duration (Zhang J et al. 2019). Substantial warming rates are to be expected for the coming decades (Hsu and Chen 2002; Lee et al. 2014; Murata et al. 2015). Mountains of southern and eastern Siberia experienced an outstanding 2–3 °C increase of mean annual air temperature over the last three decades (Fedorov et al. 2014; Desyatkin et al. 2015), while the mean winter season temperature in the Siberian Altai has increased by up to 4 °C (Kharlamova et al. 2019). Strong positive temperature trends associated with an increase in summer days and a significant decline in frost days have also been observed in Mongolian mountains (Dashkhuu et al. 2015). High-elevation areas in the Tien Shan and Pamir experienced warming rates of up to 0.5 °C (mean annual air temperature) per decade over recent decades (Chevallier et al. 2014; Deng et al. 2015; Hu et al. 2016). Significant, but slightly lower warming rates were assessed in the Caucasus (Elizbarashvili et al. 2017), Pontic, Zagros and Arabian Mountains (Donat et al. 2014; Ghasemi 2015; Yucel et al. 2015) as well as in the mountains of SE Asia (Supari et al. 2017; Tang 2019). In Australia and New Zealand, mean temperatures have warmed strongly since 1900 (c. +0.9 °C), resulting in warmer, less frosty winters (Mullan et al. 2010; Reisinger et al. 2014). However, a reduced increase of mean temperatures (0.06 °C/decade) has occurred in New Zealand since 1970, while no clear overall pattern can be derived from precipitation variations which are connected with the Southern Oscillation Index (SOI) and the Interdecadal Pacific Oscillation (IPO) (McGlone et al. 2010). Hawai'i has experienced strong warming at higher elevations, with snowfall on Hawai'i's mountain peaks being projected to almost completely disappear by 2100 (Frazier and Brewington 2020).

Europe

In congruence with the global climate response to increasing greenhouse gas concentrations, distinctive long-term temperature trends have been observed in European mountains, with regionally and seasonally different rates of warming. All of Europe has warmed significantly, in particular since the 1960s, with Scandinavia showing strongest winter warming, and SW, Central, and NE Europe particularly high summer warming (Fig. 1.5) (Kovats et al. 2014; EEA 2017). In the European Alps, annual mean temperatures increased by about 2 °C since the late nineteenth century which is a rate more than twice as large as the global or northern hemispheric average (Auer et al. 2007; Brunetti et al. 2009; APCC 2014; Gobiet et al. 2014). Warming rates increased distinctly to c. 0.5 °C per decade since the early 1980s, with the most intense warming since the 1990s, leading to an annual mean temperature increase of more than 1 °C in 25 years (Weber et al. 1997; EEA 2009). In Switzerland, the 1988–2017 summer average was by far the warmest 30-year period over the past 300 years (cf. Fig. 1.5), resulting in more frequent and more intense heatwaves, less frequent cold periods, and an upward shift of the winter zero-degree line by 300–400 m since the 1960s (CH2018 2018). Rottler et al. (2019) detected elevation-based differences in temperature trends during autumn and winter with stronger warming at lower elevations. Precipitation trends are sub-regionally differentiated. In the southern Alps, precipitation trends are small and not significant. Here, Brugnara and Maugeri

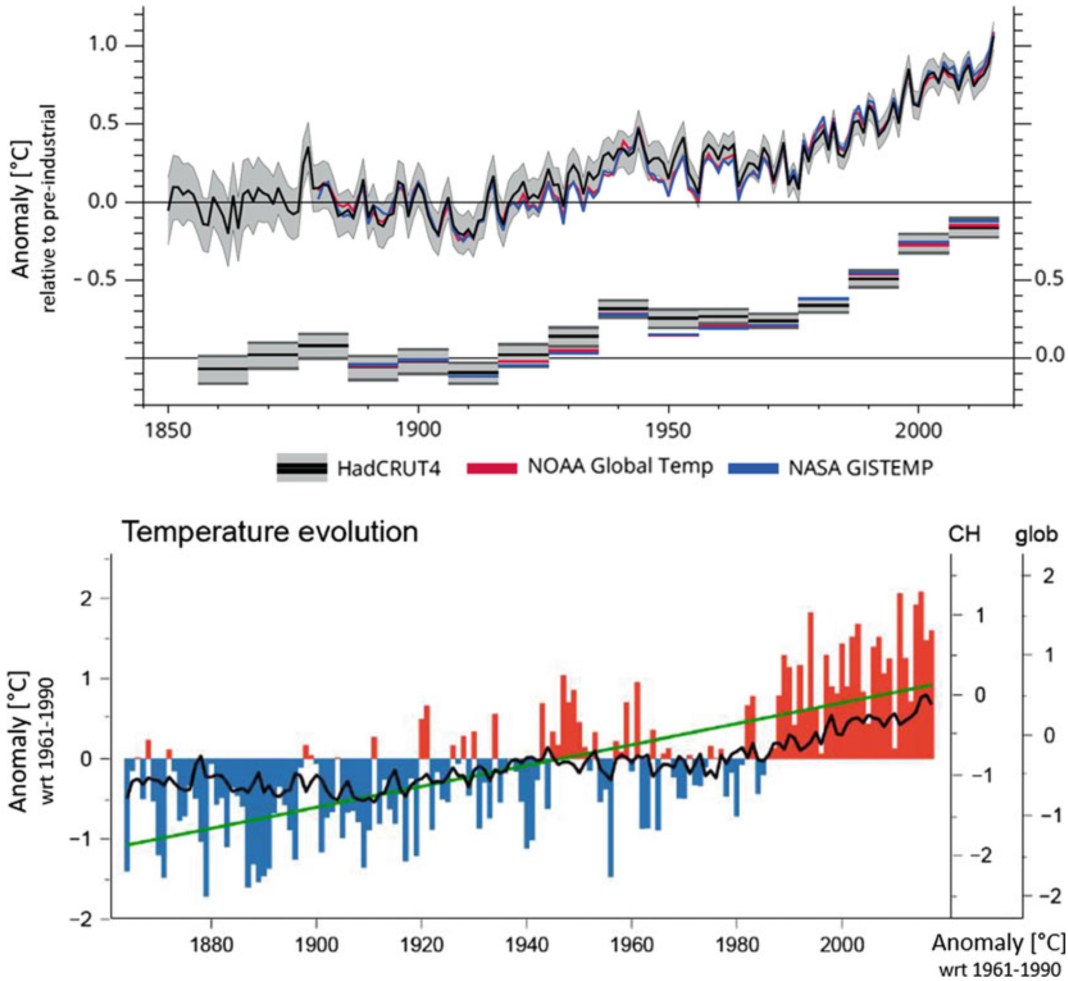


Fig. 1.5 Upper panel: European average temperatures between 1850 and 2015 over land areas relative to the pre industrial period; lower panel: Swiss and global annual mean temperatures, relative to the means for 1961–1990 (left axis) and 1981–2010 (right axes, left: Swiss series (CH) and right: global (glob)); the Swiss mean values are shown as bars, the global values as a black line; the linear trend fit to the Swiss values is shown in green. (Modified from EEA 2017; CH2018 2018)

(2019) assessed a significantly decreasing precipitation frequency over the period 1890–2017, and related this trend to a step-like reduction in cyclonic weather types over central Europe. Considerable and significant precipitation increases, however, were observed in northern Switzerland for the winter season (~20% per 100 years) as well as in the Austrian Alps (a 10 to 15% increase) over the past 150 years (APCC 2014; CH2018 2018). Likewise, the frequency of extreme precipitation events in the Alps increased by about 25% since 1900. In summary, precipitation evolution in the Greater Alpine Region shows significant regional and seasonal differences over the last century, with increases in the NW and decreases in the SE (Auer et al. 2007). Simultaneous to accelerated warming in the next decades, projected changes indicate less precipitation and more severe droughts in summer, and more precipitation in winter (Gobiet et al. 2014). The Carpathians experienced strongest warming in summer seasons, with rates of up to 2.4 °C from 1961 to 2010, and increasing annual precipitation in most of the region, except for the western and southeastern areas (Werners et al. 2014).

Climate observations in the Mediterranean region indicate increasing temperatures and decreasing precipitation, contributing to a progressive and substantial drying of the land surface since 1900. For instance, mean surface air temperature in the Pyrenees increased by 0.21 °C per decade, while precipitation decreased by 2.5% per decade in the period 1950–2010, leading to more frequent and intense droughts (EEA 2017). Warming rates are predicted to be in a similar magnitude in western and eastern Mediterranean mountains over the coming decades, the western mountain ranges such as the Sierra Nevada, the Pyrenees and the Apennines, however, will suffer to a larger extent from decreasing precipitation than the eastern Mediterranean mountains (Dinaric Alps, Balkan, Rhodopes, Pindos) (Nogués-Bravo et al. 2008, 2012). The mean temperature in Scandinavian mountains has increased significantly since the early twentieth century, with particularly warm periods in 1930–1950 and after 1980. From 1964 to 2013, mean annual temperature in the northern Scandinavian mountains increased approximately by 2.0 °C, and winter temperature (January–February) by 3.0 °C, associated with an increasing trend in precipitation (Vuorinen et al. 2017). Significant increases in mean precipitation were also observed in the Norwegian Scandes between 1900 and 2014 (Vanneste et al. 2017). A south-to-north gradient in the magnitude of precipitation increase in the Scandes is projected for the next decades (Christensen et al. 2015).

America

Over most of North America, mean annual temperature has increased over the past century, with higher latitudes of Canada and Alaska experiencing the largest temperature anomalies and warming rates more than double the global rate (Fig. 1.6). Substantial warming has been observed since the 1970s, accompanied by decreases in frost days and cold spells, increases in the occurrence of severe hot events over the USA, and increases in extremely hot seasons in northern Mexico, the USA, and parts of Canada (Vincent and Mekis 2006; Kunkel et al. 2008; Melillo et al. 2014; Romero-Lankao et al. 2014; Bush and Lemmen 2019; Cuervo-Robayo et al. 2020). In western North America, twentieth-century observations show temperature increases over the entire mountain region, from the SW to Alaska, which are higher than the global average and range mostly between 1 and 2 °C, and with minimum temperatures increasing to a greater extent than maximum temperatures (Wagner 2009). Warming rates are considerably higher in winter than in summer, exemplified by mean temperature increase of 3.3 °C in winter, 1.7 °C in spring, 1.5 °C in summer, and 1.7 °C in autumn between 1948 and 2016 in Canada (Fig. 1.7) (Bush and Lemmen 2019). As in Scandinavia and North Asia, a crude south-to-north gradient of increasing warming rates is evident (Kittel et al. 2002), and, as in Asia and Europe, higher elevations show greater temperature increases than lower elevations (Minder et al. 2018). Twentieth-century annual precipitation trends are positive over the Rocky Mountain/Great Basin region, although not always significant, and with seasonally heterogeneous trends (Wagner 2009). Extreme precipitation events have become more frequent and more intense in recent decades (Kunkel et al. 2008).

Across the system of the American Cordilleras, the Alaskan and Yukon subregions have been warming at a faster rate than any other subregion (mean annual temperature increase of up to more than 3 °C in the past 70 years), with considerably more warming in winter than in summer (Chapin et al. 2014; Lader et al. 2016; Zhang X et al. 2019). In the Pacific Coastal and Rocky Mountain ranges of western Canada, precipitation has slightly increased in most seasons. However, a statistically significant decrease in winter precipitation has been observed (Zhang X et al. 2019). Over recent decades (1970–2012), observations in the Pacific Northwest and the northern Rocky Mountains of the USA show accelerated average warming rates of c. 0.2 °C per decade, associated with longer growing

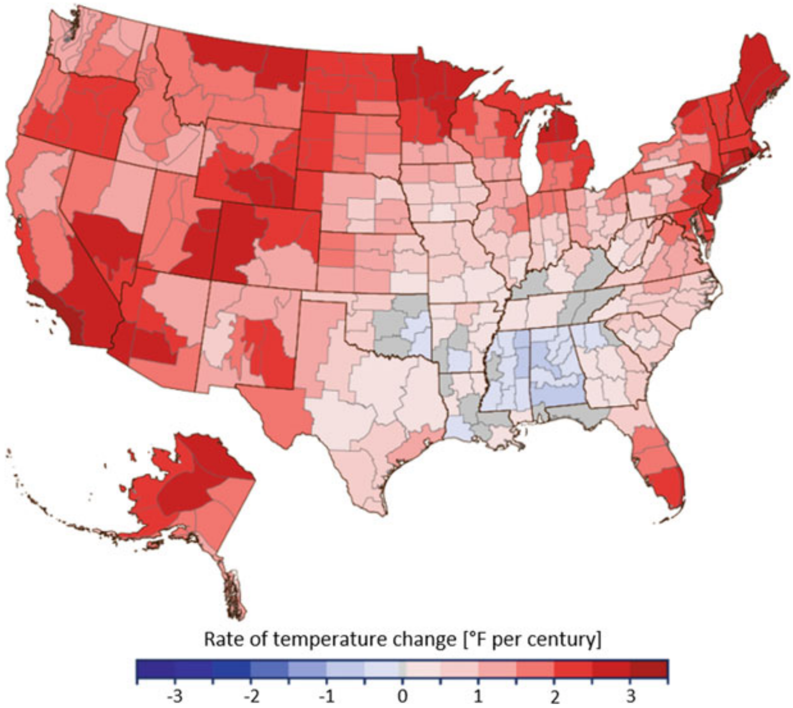


Fig. 1.6 Rate of temperature change in the United States 1901–2015 (modified from https://www.epa.gov)

seasons, increased evapotranspiration across the region, and increased climatic water deficits (Mote et al. 2013; Abatzoglou et al. 2014). In the southern Rocky Mountains and the Sierra Nevada, the decade 2001–2010 was the warmest in the 110-year instrumental record, with temperatures up to 1 °C higher than historic averages, with relatively higher spring and summer warming, fewer cold air outbreaks and more heatwaves, and with spatially varying precipitation trends (decreases in the southern part of the region, with strongest percentage declines during spring and summer, and increases in the northern part) (Hoerling et al. 2013; Garfin et al. 2014). Thus, it will get increasingly difficult to buffer drought effects in the southern mountainous regions of North America.

Significant warming, in the order of up to 1.0 °C since the 1970s, has also been detected throughout Central America and South America (Magrin et al. 2014). The tropical and subtropical Andes are being subjected to significant changes in mean climatic conditions, reflected in a mean temperature increase of about 0.1 °C per decade over the past 70 years (Fig. 1.8) (Bradley et al. 2006; Lavado Casimiro et al. 2013; Vuille 2013; Lopez-Moreno et al. 2016). Significantly positive temperature trends were also confirmed for the Patagonian Andes in the past century (Masiokas et al. 2008). After significant warming during much of the twentieth century, subtropical coastal regions experienced a recent cooling trend, in particular in central and northern Chile, related to the Pacific Decadal Oscillation (Falvey and Garreaud 2009). Higher elevations in the tropical Andes and further south to Central Chile, however, show continued warming of currently c. 0.2 °C per decade (Vuille et al. 2015). Temperatures at higher elevations are obviously now decoupled from the sea surface temperature forcing in the Pacific, which served as a strong predictor for cold or warm periods in the Andes in previous decades (Vuille et al. 2018). Irrespective of this, patterns of elevation-dependent warming have been observed throughout the Andes (e.g. Mora and Willems 2012; Ruiz et al. 2012, Schoolmester et al. 2018).

Precipitation trends are weaker and spatially much more heterogeneous. Stations in the Andes of Ecuador, Peru, and Bolivia showed a trend

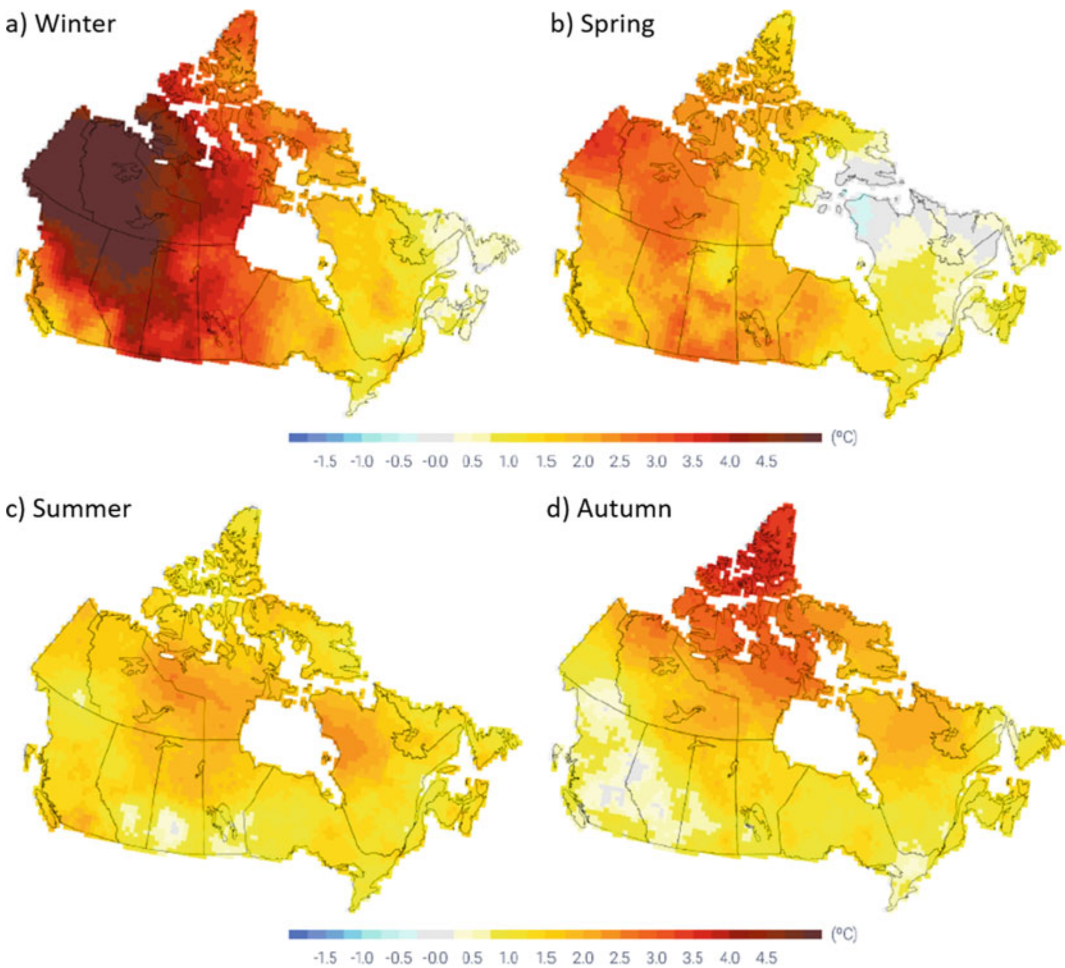


Fig. 1.7 Trends in seasonal temperatures across Canada; observed changes (°C) in seasonal mean temperatures between 1948 and 2016 for the four seasons. (Modified from Bush and Lemmen 2019)

towards increased precipitation north of ~11°S between 1950 and 1994, while most stations located further south showed a precipitation decrease (Vuille et al. 2003), also in Patagonia (Masiokas et al. 2008). However, precipitation trends are not significant over recent decades, and most of the variability in the data appears to be associated with the ENSO (El Nino Southern Oscillation) phenomenon (Lavado Casimiro et al. 2013; Salzmann et al. 2013; Rau et al. 2017). In general, climate anomalies such as ENSO and large-scale ocean-atmospheric indexes have a considerable influence on temperature and precipitation fluctuations in South America.

Africa

Across the continent of Africa, mean annual temperatures have increased by 0.5 °C or more in the past 50–100 years (Fig. 1.9), with minimum temperatures warming more rapidly than maximum temperatures, and temperature anomalies being significantly higher for the period 1995–2010 compared to previous decades (Toulmin 2009; Collins 2011; Niang et al. 2014). Observed and projected temperature rise is comparatively high in NW Africa, in particular in the Atlas Mountains. A very strong warming of about 6 °C is expected here in the course of the twenty-first century while the precipitation trend

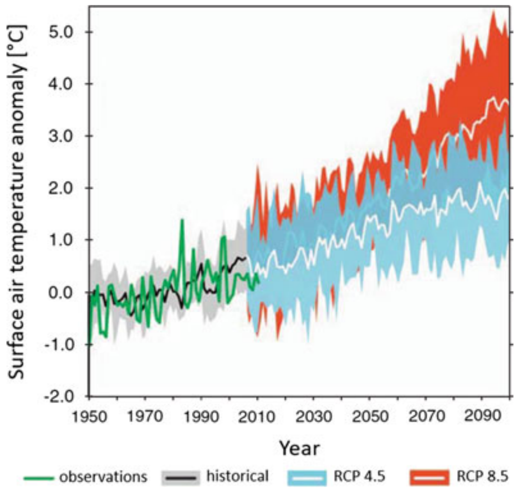


Fig. 1.8 Observed and simulated annual mean air temperature anomalies in the tropical Andes (departures from 1961–1990 mean) derived from station data (green, 1950–2010), historical CMIP5 (grey, 1950–2005), and future CMIP5 scenarios (light blue, RCP 4.5; red, RCP 8.5, 2006–2100). (Modified from Vuille et al. 2018)

is distinctly negative, leading to an earlier onset and longer duration of droughts (Patricola and Cook 2010; Bouchaou et al. 2011; Schilling et al. 2012). Mountains and highlands of East Africa also experienced significant warming over recent decades, up to 1.8 °C since 1950 (Jury and Funk 2013), while long-term precipitation trends are not significant, but rainfall is recently declining in some parts of the region (Anyah and Qiu 2012; Viste et al. 2013; Mengistu et al. 2014; Omondi et al. 2014). A recent increase of warming rates to 0.5 °C per decade was reported for the Rwenzori Mountains in Uganda (Taylor et al. 2006). In Ethiopia, Kenya, and Tanzania, increases in maximum and minimum temperatures are accompanied by increasing trends in warm nights, warm days, warm spell days, and mostly a non-significant change in precipitation indices (Gebrechorkos et al. 2019). Ethiopia's eastern highlands, however, experience significant climate-induced drought and stress on crop and livestock productivity, while large regions of western Ethiopia are becoming wetter (Brown et al. 2017). Most of southern Africa has also experienced significant warming over recent decades (Kruger and Sekele 2013), with marked recent temperature increases in the Drakensberg system (Morris 2017).

1.2.2 Impacts on the Cryosphere and Hydrosphere

1.2.2.1 General Overview

Over recent decades, considerable changes have been observed in cryospheric components (snow, ice, glaciers, permafrost) in mountains of the world that serve as vivid illustrations of mountains being at the forefront of climate change impacts (Hock et al. 2019; Pihl et al. 2019). Changes in cryospheric land conditions potentially induce important albedo feedbacks to the regional and global climate. Climate warming causes cascading effects on cryospheric and related hydrological processes that affect not only mountain catchments but also the lowlands. The cascade of effects extends to human livelihoods, economy, and ecosystems. Widespread changes of the cryosphere and associated changes in water cycle and balance and river discharge regimes have inevitable consequences for erosion rates, sediment and nutrient fluxes, and the biogeochemistry of rivers and lakes, and finally for water quality, aquatic habitats, and respective biotic communities (Huss et al. 2017). Changes of the cryosphere also affect terrestrial communities and ecosystems significantly, for instance, by creating new habitats in glacier forefields, by modifying the length of the growing season and the phenology of plant production and consumers, and by altering soil moisture conditions and nutrient availability. Ultimately, ecosystem functioning is affected due to a novel constellation of site conditions and competitive relationships, and associated changes in species compositions and primary productivity. Water supply from the cryosphere is indispensable for socio-economic systems in both mountains and lowlands. Meltwater from snow and ice is essential for drinking water supplies, irrigated agriculture, mining, hydropower generation, industries, tourism, and other activities (Beniston and Stoffel 2014; Huss et al. 2017).

The snow cover is the largest cryosphere component. Global observations show that climate change has caused a general reduction in

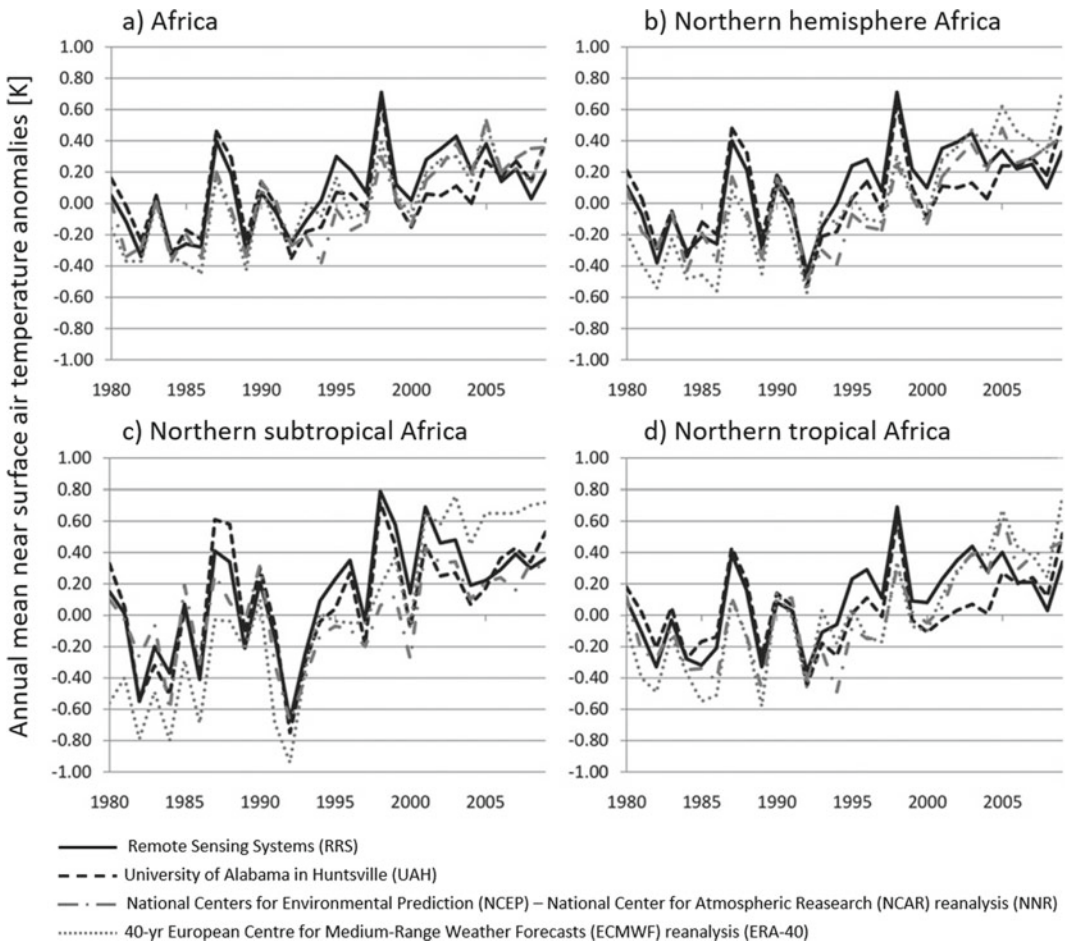


Fig. 1.9 Annual mean near-surface air temperature anomalies (K) between 1979 and 2010 for Africa and selected subregions, with black lines indicating satellite data (solid: RSS; dashed: UAH), and grey lines reanalysis data (dashed-dotted: NNR; dotted: ERA-40). (Modified from Collins 2011)

low-elevation snow cover in recent decades (Fig. 1.10) (Bormann et al. 2018). In nearly all mountain regions around the globe, snow cover duration (SCD) has declined, particularly at lower elevations, with an average decline rate of 5 days per decade (Hock et al. 2019). Snow-covered area (SCA) and snow depth are also decreasing significantly, albeit with high year-to-year variation. Snow cover will further decline in the next decades, a decrease by 10–40% is expected for the period 2031–2050 compared to 1986–2005 (Hock et al. 2019). On the other hand, increased snowfall will occur at higher elevations where the rain/snow partitioning is no longer affected by rising temperatures, and where total winter precipitation is increasing (Kapnick and Delworth 2013). Snow accumulation is critical for water availability in many regions. Such snow-dependent regions are expected to experience increasing stress from the imminent shift towards low snow years within the next three decades and from extreme changes in snow-dominated water resources (Diffenbaugh et al. 2013).

As key indicators and unique demonstration objects of ongoing climate change, glaciers have attracted tremendously increased scientific

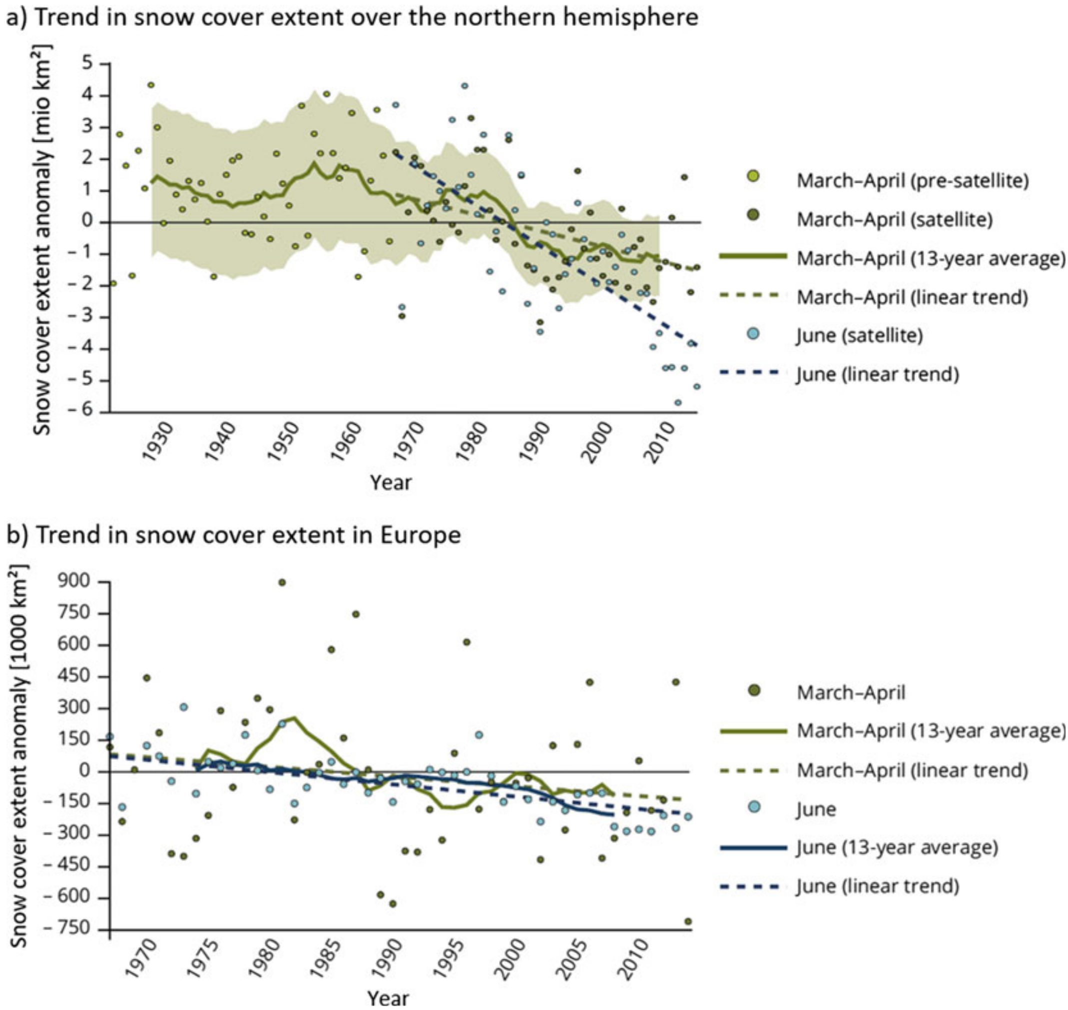


Fig. 1.10 Satellite-derived trends in snow cover extent over the northern hemisphere and Europe 1967–2015; the time series for the northern hemisphere is extended back to 1922 by including reconstructed historical estimates. (Modified from EEA 2017)

interest and accelerating international media attention. Numerous new records of annual mass loss were observed in the past two decades, indicating implications for the water cycle that affect continental-scale water supply and even global-scale sea levels. Glacier mass loss provides a more direct evidence of climate change in remote mountains where meteorological observations are hardly available. Global glacier recession is accelerating (Fig. 1.11), with atmospheric warming considered to be the primary driver, modified by other meteorological variables and internal glacier dynamics (Marzeion et al. 2014; Vuille et al. 2018; Hock et al. 2019). Over the last decades, declines in glacier area, length, and mass have condensed to a globally widely coherent picture of mountain glacier recession, albeit with interannual and regional variations (Zemp et al. 2015). At a global scale, glacier mass loss increased by c. 30% between 1986–2005 and 2006–2015 (Zemp et al. 2019). During the latter period, mountain glaciers lost about 500 kg of mass per square metre per year, a total of 123±24 Gt (billion

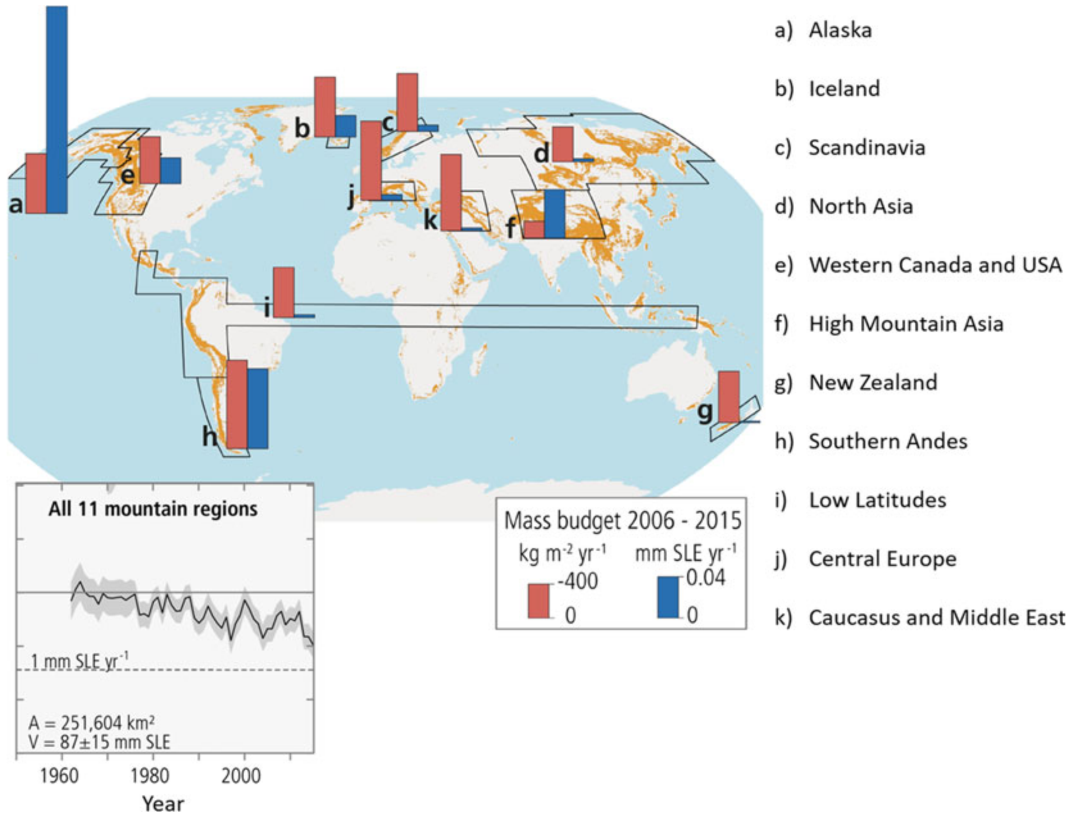


Fig. 1.11 Glacier mass budgets for eleven mountain regions; red and blue bars refer to regional budgets averaged over the period 2006–2015 in units of kg m^{-2} yr^{-1} and mm sea-level equivalent (SLE) per year, respectively. (Modified from Hock et al. 2019)

tonnes) per year (excluding the Arctic and Antarctic) (Hock et al. 2019; Pihl et al. 2019). Most negative glacier mass budgets were observed in the Southern Andes, Caucasus/ Middle East, European Alps and Pyrenees, with total mass loss and corresponding contribution to sea level between 2006 and 2015 being largest in Alaska, followed by the Southern Andes and High Asia (Hock et al. 2019). Notwithstanding the global trend of glacier recession, glaciers in various mountain ranges have shown intermittent re-advances or mass gains due to locally restricted climatic causes or internal glacier dynamics (WGMS 2008). Century-scale projections for mountain glaciers show substantial mass loss by 2100 relative to 2015 in the order of 18% for scenario RCP2.6 and 36% for scenario RCP 8.5 (Hock et al. 2019).

Permafrost is another important component of the cryosphere in high mountain regions, in particular in the Northern Hemisphere. Mountain permafrost accounts for c. 25–30% of the global permafrost occurrence, its distribution is spatially highly heterogeneous (Hock et al. 2019). It significantly influences energy balance, terrain stability-related geophysical hazards, ground and subsurface hydrology, water quality, river sedimentation, and infrastructure. Permafrost degradation due to global warming contributes to mountain slope destabilization and increased mass-movements and related hazards (Haeberli et al. 2017; Patton et al. 2019). As the understanding of permafrost depends on ground and subsurface temperature observations, which are logistically demanding and expensive, it remains largely understudied in many mountain ranges. At a global scale, mountain permafrost warming has been shown to accelerate recently (Fig. 1.12) and to exceed values of the late twentieth century, with an average warming rate of 0.19 °C per decade between 2007 and 2016 (Biskaborn et al. 2019), while general warming, ground-ice loss and permafrost degradation has been observed over longer time periods (e.g. Cao et al. 2018; Noetzli et al. 2018; Mollaret et al. 2019). In general, temperature increase in colder permafrost was greater than in warmer permafrost. Mountain permafrost is expected to undergo increasing thaw and degradation during the twenty-first century, projections reveal increased loss of permafrost under stronger atmospheric warming (Hock et al. 2019).

Changes in the cryosphere have wide-ranging consequences for freshwater availability in both mountain and downstream regions since streamflow timing and magnitude is largely controlled by the meltwater supply from cryospheric components (Rasul and Molden 2019). Runoff from alpine catchments is particularly critical for the water supply in summer months when other water sources in the lowlands are often limited. With regard to climate-cryosphere-hydrosphere interactions in mountain regions, reduced ice and snow cover triggers major shifts in seasonal runoff regimes. In snow and glacier-dominated river basins, recent observations indicate emerging trends of increased average winter runoff, earlier spring snowmelt runoff peaks, and declining summer runoff in many basins. A decreasing ratio of snow to rainfall, increased snowmelt, and local/regional precipitation increases contribute to increased winter runoff, while less snowfall and decreasing glacier melt after peak water result in lower summer runoff. Peak water in glacier-fed rivers (the turning point from annual glacier runoff increases to declines) has already passed in mountain regions with predominantly smaller glaciers (e.g. tropical Andes, Canadian Rocky Mountains, European Alps), while glacier runoff will continue to increase in the next decades in mountain catchments with large ice volumes (northern North America, parts of the HKH region, Central Asia) where peak water will be reached in the late twenty-first century (Huss et al. 2017; Huss and Hock 2018; Hock et al. 2019; Hoelzle et al. 2019).

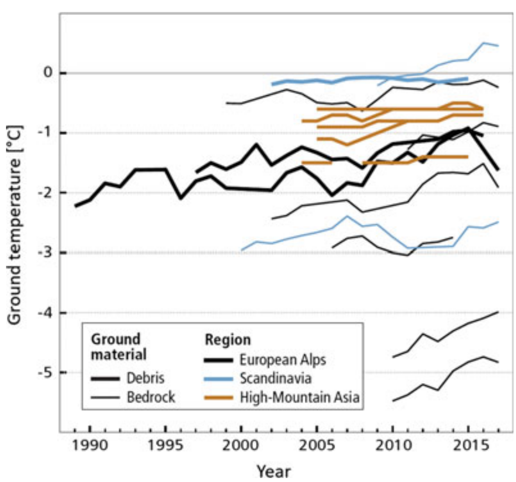


Fig. 1.12 Mean annual ground temperature from boreholes in debris and bedrock in the European Alps, Scandinavia and High Mountain Asia; the depth of measurements is approximately 10 m. (Modified from Hock et al. 2019 after Noetzli et al. 2018)

1.2.2.2 Regional Overview

Asia and Australasia

Although comprehensive observations on snowpack parameters in Asian mountains are still limited, growing and ample evidences from satellite-based global to local studies suggest that the snow cover has significantly declined, particularly since the 1960s (Dietz et al. 2013; Rohrer et al. 2013; Singh et al. 2014; Bolch et al. 2019). The HKH and Tibetan regions show overall negative trends in snow accumulation rates (Bolch et al. 2019). Over the period of 2000–2010, the annual (−1.25%) and seasonal snow-covered area (−1.04 to −0.01%) decreased, except for the autumn season (5.6%) (Gurung et al. 2011). However, westerly dominated basins (Indus basin, NW Himalaya) show increases in winter snow cover (Bolch et al. 2019; but see also Li et al. 2018). Increasing snow-covered area trends in the Karakoram/NW Himalaya contrast with declining trends in the Ganga and Brahmaputra river basins (Singh et al. 2014; Bilal et al. 2019). Declining trends of annual and seasonal snow-covered area were also assessed for southern slopes of NW Himalayan river basins (Jhelum and Shyok to Satluj and Beas), except for winter seasons over 2001–2012 (Sharma et al. 2014). Barman and Bhattacharjya (2015) reported a declining snow-covered area trend in the Brahmaputra river basin, except in winter seasons between 2002 and 2012. A slight decline (0.01% a^{-1}) over the Tibetan Plateau has been observed since the early 2000s (Duo et al. 2014; Li et al. 2018). Based on long-term data (1972–2017), Bormann et al. (2018) found overall declining trends in High Asia, with a slight increase in the Karakoram and in the East Himalaya. In the Siberian region including Kamchatka, the snow-covered area has declined significantly (0.8 × 104 km^2 a−^{1}) over 1970–2012 (Yu et al. 2017). Distinctly declining trends (up to 0.8 × 10^2 km^2 a^{-1}) were assessed for the Pamir, Alay, and Altai over 2000–2015, while the Tien Shan and Kunlun show mixed trends (Dietz et al. 2013; Liu J et al. 2017). In the Tien Shan, negative trends in summer (−0.02% a^{-1}) and winter (−0.1% a^{-1}) contrast with positive trends in spring and autumn (0.1% a^{-1}) (Tang et al. 2017). Another significant decrease in snowpack parameters was detected in the Zagros Mountains and in the Greater Caucasus (Notarnicola 2020).

Snow cover duration, as affected by the precipitation and temperature changes in pre- and post-winters, has decreased in the HKH region, Tien Shan, Kunlun, Altai, and Kamchatka by up to 30 days per decade between 1982 and 2013 (Bulygina et al. 2009; Dietz et al. 2013; Tang et al. 2013; Ye and Cohen 2013; Chen et al. 2016). A large decrease of snow cover duration (4 days a^{-1} between 2000 and 2015) was detected in the Nyainqentanglha Mountains (SE Tibet) (Wang et al. 2017; Notarnicola 2020). By contrast, increases were reported from NE Tibet and some Siberian mountain ranges (Chen et al. 2016). Significantly decreasing snow cover and snow duration is projected for the Southern Alps in New Zealand and alpine regions in Australia (Hennessy et al. 2008; Hendrikx et al. 2012).

Glaciers across Asia have experienced sustained mass loss since the mid-nineteenth century, with accelerated loss in recent decades, except for some of the glaciers in the Karakoram, Pamir, Kunlun, Tien Shan, and Kamchatka which have not changed significantly or, in case of surge-type glaciers, have shown area increases. Recent estimates of total glacier mass change in High Mountain Asia are in the order of −19.0 ±2.5 Gt yr^{-1} for the period 2000–2018, with greatest total mass loss across the Himalayas, Nyainqentanglha, and the Tien Shan and positive mass balance in the western Kunlun Shan and eastern Pamir (Fig. 1.13) (Brun et al. 2017). The average glacier area loss in the entire HKH region was estimated at 0.35% a^{-1} between 1970 and 2000; the rate increased to 0.42% a^{-1} between 2000 and 2010 (Bolch et al. 2019). Simultaneously, the glacier mass balance rate has increased from −0.26 (m w.e.[water equivalent]$^{-1}$) (1970–2000) to −0.37 (m w.e.$^{-1}$) in 2000–2010, with some regional variations and even anomalies (Azam et al. 2018; Bolch et al. 2019). The Imja–Lhotse Shar glacier in the Khumbu region in Nepal showed an exceptionally large loss rate of −1.45±0.52 m w.e. yr^{-1} for 2002–2007, with enhanced ice losses by calving into the Imja Lake (Bolch et al. 2011). There is a strong E-W gradient of glacier retreat, with average glacier area change rates of −0.81% a^{-1} in the eastern Himalaya decreasing to −0.37 and −0.34% a^{-1} in the central and western Himalaya between 2000 and 2010; area loss rates slightly slowed down in the central and western Himalaya, while an increase was observed in the eastern Himalaya during this period (Bolch et al. 2012, 2019; Azam et al. 2018). On the contrary, glacier area changes in the Karakoram show a divergent pattern that is known as the 'Karakoram anomaly' (Hewitt 2005, 2007). Non-surge-type glaciers were relatively stable and surge-type glaciers showed large increases as well as decreases over the past decade (Bhambri et al. 2017; Azam et al. 2018; Bolch et al. 2019). Accordingly, most Karakoram glaciers had a positive mass balance in recent decades (Kääb et al. 2012, 2015; Gardelle et al. 2013; Pratap et al. 2016; Berthier and Brun 2019; Shean et al. 2020). The glacier mass balance anomalies in the HKH region can be explained by contrasting meteorological conditions, reflected in differing energy balances, accumulation regimes and melt dynamics at high elevations (Bonekamp et al. 2019), but the understanding is far from complete (Farinotti et al. 2020). Strong variations in glacier mass balances in High Mountain Asia vividly illustrate that the sensitivity of glaciers to climate change is regionally variable.

The spatial patterns of the terminus change rates of glaciers (>−80 to >80 m a^{-1}) across the HKH correspond to glacier area changes. Over recent decades, glacier terminus recession rates have been assessed to be highest in the eastern Himalaya, while a considerably lower glacier recession is observed in the central and western Himalaya, and partially a surging/advancement (up to 2.5 km) in the Karakoram (Hewitt 2007; Quincey et al. 2015; Mal et al. 2016; Bhambri et al. 2017; Azam et al. 2018). Recently, recession rates of large glaciers in the central and western Himalaya (Gangotri, Milam, Bara Shigri) slowed down (Bhambri et al. 2012; Bhattacharya et al. 2016; Chand et al. 2017; Mal et al. 2019), while the glaciers of the NW Himalaya showed variable, often lower change rates or were relatively stable (Schmidt and Nüsser 2009, 2012; Chand and Sharma 2015; Chudley et al. 2017). Nevertheless, over longer time scales significant glacier retreat and thinning becomes obvious, as exemplified by the Chungpare Glacier at Nanga Parbat (Nüsser and Schmidt 2017). The average glacier area loss rate on the Tibetan Plateau was estimated to be slightly lower (0.27% a^{-1}, with <1.5% of glaciers advanced) compared to the surrounding regions between 1970 and 2009, with higher rates in the SW and SE, and lower rates in the inner, W, NE, E and N parts of the plateau (Bolch et al. 2010b; Wei et al. 2014; Ye et al. 2017). Glacier recession has fragmented larger glaciers into smaller ones, the number of glaciers in Nepal and Bhutan, for instance, increased by 11% and 15% (24% and 23% area loss), respectively, between 1980 and 2010 (Bajracharya et al. 2014a, b). Likewise, a distinct increase in number and area of moraine-dammed glacial lakes was assessed in recent decades, formed due to thinning, flow stagnation and recession of glacier tongues, and fed by glacier meltwater (Fig. 1.14) (Gardelle et al. 2011; Somos-Valenzuela et al. 2014; Zhang et al. 2015; Krause et al. 2019). Hence, glacial lake outburst floods (GLOFs), which have resulted in catastrophic damages and fatalities in the past decades, pose an increasing risk, with the southern Himalaya being a GLOF hotspot region (Fig. 1.15) (Nie et al. 2017; Veh et al. 2019). GLOF frequencies are predicted to increase during the next decades (Harrison et al. 2018). Projections for different RCP scenarios show that much of the glacier ice in High Mountain Asia will disappear towards the end of the century, with potentially serious consequences for regional water management and mountain communities (Kraaijenbrink et al. 2017; Mukherji et al. 2019; Immerzeel et al. 2020). Decreasing water supplies from cryosphere change will affect particularly irrigation-dependent agriculture in the Indo-Gangetic Plains (Biemans et al. 2019) and in arid mountain regions, where local farmers are forced to develop adaptive strategies (Nüsser et al. 2012, 2019a, b; Parveen et al. 2015; Rasul et al. 2020).

Siberian mountains have experienced a substantially high glacier area loss since 2000 (3.4% a^{-1}) compared to the low recession rate since the Little Ice Age (0.29% a^{-1}) (Osipov and Osipova 2014). In Kamchatka, the average glacier area loss rate was 0.33% a^{-1} between 1950 and 2000 (Khromova et al. 2014); it increased substantially to 1.7% a^{-1} in recent years, leading to the disappearance of 46 glaciers between 2000 and 2014 (Lynch et al. 2016). Glacier reductions on the Kamchatka Peninsula range from 10 to 70% over recent decades (Khromova et al. 2019). The area shrinkage of glaciers in the Altai, the Urals and the Tien Shan is also remarkably high (between 0.32 and 0.62% a^{-1}) over the period from the 1950s until recently, associated with respective negative mass balance rates (Shahgedanova et al. 2010; Khromova et al. 2014; Farinotti et al. 2015; Wei et al. 2015; Ganyushkin et al. 2017; Zhang et al. 2017; Barandun et al. 2018). In the Chinese part of the Tien Shan, 182 glaciers vanished in recent decades (Baojuan et al. 2017), some glaciers, however, have shown advances (Shangguan et al. 2015). Even higher recession rates were assessed in the Pamir Alay

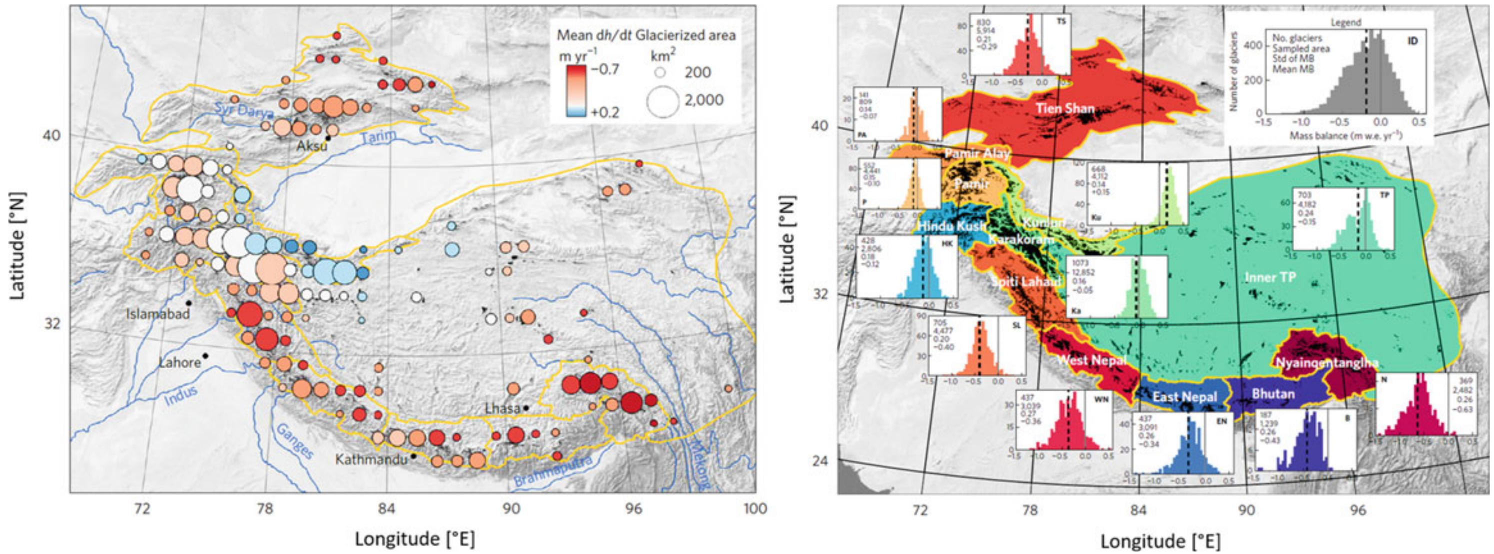


Fig. 1.13 Spatial pattern of glacier elevation changes and mass balance for High Mountain Asia (2000–2016). Left panel: Regional glacier mean elevation change on a 1° × 1° grid. Right Panel: Region-wise distribution of glacier-wide mass balance for every individual glacier (>2 km^2), represented in histograms of the number of glaciers (y-axis) as a function of mass balance (x-axis in m w.e. yr^{-1}); the black dashed line represents the area-weighted mean; numbers denote the total number of individual glaciers, the corresponding total area in km^2, the standard deviation of their mass balances and the area-weighted average mass balance in m w.e. yr^{-1}. (Modified from Brun et al. 2017)

Fig. 1.14 The fast retreat of Himalayan glaciers has resulted in the formation and expansion of meltwater lakes as in the former snout area of Gangapurna glacier (3550 m), Nepal, creating risks from GLOF events. (*Photo* © Udo Schickhoff, September 23, 2013)

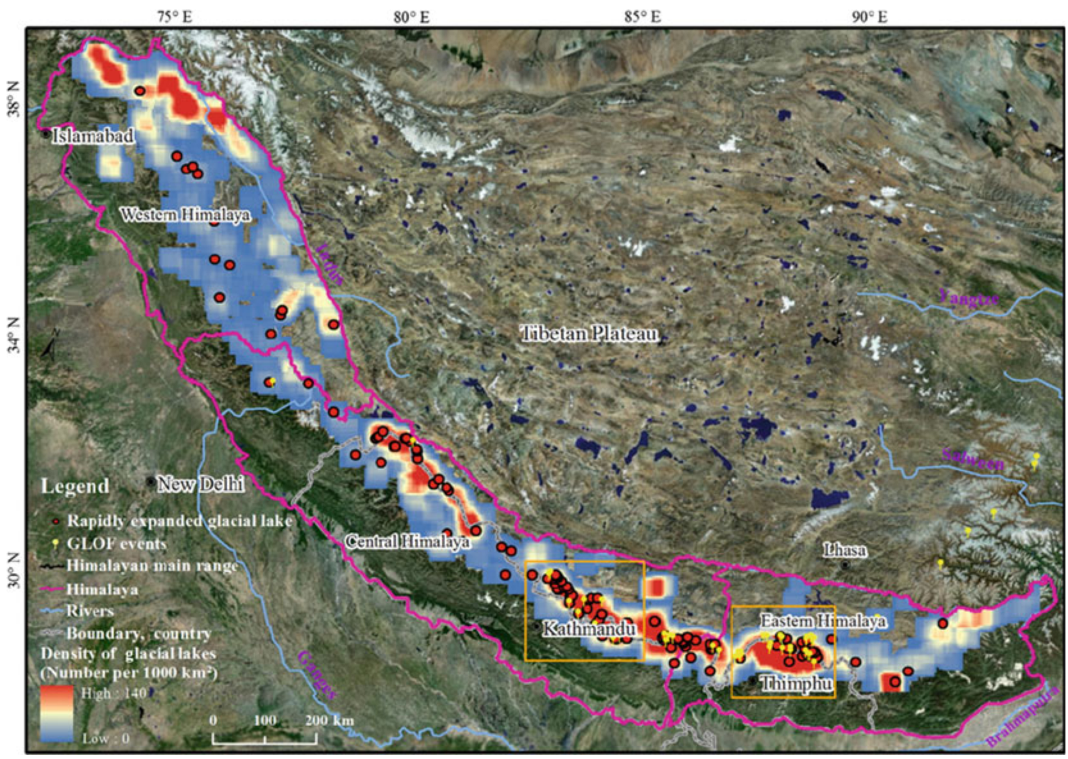


Fig. 1.15 Glacial lakes in the Himalaya in 2015: Spatial distribution of rapidly expanded glacial lakes and historical GLOF events in the Himalaya (potential vulnerable areas of GLOFs in orange boxes). (Modified from Nie et al. 2017)

(0.84% a^{-1} over the period 1978–2001) (Khromova et al. 2014), where a total of 142 glaciers disappeared (Holzer et al. 2016), while some fluctuations are also observed (Bolch et al. 2019). Recent glacier area loss rates in the Caucasus increased to 0.69% a^{-1} between 1986 and 2014 (Tielidze and Wheate 2018).

Tropical glaciers in Australasia show a dramatic recession over recent decades. The glacier areas on Puncak Jaya (4884 m a.s.l.), the highest mountain on the island of New Guinea, were found to decrease by 85% between 1988 and 2015 (Veettil and Wang 2018a), suggesting that these tropical glaciers might disappear before 2050 (Veettil and Kamp 2019). Specific climate conditions may result in exceptional terminus advance of some glaciers, opposed to the global trend. This is the case in New Zealand where several maritime glaciers advanced between 1983 and 2008, including the famous Franz Josef and Fox glaciers, which are steeply inclined and react swiftly and similarly to climate forcing. The glacier advance phase resulted predominantly from discrete periods of reduced air temperature, associated with anomalous southerly winds and low sea surface temperature in the Tasman Sea region (Mackintosh et al. 2017; see also Cullen et al. 2019). Nevertheless, the total ice volume of the Southern Alps for the small and medium glaciers has decreased from 26.6 km^3 in 1977 to 17.9 km^3 in 2018 (a loss of 33%), with accelerating ice loss for the period 1998–2018 (Salinger et al. 2019). Particularly, gentle-sloping, debris-covered glaciers with terminal lakes in the Southern Alps are in decline, as exemplified by the Tasman Glacier which has undergone c. 5 km of retreat into a terminal lake since the early 1980s (Dykes et al. 2011).

Permafrost research in High Mountain Asia is still limited. Nevertheless, there is growing evidence of permafrost warming and degradation. In the extended HKH region, permafrost research has focused on the Tibetan Plateau. It is generally assumed that most permafrost has undergone warming and thaw in recent decades (Zhao et al. 2010; Gruber et al. 2017; Bolch et al. 2019). The Tibetan Plateau is estimated to have the highest decadal permafrost area loss in the northern hemisphere, considerably increasing from 1×10^4 km^2 over the period 1901–2009 to 9×10^4 km^2 between 1979 and 2009 (Guo and Wang 2017). Thermal degradation of permafrost and increasing thickness of the active layer is widespread in Tibet, affecting c. 88% of the permafrost area of the 1960s (Ran et al. 2018). Local studies on the Himalayan South Slope suggest widespread permafrost degradation and the rise of permafrost lower limits by several hundreds of metres since the 1970s (Fukui et al. 2007; Allen et al. 2016). Significant warming and associated degradation of permafrost were also ascertained for Siberian and Mongolian high mountains and the Tien Shan (Marchenko et al. 2007; Sharkhuu et al. 2007; Guo and Wang 2017; Liu G et al. 2017; Biskaborn et al. 2019; Munkhjargal et al. 2020). In New Zealand, a connection between degrading permafrost and the occurrence of rock avalanches and other landslides is suspected (Allen et al. 2011).

Both climate change and anthropogenic activities, especially hydropower projects and irrigation, have significantly affected the hydrology in Asian mountains (river discharge, hydrological budgets) during the past century (Bhutiyani et al. 2008; Xu et al. 2009; Haddeland et al. 2014; Singh S et al. 2016; Scott et al. 2019). River runoff in eastern and Central Asian river basins decreased up to 15% during 1971–2000, even succeeded by the northwestern HKH, Pamir, Kunlun Shan, Qilian Shan, and Caucasus where the runoff decreased by 15–30% during the same period (Haddeland et al. 2014). Hydrological changes that have only been triggered by climate change are difficult to assess in detail due to, inter alia, poor understanding of the role of snow and ice in the regime of catchment basins, interannual variability of meteorological conditions, hardly available long-term series of river discharge, and multiple factors influencing streamflow. Trends may change in space and time within single basins, thus, conclusive evidence of either declining or increasing streamflow trends in the extended HKH region cannot yet be provided (Scott et al. 2019). Nevertheless, several review-based and observational studies on glacier- and snow-fed major basins indicate that river runoff has increased in some basins (Brahmaputra, Salween, Mekong), has no significant change/spatio-temporal mixed responses (Indus, Yangtze), and has decreased in others (Ganges, Yellow River) (Xu et al. 2009; Shrestha and Aryal 2011; Miller et al. 2012; Singh S et al. 2016; Hasson et al. 2017; Scott et al. 2019). Glacierized basins on the Tibetan Plateau show increased discharge, correlated to increased summer and winter temperatures and earlier snowmelt (Ye et al. 2005; Yao et al. 2007; Lin et al. 2008). Modelling studies for the HKH region predict shifts in the timing and magnitude of streamflows, but no significant changes or not more than minor increases in overall annual flows (Immerzeel et al. 2013; Lutz et al. 2014). In general, runoff in catchments with large ice volumes is projected to increase in the next decades indicating later peak water while basins with smaller ice volumes will face a decrease in runoff indicating earlier peak water (Fig. 1.16) (Hock et al. 2019).

The pattern of heterogeneous streamflow responses has been observed in other Asian mountain ranges and basins as well. Contrasts between individual basins become obvious when basins of the HKH region (Indus, Ganges, Brahmaputra) with small melt-to-discharge ratios due to the coincidence of glacier melt season and monsoon season are compared with Central Asian watersheds with a summer-dry climate where glacier melt substantially contributes to

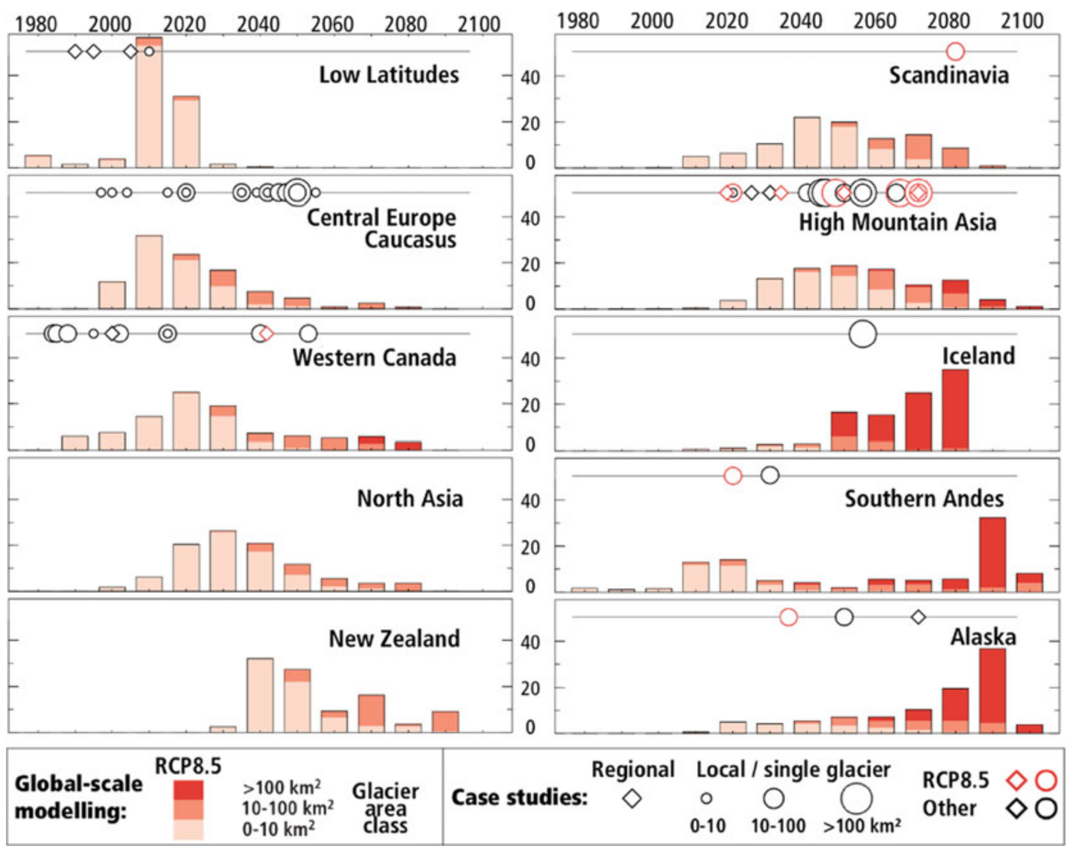


Fig. 1.16 Timing of peak water from glaciers in different regions under the RCP8.5 scenario; shadings of the bars distinguish different glacier sizes indicating a tendency for peak water to occur later for larger glaciers; circles mark timing of peak water from individual case studies, and refer to results from individual glaciers regardless of size or a collection of glaciers covering <150 km^2 in total, while triangles refer to regional-scale results from a collection of glaciers with >150 km^2 glacier coverage (Modified from Hock et al. 2019)

streamflow in July and August (Huss et al. 2017). River discharge in the glacier-dominated Aksu basin (Tien Shan) has increased in summer and winter over the past 50–60 years (Chen et al. 2006; Krysanova et al. 2015; Duethmann et al. 2015), while downstream stations at the main Tarim River show declining trends due to human abstraction of water (Tao et al. 2011). Declining snow cover thickness and duration in the central and western Tien Shan is associated with a decrease in river runoff (Aizen et al. 1997). Increased discharge volumes are reported for the Pamir (Chevallier et al. 2014), also for the northern Caucasus (Rets et al. 2018), and for the Southern Alps of New Zealand (Gawith et al. 2012). Discharge has recently decreased in some Siberian and Mongolian basins (Frolova et al. 2017; Dorjsuren et al. 2018).

Europe

Over recent decades, changes in the mountain cryosphere have already affected landscapes, hydrological regimes, water resources, and infrastructure, with significant downstream impacts in terms of quantity, seasonality, and quality of water (Beniston et al. 2011). Impacts related to climate-cryosphere interactions will continue to cause changes to such an extent that Europe's mountain landscapes will have a completely different visual appearance by the end of the twenty-first century. Seasonal snow lines will shift to much higher elevations, glaciers at low- and mid-range elevations will have disappeared, and even large valley glaciers will be characterized by significant retreat and mass loss (Beniston et al. 2018).

Numerous long-term observations in the European Alps show significantly negative current snow cover trends below 2000 m a.s.l. and negative or no clear trends above 2000 m, while the decadal variability of the snow cover is strong (Fig. 1.17) (Scherrer et al. 2004, 2013; Durand et al. 2009). Recently, Klein et al. (2016) detected a marked decline in all snowpack parameters over the period 1970–2015 irrespective of elevation, with significantly shortened snow cover duration by 8.9 days per decade on average which is largely driven by earlier snowmelt. Marty et al. (2017) provided evidence of a large-scale decline in snow water equivalents, while Schöner et al. (2019) found a clear decrease in mean snow depth over much of the Austrian and Swiss Alps. Similar trends are observed in the Tatra Mountains (Gadek 2014). The existence of a permanent snow cover during summer is very unlikely towards the end of the century, even at the highest elevations in the Alps (Beniston et al. 2018). This has profound implications for the remaining glaciers (Figs. 1.18, 1.19) that have already experienced a substantial mass loss since the nineteenth century and will face an increasing pace of mass loss (Zemp et al. 2015). The ice volume loss in the European Alps is estimated to be c. 50% during the period 1900–2011 (Huss 2012), while the glacier area in Switzerland decreased by 28% between 1973 and 2010 (Fischer et al. 2014), and in Austria by 17% between 1969 and 1998 (APCC 2014), resulting in the disintegration of many glaciers. The reduction in glacier area is even more critical in the case of the small glaciers in southern Europe. In the Pyrenees, Rico et al. (2017) assessed a decline of the glacier area by 88% between 1850 and 2016, with a rapid wastage since the 1980s, confirming the recently accelerated shrinkage trend. Small glaciers in temperate and southern Europe are likely to completely disappear, and even large valley glaciers will have lost much of their current volume by the end of the century (Jouvet et al. 2009; Linsbauer et al. 2013; Zekollari et al. 2014, 2019).

In Norway, Dyrrdal et al. (2013) observed a decrease in snow depth and number of snow days at lower elevations and in regions with warmer winter climate since the early 1960s, only some stations in higher mountain regions show positive trends, in particular in colder regions in the western part of South Norway. Declining snow depths at lower elevations and a shortened duration of snow cover was also assessed in northern Finland and related to large-scale climatic indices (Kivinen and Rasmus 2015). The glacier area in Norway has been reduced by c. 10% between 1960 and the 2000s (Winsvold et al. 2014). While the mass balance of

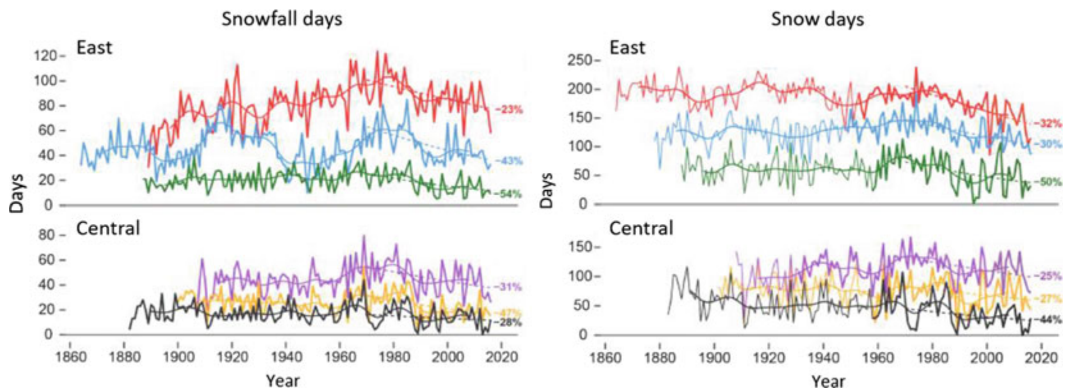


Fig. 1.17 Number of days with snowfall (daily new snow sum ≥ 1 cm) and number of days with snowpack (daily snow height ≥ 1 cm) at Swiss stations; the annual values are shown as a bold line; the thin line represents a 20-year Gaussian smoother. Top: Eastern Switzerland stations (Sils-Maria: red, Elm: blue, Chur: green). Bottom: Central Switzerland stations (Einsiedeln: purple, Meiringen: orange, Luzern: black). The dashed lines and numbers show the linear trends in the period 1970–2016. (Modified from CH2018 2018)

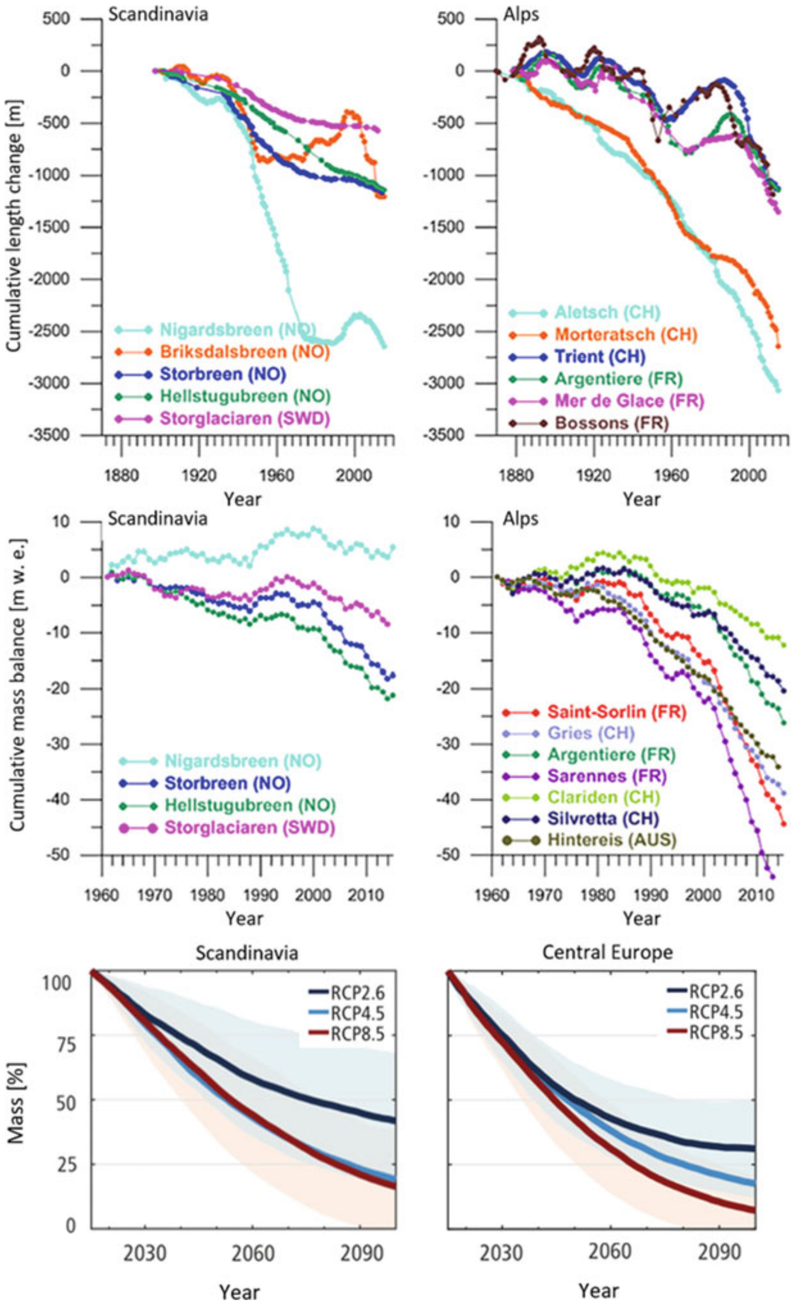
Norwegian glaciers is generally negative in the past 50–60 years with the decade 2001–2010 being the most negative, many maritime glaciers showed intermittent periods of positive mass balance in the late 1980s and 1990s due to higher snow accumulation (Andreassen et al. 2016, 2020), linked to the positive NAO (North Atlantic Oscillation) phase during that period (cf. Bonan et al. 2019). Massive volume losses in the order of 64–81% are predicted for the Scandinavian glaciers for the twenty-first century, some ice caps might lose up to 90% of their current volume, and many glacier tongues will disappear (Beniston et al. 2018).

Direct temperature monitoring and indirect geophysical surveys show accelerated permafrost warming in the Alps and in Scandinavia over recent decades. At monitoring sites in the Alps, the current mean annual ground temperature trend (10–20 m depth) is up to 1.0 °C per decade (Noetzli et al. 2018; Hock et al. 2019). In South Norway, mean ground temperature increase at 6.6–9.0 m depth ranged from $\sim$0.015 to $\sim$0.095 °C a^{-1} between 1999 and 2009 (Isaksen et al. 2011). Increasing permafrost temperatures and observed expansions of active-layer thickness (PERMOS 2016) suggest ongoing permafrost degradation, resulting in an increased frequency of slope instabilities in mountain ranges and to a higher magnitude of mass wasting processes such as rockfalls, rockslides, icefalls, landslides, and debris flows (Stoffel et al. 2014; Patton et al. 2019). Changes in the cryosphere of Europe's mountains will have severe hydrological implications, including a transition of runoff regimes from glacial to nival and from nival to pluvial, as well as shifts in the timing of discharge maxima (Beniston et al. 2018). In glacierized catchments, the glacier melt contribution to runoff will be reduced significantly by the end of the century, with peak discharge occurring 1–2 months earlier in the year (Hanzer et al. 2018). The altered seasonality of high-elevation water availability will have serious consequences for water storage and management in reservoirs for drinking water, irrigation, and hydropower production (Beniston et al. 2018).

America

Alaska has been one of the regions on Earth with highest warming rates over recent decades, with temperature increase being more than twice as high as in the contiguous United States. As a consequence, Alaska experienced a considerable decrease of the snow cover and a significant shrinkage of the ice mass of most of its glaciers, still accounting for 12% of the global ice-covered area outside the Antarctic and Greenland ice sheets (Kienholz et al. 2015). More than 90% of

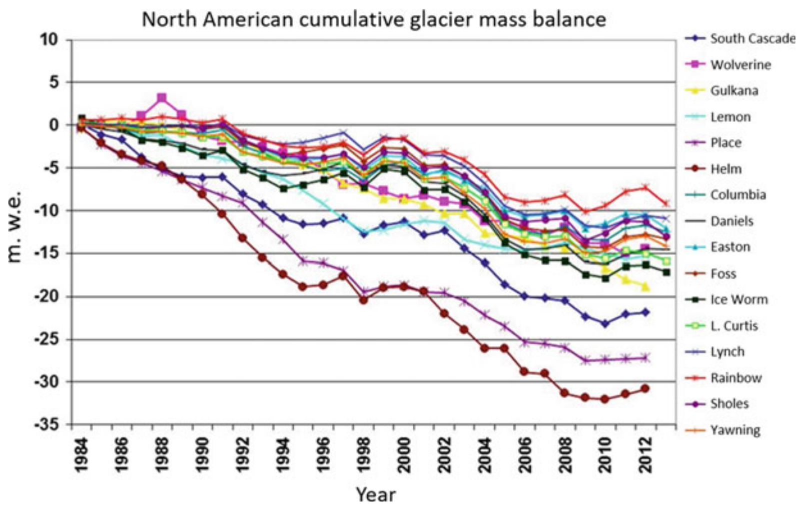
Fig. 1.18 Length and surface mass balance changes documented with in situ measurements for glaciers in Scandinavia and in the European Alps. (Modified from Beniston et al. 2018). Lower panels: Projected glacier mass evolution for Scandinavia and Central Europe between 2015 and 2100 relative to each region's glacier mass in 2015 (100%) based on three RCP emission scenarios (modified from Hock et al. 2019)



Alaska's glaciers are retreating (Thoman and Walsh 2019), as in other regions of North America (Fig. 1.20). Some mountain ranges and individual large glaciers are particularly affected, vividly illustrated by the Chugach Mountains on the south coast of Alaska, where a significant decrease in glaciation was observed. Here, the Columbia Glacier, one of the shrinking tidewater glaciers, is currently in a dramatic retreat. With a loss of about half of its volume since 1957 and 20 km of its length in the past three decades (McNabb and Hock 2014; Carlson et al. 2017a, b), the Columbia Glacier is one of the fastest changing glaciers in the world. Mass losses of

Fig. 1.19 Retreat of Rhone glacier, Switzerland, from the end of the Little Ice Age (1856) to 2005. (Obtained/modified from DFB AG; Copland 2011)

Fig. 1.20 North American cumulative glacier mass balance 1984–2013. (Modified from www.antarcticglaciers.org after M. Pelto)



Alaskan glaciers have been immense (Fig. 1.21). Estimates are in the order of 75±11 Gt per year between 1994 and 2013 (Larsen et al. 2015). Projections suggest continued and substantive glacier retreat and negative mass balances in the coming decades, with volume losses between 32 and 58% by 2100, making Alaskan glaciers large contributors to sea-level rise (Huss and Hock 2015) (cf. Fig. 1.11). The regional equilibrium line altitude is also projected to shift upward by 105 to 225 m, associated with a considerable decrease in snow precipitation (despite an increase in total precipitation), a shift to rain-dominated watersheds at lower elevations, shorter snow seasons, and warming permafrost (McGrath et al. 2017; Littell et al. 2018; Thoman and Walsh 2019).

The effects of widespread warming on the cryosphere such as shorter snow cover duration, earlier spring peak streamflow, thinning glaciers, and thawing permafrost are also evident in the mountain ranges of western Canada and the conterminous United States. These effects are projected to intensify in the coming decades. In the Rocky Mountains of Canada, a spatially coherent pattern of decreasing snow depth and snow cover duration and extent was detected for the period 1950–2013 (Fig. 1.22), with an average decline of the annual snow cover duration of about 4 days per decade, almost entirely due to reductions occurring during the spring season (DeBeer et al. 2016). Mountain glaciers in western Canada are receding at all latitudes, with rates of loss accelerating in the last few decades. While glaciers have exhibited a wide range of local changes from small net advances to complete disappearance, a decline in glacier cover of c. 25% over the past decades was observed in

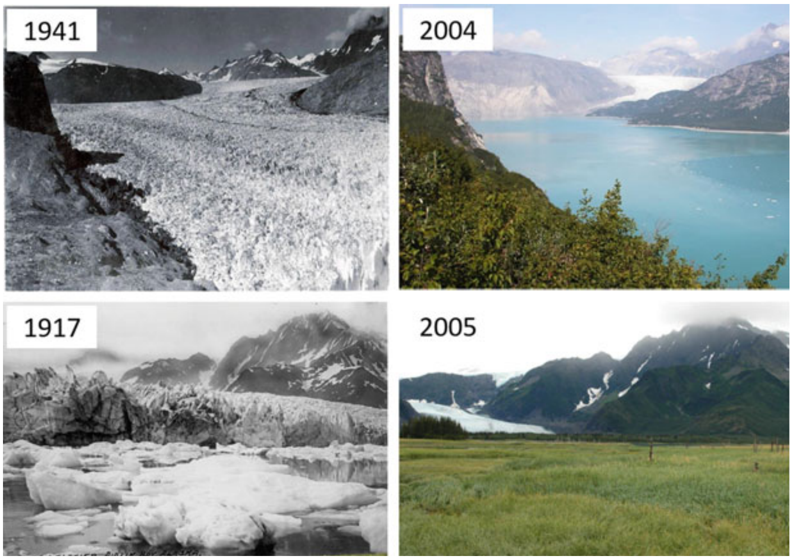


Fig. 1.21 Top: Retreat of Muir Glacier, Alaska, 1941–2004 (left photo by William O. Field; right photo by Bruce F. Molnia, USGS). Bottom: Retreat of Pedersen Glacier, Alaska, 1917–2005 (left photo by Louis H. Pedersen; right photo by Bruce F. Molnia, USGS); photos obtained from the Glacier Photograph Collection, Boulder, Colorado USA, National Snow and Ice Data Center/World Data Center for Glaciology

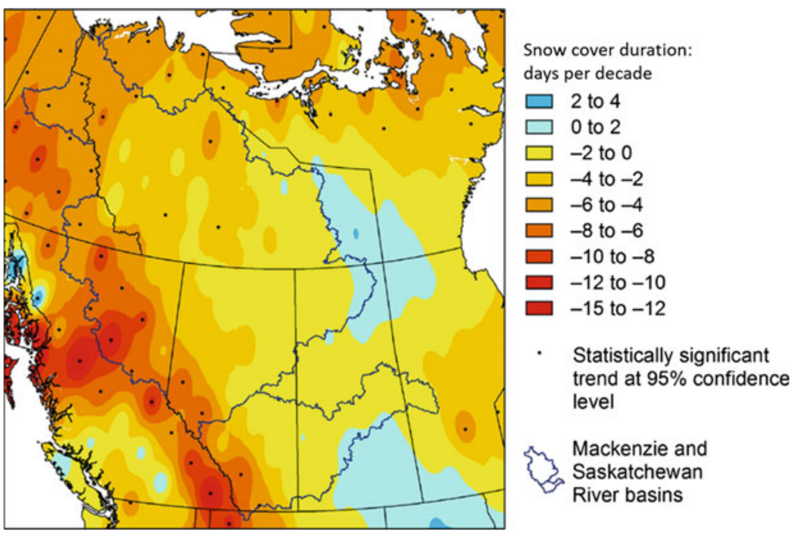


Fig. 1.22 Trends in spring season (February–August) snow cover duration for the period 1972–2013; spatial patterns indicate an above-average decline in the Rocky Mountains. (Modified from DeBeer et al. 2016)

most studies (Bolch et al. 2010a; Tennant et al. 2012; Beedle et al. 2015). A current hotspot of glacier shrinkage is located in the southern Coast Mountains in British Columbia, where the rate of mass loss over the period 2009–2018 was −7.4 ±1.9 Gt per year, about 20% higher than over the period 1985–1999 (Menounos et al. 2019). Projections for 2100 show drastic decreases of glacier area and volume in western Canada, with the volume of glacier ice shrinking by 70±10% relative to 2005, triggering severe hydrological implications and related impacts on aquatic ecosystems, agriculture, forestry, alpine tourism and water quality (Clarke et al. 2015).

The trend towards glacier recession, reduced mountain snowpack and earlier spring snowmelt runoff peaks is also widespread in the western United States, where glaciers cover an area of only 533 km^2, which is only 4% of the glacier area in western Canada (Menounos et al. 2019). Recent estimates suggest a decrease of the glacier and perennial snowfield area by 39% since the mid-twentieth century (Fountain et al. 2017). In Glacier National Park (Montana), only 35% of the Little Ice Age glaciers persisted by 2005 (Martin-Mikle and Fagre 2019). Glaciers in the Pacific Northwest, the most glacierized region in the conterminous United States, have displayed ubiquitous patterns of retreat and long-term negative trends in glacier area, resulting most likely in an immense reduction in late summer discharge volumes (up to 80%) by the end of the century due to post-peak declines in glacier melt and seasonal snowmelt (Frans et al. 2018). Observed declines in snowpack are dramatic, with over 90% of snow monitoring sites with long records across the western United States showing declines, regardless of phase changes in the Pacific Decadal Oscillation (PDO). Snowpack has declined on average by 21% or 36 km^3 since 1915, greater than the volume of water stored in the West's largest reservoir, Lake Mead (Mote et al. 2018). Decreases in snow water equivalent are generally larger at lower elevations. In the Cascade Mountains, area-averaged snowpack decreased by c. 20% since the 1950s, spring snowmelt occurred up to 30 days earlier, the share of late winter/early spring streamflow in annual flow increased by up to 20% or more, while the summer flow fraction decreased by up to 15% (Mote et al. 2014). Further shifts to earlier snowmelts and to substantially lower summer flows are projected (Elsner et al. 2010). Data from the Colorado Front Range also indicate ongoing degradation of mountain permafrost (Leopold et al. 2014).

While southern Sierra Nevada stations at higher elevations showed an upward trend in snow water equivalent over the twentieth century, with increased precipitation more than compensating for the overall warming, massive declines in peak snow water equivalent are projected for the Sierra Nevada and the southern Rocky Mountains for the coming decades (Garfin et al. 2014). Snowpack lows are particularly evident at lower Sierra Nevada elevations. 2015 saw a record low snowpack in the Sierra Nevada (Margulis et al. 2016). The estimated return interval for the 2015 1 April snow water equivalent value was calculated to be 3,100 years, highlighting its exceptional character (Belmecheri et al. 2016). As many watersheds in the Southwest of the United States depend on snowpack to provide the majority of the annual runoff, lower snow water equivalents imply reduced reservoir water storage. Reductions in runoff, streamflow, and soil moisture pose increased risks to the water supplies needed to maintain the Southwest's cities, agriculture, and ecosystems (Garfin et al. 2014). The glaciers of the Sierra Nevada show recently accelerated retreat rates, the absolute ice loss, however, is rather low due to the small glacier mass. Glacier areas have declined by more than half over the past century, and most glaciers will disappear completely from 2070 onwards if the current rate of loss continues (Basagic and Fountain 2011). An even faster disappearance is expected for the small, rapidly receding glaciers on the Mexican volcanoes which showed an overall glacier area loss of 75% between 1973 and 2017, implying water shortages in the surrounding areas. Ice-covered areas are only left on Volcán Citlaltepétl and Volcán Iztaccíhuatl, whereas Volcán Popocatepétl has lost its glaciers due to eruptive activity, even though the glacier shrinkage has started long before the appearance of eruptive products (Veettil and Wang 2018b; Cortés-Ramos et al. 2019).

A new map of snow cover changes in global mountain regions shows the Andes, in particular the southern Andes, as one of the hotspots of negative trends in snow parameters, with the area between Chile and Argentina (latitudes 29 to 42° S) exhibiting an overall snow cover duration decrease between 2500 and 4000 m of −26.6 days, and an earlier last snow day of −21.1 days over the period 2000–2018 (Fig. 1.23) (Notarnicola 2020). Saavedra et al. (2018) observed even more negative snow cover changes, with more pronounced snow loss on the east side of the Andes, and a significant increase in snowline

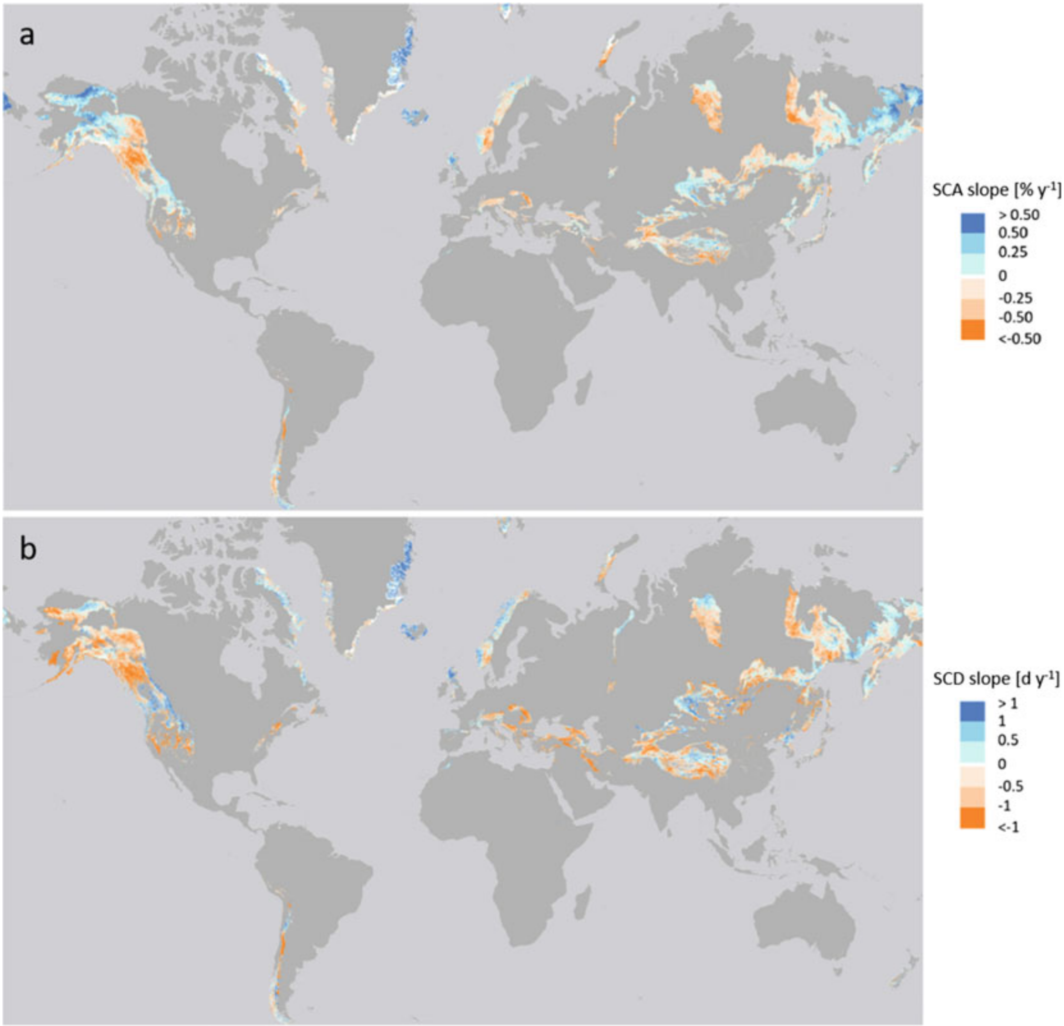


Fig. 1.23 Snow cover changes in global mountain regions shown as spatial distribution of positive and negative Sen's slopes resulting from MODIS products in the period 2000–2018; **a**: Snow-covered area (SCA slopes); **b**: Snow cover duration (SCD slopes). (Modified from Notarnicola 2020)

elevation south of 29–30°. Malmros et al. (2018) obtained similar results indicating adverse impacts on downstream water resource availability to agricultural, densely populated regions in central Chile and Argentina. The tropical Andes exhibit more heterogeneous snow cover trends. Mernild et al. (2017) simulated nonetheless a decrease in the number of snow cover days and in snow cover extent for the period 1979–2014.

Glaciers along the Andes exhibited a large-scale retreat over the past several decades; they are considered to be among the fastest shrinking glaciers on Earth. The recent dramatic recession of Andean glaciers is unprecedented since the maximum glacier extension of the Little Ice Age. Total Andean glacier mass change over the period 2000–2018 is estimated to be -22.9 ± 5.9 Gt yr^{-1}, thus comparable to the glacier mass change in entire High Mountain Asia (see above). The most negative mass balances over this period were assessed in the Patagonian Andes (-0.78 ± 0.25 m w.e. yr^{-1}), followed by the Tropical Andes (-0.42 ± 0.24 m w.e. yr^{-1}), while the Dry Andes showed relatively moderate losses (-0.28 ± 0.18 m w.

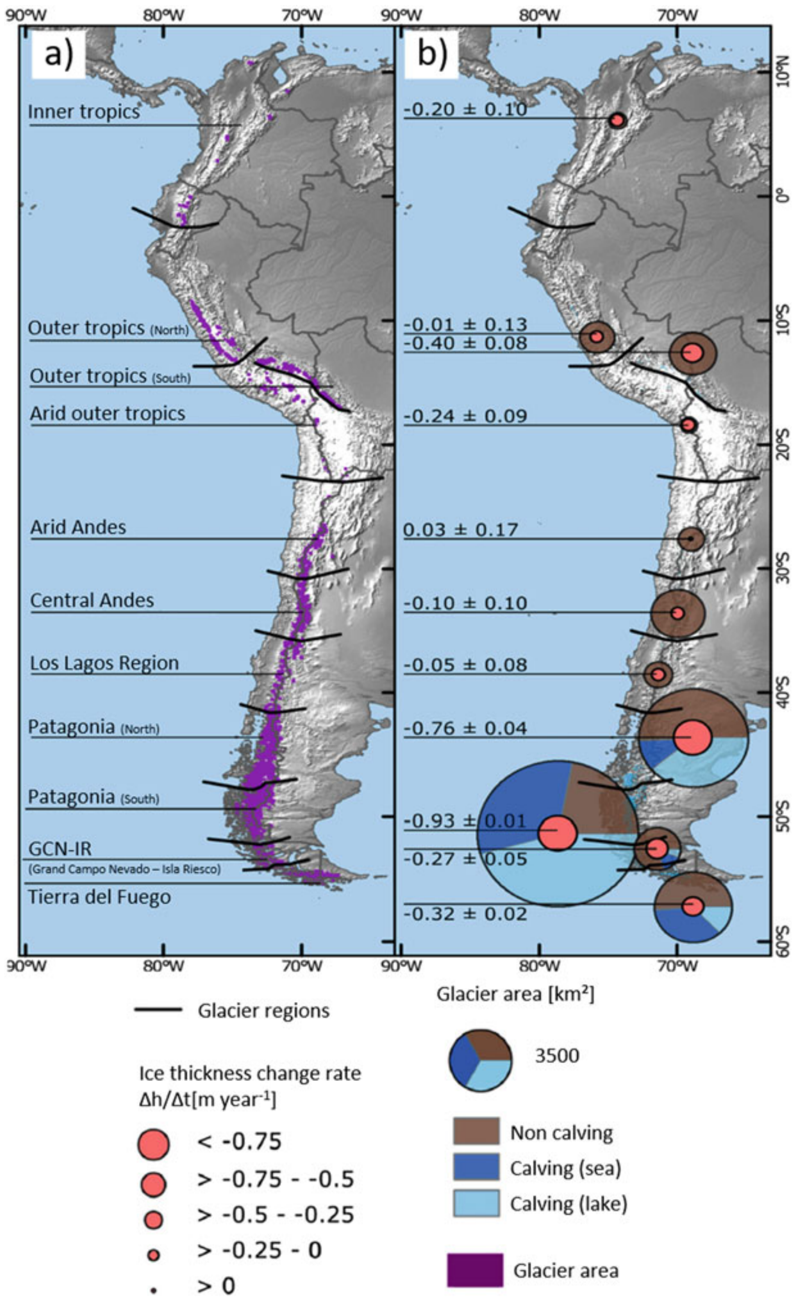


Fig. 1.24 Glacier regions in South America and ice thickness change rates 2000–2011/15; **a**: glacierized areas in purple, with black lines delimiting glacier regions; **b**: inner circles symbolize average ice thickness change rate of glacier regions; outer circles show glacierized area and proportion of respective glacier types. (Modified from Seehaus 2020 after Braun et al. 2019)

e. yr^{-1}) (Fig. 1.24) (Dusaillant et al. 2019). Braun et al. (2019) detected lower values for Andean glaciers and highlighted the massive ice loss of Patagonian icefields. Across the Patagonian Andes, the glacierized area was reduced by c. 20% within the last ~150 years (Meier et al. 2018). Dramatic examples of glacier recession include the Jorge Montt Glacier and the O'Higgins Glacier, the fastest shrinking glaciers in Chile, which lost 20 km and 15 km, respectively, of its length over the twentieth century (Schoolmeester et al. 2018). Accelerated mass loss is recently reported for glaciers of the dry Chilean Andes (Kinnard et al. 2020).

Mass loss from glaciers across the Andes of Colombia, Ecuador, Peru and Bolivia has been substantial, not seldom dramatic in recent decades, with a rather homogeneous pattern of glacier shrinkage and an accelerated retreat rate after 1976, followed by further increases after 2000 and 2013 (Rabatel et al. 2013; Mernild et al. 2015; Seehaus et al. 2019, 2020). Since the 1950s, glacier surface area has decreased to almost zero in Venezuela, which is about to become an ice-free country (Braun and Bezada 2013). In Colombia, the current glacier extent is 36% less than in the mid-1990s, 62% less than in the mid-twentieth century, and almost 90% less than the Little Ice Age maximum extent, and it is predicted that only the largest glaciers on the highest peaks will persist until the second half of this century (Rabatel et al. 2013, 2018). At the Chimborazo volcano in Ecuador, the loss of surface area was 72% between 1962 and 2016 (Schoolmeester et al. 2018). Many glaciers of the tropical Andes show comparatively sensitive and rapid responses to climatic changes, including an enhanced recession during El Niño events. Small glaciers at lower elevations (<5000 m a.s.l.) that do not have a permanent accumulation zone have already completely disappeared or will disappear within the next years/decades (Rabatel et al. 2013; Seehaus et al. 2019). In the Cordillera Blanca in Peru, the world 's most extensively glacier-covered tropical mountain range, glaciers have been rapidly receding as well over the past few decades (Schoolmeester et al. 2018). Projected warming will also result in the loss of permafrost. It is predicted that permafrost areas in the Bolivian Andes will shrink from present day extent by up to 95% under warming projected for the 2050s and by 99% for the 2080s and that almost all of the currently active Bolivian rock glaciers will be lost by the end of the century (Rangecroft et al. 2016).

Projections for the end of the century indicate that the future rise of the equilibrium line altitude may lead to further disappearance of glaciers at inner tropical sites under high emission scenarios, whereas outer tropical glaciers which are more strongly affected by future changes in the hydrologic cycle may persist as smaller glaciers (Vuille et al. 2018). The high ice loss rates of Andean glaciers result in a temporary increase in dry season water supply downstream. Meltwater supplies play a significant role in wetland cover dynamics in the high Andes (Dangles et al. 2017). Peak water, however, has already passed in many glacierized catchments, and, in the long term, dry season river discharge will decrease due to future glacier shrinkage, contributing to emerging water resource crises and environmental hazards for both urban and rural populations relying on glacier-fed streams for agriculture and livelihoods (Thompson et al. 2017; Vuille et al. 2018).

Africa

Snowpack is on the decline in North Africa and thus in accordance with trends in other Mediterranean regions, notwithstanding the fact that the persistence of snow cover is highly variable in space and time (Fayad et al. 2017). In the Atlas Mountains, a statistically significant long-term trend has not been detected yet (Marchane et al. 2015). However, a combination of warming and reduced precipitation, associated with earlier springtime melting, will result in reduced snowpack, adversely affecting the supplies of meltwater for lowland areas in Morocco (García-Ruiz et al. 2011; Marchane et al. 2017). The Drakensberg Range in the Lesotho Highlands is characterized by a very high inter- and intra-annual variability of snow coverage (Wunderle et al. 2016), and at the same time by a decadal trend of declining snow depth and snow cover duration, with much lower values in comparison to the late nineteenth century (Grab et al. 2017).

Climate change impacts on the cryosphere are most obvious in East Africa where the only African mountains are located which have glaciations in their summit regions (Kilimanjaro [5895 m], Mount Kenya [5199 m] and Rwenzori [5109 m]). All these glaciers show an extraordinary recession over the past century, with a loss of more than 80% of the glacier area on all three mountains (Fig. 1.25). Analysed mass and energy fluxes on selected glaciers on Mount Kenya and Kilimanjaro suggest that the

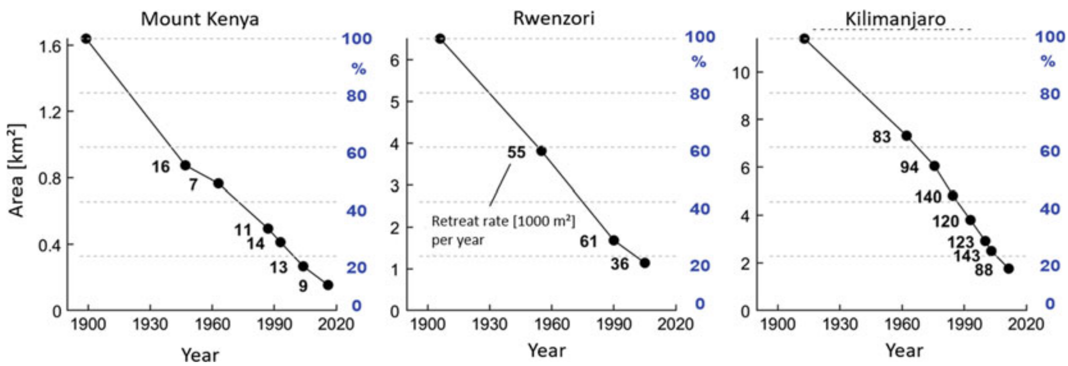


Fig. 1.25 Loss of glacier area on Mt. Kenya, the Rwenzori Mountains, and Mt. Kilimanjaro; absolute values on left y-axes, percentage change, relative to the first available area value on right y-axes; numbers in bold show mean area loss between points in time (in ‘000 m^2 a^{-1}). (Modified from Prinz and Mölg 2020)

frequency and amount of solid precipitation is the dominant local climatic factor for the mass balance of these glaciers, with decreasing snowfall being interpreted to be a concomitant effect of global warming (Mölg et al. 2009; Prinz et al. 2016). Ice loss is particularly severe on the Lewis Glacier, the largest glacier on Mount Kenya, which has already lost 90% of its area and 95% of its volume since the end of the nineteenth century (Prinz et al. 2018; Chen et al. 2018). 8 glaciers vanished completely until 2004 (Hastenrath 2005). Since 2010, the mass loss of Lewis Glacier has been accelerating due to glacier disintegration, yet another glacier disappeared completely, and Prinz et al. (2018) predict that Mt. Kenya’s glaciers will be extinct before 2030, if current retreat rates continue. The loss of ice cover on Kilimanjaro is similarly dramatic (Fig. 1.26), with glaciers having retreated from their former extent of 11.40 km^2 in 1912 to 1.76 km^2 in 2011 (Cullen et al. 2013). About the same magnitude of glacier recession was reported for the Rwenzori Mountains, where only the higher elevated glaciers on Mt. Stanley have shown a slower decrease (Kaser and Osmaston 2002; Mölg et al. 2006). Nevertheless, the Stanley glacier had almost vanished by 2008 (Mumba 2008; Spinage 2012). The scenario of a complete disappearance of all ice at Kilimanjaro and Rwenzori is likely to occur between 2040 and 2060 (Mölg et al. 2003; Cullen et al. 2013). The East African glaciers do not play a major role in the regional water balance, however, they are of great importance for the tourism potential in the respective regions.

1.2.3 Biotic Responses

1.2.3.1 General Overview

High mountain ecosystems and their biodiversity are affected by climate change at an accelerated pace. It is evident from long-term ecological monitoring and large-scale assessments that the high levels of warming to which mountain ecosystems are exposed have resulted in substantial redistributions and losses of habitats and species, and in increased vulnerability to additional stressors such as invasive species or disturbances (Jentsch and Beierkuhnlein 2003; Pauchard et al. 2009; Pauli et al. 2012; Wipf et al. 2013; Alexander et al. 2016; Dainese et al. 2017; Lamprecht et al. 2018; Steinbauer et al. 2018; Pauli and Halloy 2019; Petriccione and Bricca 2019). Climate change effects on temperature, snow, moisture, and nutrient regimes potentially cause alterations in plant physiology and phenology, species interactions, community structure, species distributions, and ecosystem processes (Körner 2003; Winkler et al. 2019), with higher losses of biodiversity and habitats occurring with higher levels of climate warming (Nunez et al. 2019). Respective changes are increasingly observed, the knowledge of the

Fig. 1.26 Retreat of glaciers on Mount Kilimanjaro 1912–2006; 85% of the ice has disappeared during this period. (Modified from Thompson 2010)

alteration of mountain ecosystems, however, is still profoundly deficient, in particular in many of the underresearched mountain ranges in the Global South (Schickhoff and Mal 2020). Species responses to climate change are driven by the capacity to persist in situ by altering fitness-related traits through plastic adjustment or genetic adaptation to novel stresses such as longer growing seasons, increasing temperatures, and less infertile soils. Plants at higher elevations have a low capacity to persist in situ since traits such as slow growth or dwarfism are genotypically determined, and the phenotypic plasticity is constrained under harsh climatic conditions. A greater potential for montane and alpine species to adapt and to survive rapid anthropogenic climate change lies in distributional shifts to track preferred bioclimatic conditions (Schickhoff 2011, 2016a; Pauli and Halloy 2019; Winkler et al. 2019; Winkler 2020). However, clonal, relatively slow dispersal strategies are not uncommon at high elevations which restrict the potential of shifting range limits. Magnitude and rate of climate change as well as induced alterations of abiotic and biotic site conditions will overstretch the adaptive capacity of many species, increased extinction risks and losses of biodiversity are thus inevitable (Thuiller et al. 2005; Moritz and Agudo 2013). To date, only a small percentage of countries is on track to achieve respective national biodiversity targets within the framework of the Convention on Biological Diversity. A fundamental embedding of mountain biodiversity in national biodiversity conservation strategies is necessary in order to better meet the objectives of the UN Sustainable Development Goals (UN 2020).

Changes in species distribution ranges as a complex response to novel constellations of bioclimatic and other site conditions are increasingly observed in mountain regions, with range extensions to higher elevations being considerably overrepresented, compared to range contractions. Species from lower elevations are now colonizing habitats on mountain summits at a rate which is five times faster than half a century ago (Fig. 1.27) (Steinbauer et al. 2018; Pauli and Halloy 2019). Implications of the establishment of these 'neonative' species (Essl et al. 2019) include the gradual transformation of species composition and community structure of resident communities, in particular in the alpine and nival zones. Warmth-demanding and/or less cold-adapted species become more dominant, while strongly cold-adapted high-elevation species are declining in abundance and frequency. Severe area losses of these cryophilic species are expected since upward range shifts are constrained by limited available space (Engler et al. 2011; Elsen and Tingley 2015; Lenoir and Svenning 2015; Freeman et al. 2018). Species-specific migration rates are very different, suggesting that the interaction of multiple internal species-specific traits controls the response to changed climatic conditions (cf. Roux and McGeoch 2008). Asynchronous responses to external driving forces result in 'no-analogue communities' with modified competitive conditions (Williams and Jackson 2007). Novel biocoenoses with modified dominance relationships, competitive conditions and population densities inevitably affect ecosystem functioning, and thus the provision of ecological services and the resilience to disturbances (Pecl et al. 2017).

Treeline ecotones are as well subjected to reinforced dynamics in recent decades, though not yet necessarily reflected in treeline advance. Since the elevational position of natural alpine treelines is primarily caused by heat deficiency (Holtmeier 2009; Körner 2012, 2020), and treeline elevations have responded to climate oscillations throughout the Holocene (Tinner and Theurillat 2003), treelines are often considered to be sensitive indicators of global warming. The findings of observational studies on treeline shifts, however, give evidence of both advancing treelines and insignificant treeline responses (Holtmeier and Broll 2005, 2007, 2017a; Schickhoff et al. 2015). A global meta-analysis of treeline response to climate warming showed advancing treelines at 52% and persistent treelines at 47% of the studied sites (Harsch et al. 2009). A recent meta-analysis across the Northern Hemisphere found almost 90% of treelines ascending over the past century and c. 10% remaining stable while the mean hemispheric shift rate was much lower than expected from climate warming (Lu et al. 2020). Inconsistent responses indicate a highly heterogeneous sensitivity of alpine treelines to the effects of climate warming which is not surprising given the multitude of after-effects of treeline-landscape

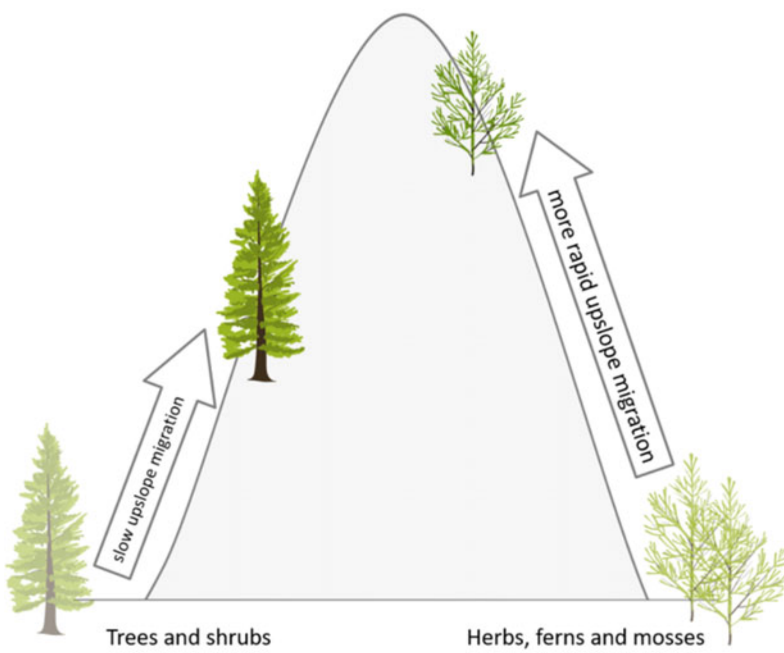


Fig. 1.27 Short-lived plant species exhibit a more rapid upslope migration as compared to long-lived plant species; upward range shifts may be constrained by limited available space. (Modified from Lenoir and Svenning 2013)

history (past climate fluctuations, natural and anthropogenic disturbances) that determine treeline position, spatial patterns and successional stages (Holtmeier and Broll 2017a). For example, human impacts are almost omnipresent at treeline environments in Africa, Asia, and Europe where mountain regions are settled since ancient times, and effects of land use history and dynamics overlap with those of multiple ecological and biophysical factors. Thus, a potential advance of a particular treeline (at local scale) to higher elevations is very difficult to predict. A global comparison of the response variability of different treeline forms revealed a certain correlation between spatial patterns and response dynamics of treeline ecotones, with the majority of diffuse treelines showing an advance (Harsch and Bader 2011; Bader et al. 2020). However, other factors and interrelationships, for instance species-specific traits and response patterns of treeline-forming tree species, may superimpose the response trend of treeline forms (Treml and Veblen 2017). The interactions between climatic changes as regional to global input variables and facilitating, modulating or overriding site factors at the local scale (the complex of abiotic and biotic local site conditions and their interactions and feedback systems including human impact and the entire treeline-landscape history) control current spatial patterns and temporal dynamics in treeline ecotones (Wieser et al. 2014; Elliott 2017; Holtmeier and Broll 2017a; Schickhoff et al. 2020). Lagged changes in treeline positions should not obscure the fact that current warming trends are favourable to growth, development, and reproduction of tree species in many treeline environments. Assuming that alpine treelines would have tracked global warming some day and reached a new steady state at higher elevations, the shrinking of lower and upper alpine/nival life zones would be dramatic. An upslope extension of mountain forests corresponding to a 2.2 K warming is likely to lead to a global loss of c. 24% of the lower alpine zone and of c. 55% of the upper alpine and nival zones (Körner 2012). Large-scale treeline shifts would have serious implications for diversity and function of high elevation ecosystems (Greenwood and Jump 2014).

Mountain endemics and other species with spatially restricted populations will be particularly affected by large magnitudes of climate change, fragmenting populations and reducing vigour and viability of species. In addition, endemic species are particularly vulnerable to genetic swamping due to introgressive hybridization (Gómez et al. 2015). Species in regions with declining precipitation are exposed to a higher risk as well (Engler et al. 2011). Global meta-analyses give impressive evidence of climate change-induced species migrations and range extensions (cf. Tomiolo and Ward 2018). Low dispersal abilities and slow migration rates, however, will prevent many species from keeping pace with the relocation of climatically suitable habitats (Settele et al. 2014). Many alpine plants depend on a certain required day length to become phenologically active (photoperiodism) (Keller and Körner 2003). Many other alpine plants, however, adjust sequences of phenological events to the rise in temperatures and to the advance in the timing of snowmelt. Thus, phenological shifts are considered to be sensitive indications of the response to climate warming. A phenological fingerprint of climate warming has been detected on a global scale, most pronounced at higher latitudes and higher elevations (Peñuelas et al. 2013; Piao et al. 2019).

The response of plants to climate change with regard to fitness and primary productivity has significant feedbacks to the global climate since the terrestrial biosphere plays a key role in the global carbon cycle, i.e. changes in primary production imply changes in the carbon storage of ecosystems. In recent decades, the global net primary production has slightly increased (Settele et al. 2014). Significant biotic responses also include alterations of the dense network of functional relationships, interdependencies and mutual interferences between species. Ecological or biotic interactions ensure a certain degree of self-regulation and resilience of ecosystems. Examples of spatial and temporal decoupling of interacting species, e.g. between herbivores and their food plants, and other mismatches in interactions with potentially adverse effects on ecosystem services are increasingly documented (Valiente-Banuet et al. 2015). Shifts in species composition and community structure are to an increasing extent caused by biological invasions into mountain regions (Pauchard et al. 2009, 2016; Alexander et al. 2016). Invasive species are predominantly climate change winners since they are often thermophilic and very adaptive due to wide ecological amplitudes. The expansion of non-native species further contributes to a reorganization of higher-elevation communities and alters ecological interactions and the provision of ecosystem services. It also contributes to homogenization effects of mountain ecosystems and biota (Jurasinski and Kreyling 2007; Malanson et al. 2011).

The focus in the following regional overview of biotic responses is on terrestrial vascular plants as the main primary producers in high mountain ecosystems and on vegetation as major structural component. Upslope extension of distribution ranges is also evident for numerous animal species and has been documented for many taxonomic groups (Gonzalez et al. 2010; Chen et al. 2011).

1.2.3.2 Regional Overview

Asia and Australasia

Representing four of the 34 global biodiversity hotspots and numerous ecoregions with significant conservation value, the HKH represents a major centre of global biodiversity (Myers et al. 2000; Pandit et al. 2014; Bhattacharjee et al. 2017; Xu et al. 2019). Impacts of climate change such as increasing temperature variability and declining precipitation during the dry season will affect the majority of species, thus threatening biodiversity conservation and the maintenance of mountain ecosystem integrity. A recent assessment based on satellite-derived NDVI datasets indicates that the length of the growing season in the HKH has increased by 4.25 days per decade over the last five decades (Krishnan et al. 2019a), in line with an overall greening trend in NDVI magnitude and an earlier green-up in most parts of the HKH region (Panday and Ghimire 2012; Mishra and Mainali 2017; Baniya et al. 2018). In particular, subnival vegetation above 5000 m has expanded (Fig. 1.28) (Anderson et al. 2020), and will have more space available for future expansion (Keenan and Riley 2018). Respective observations of shifts in species-specific phenological patterns are still limited, but indicate large-scale changes. Several *Rhododendron* and other species were found to currently flower several weeks earlier than in previous decades (Xu et al. 2009; Gaira et al. 2011; Mohandass et al. 2015; Negi and Rawal 2019; see also Yang et al. 2017 and Dorji et al. 2020 for the Tibetan Plateau). Adhikari et al. (2018) reported significant changes in phenological patterns in a treeline ecotone in Uttarakhand over recent decades, with the majority of species showing advanced flowering and extended vegetative phases. Mean date of leafing and flowering in lower elevation forests (Sal, Pine, and Oak forests) in the same region has advanced by 1–2 weeks within a period of 30 years (1985–2015) (Singh and Negi 2016). However, no significant changes over the past century were found for flowering phenology of *Rosaceae* species in the Hengduan Mountains, indicating that phenological responses to climate change are more complex than commonly assumed (Yu et al. 2016).

In the vast HKH region, climate change-induced shifts in species distributions and species composition of communities have been occurring, largely without being noticed or documented yet by science. In a first detailed study on upslope migration, Telwala et al. (2013) provided evidence of warming-driven elevational range shifts in 87% of 124 studied endemic plant species in alpine Sikkim over the last 150 years. Considering shifts of species' upper elevation limits of up to c. 1000 m, present-day plant assemblages and community structures are definitely different from those of the nineteenth century. In recent years, long-term research plots have been established within the Himalayan GLORIA (Global Observation Research Initiative in Alpine Environments) sub-network in order to monitor in detail species

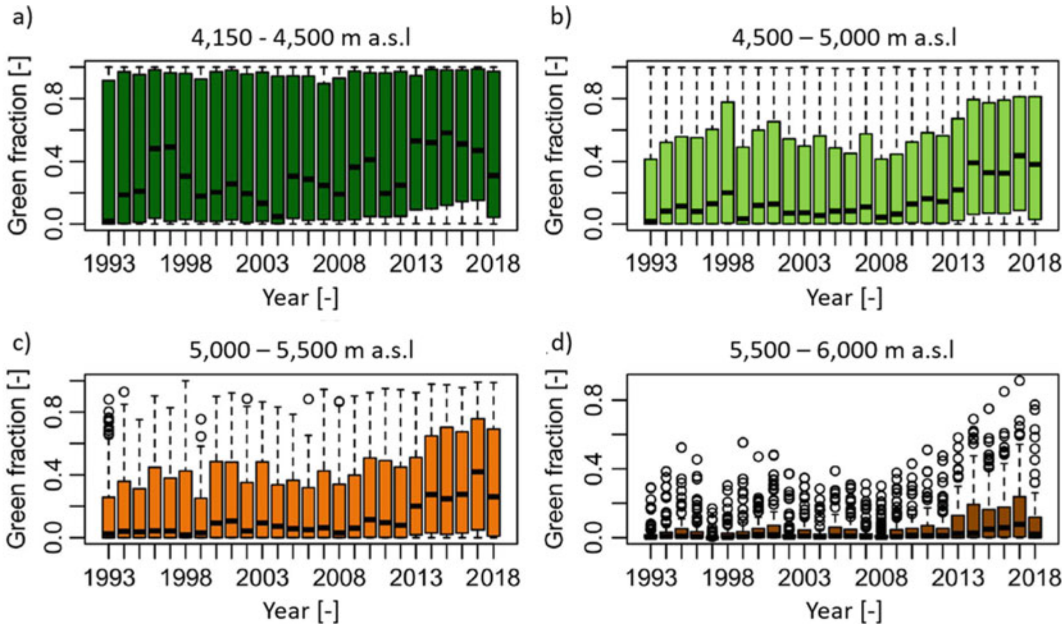


Fig. 1.28 Boxplots of green pixel fraction values for the broader HKH region (1993–2018), showing weakly positive time series trends in vegetated ground, separated by elevational zones; the extent of the boxes represent the 25th and 75th percentiles (quartiles), the bold middle line is the 50th percentile (median), the whiskers are the minimum and maximum values which fall within 1.5 times the interquartile range and the circles represent values beyond this range (outliers). (Modified from Anderson et al. 2020)

distribution patterns (Salick et al. 2014; Sekar et al. 2017). First resampling of plots in the eastern Himalaya (Hengduan Mountains) after seven years yielded the result that alpine plants on high-elevation summits increased in number of species, in frequency and in diversity, and that even Himalayan endemic species showed positive population trends (Salick et al. 2019, 2020). Here, modelling studies also project upslope expansion of distribution ranges (You et al. 2018). Likewise, the total number of vascular plant species on summits in the Kashmir Himalaya increased between 2014 and 2018 (Hamid et al. 2020). Warming-induced upward migration of plants was also observed in Ladakh by Dolezal et al. (2016), who resurveyed outpost populations of subnival plants after ten years and found an extension of the elevational range of 120–180 m. On the Tibetan Plateau, experimental studies highlighted the critical role of soil moisture for plant communities' response to climate warming: Alpine meadows showed increases in net primary productivity, while alpine steppes experienced decreasing productivity, decreasing cover of graminoids and forbs, and rapid species losses due to warming-induced drought conditions (Ganjurjav et al. 2016). The performance of the dominant species in central Tibetan Plateau alpine meadows, the shallow-rooted *Kobresia pygmaea*, in terms of plant cover, reproductive phenology/success and competitiveness was also found to be largely controlled by soil moisture which tends to decrease under climate warming (cf. Dorji et al. 2013, 2018). In line with these results, Lehnert et al. (2016) found that variability in precipitation and soil moisture outweighs overgrazing as the primary driver of recent large-scale vegetation changes on the Tibetan Plateau. Recently deglaciated terrain represents another highly dynamic alpine habitat. The surface area of deglaciated glacier forelands has been increasing considerably due to the ongoing recession of the vast majority of HKH glaciers. Vegetation successions on glacier forelands have not been addressed in greater detail so far. Some preliminary studies are available analysing the colonization of glacier forelands by pioneer species (Mong and Vetaas 2006; Vetaas 2007; Miehe 2015; Bisht et al. 2016).

Treeline dynamics and treeline shifts in the HKH region mostly result from combined effects of land use change and climate change (Schickhoff et al. 2015, 2016b; Shrestha et al. 2015; Suwal et al. 2016). HKH treelines are almost exclusively lowered from their natural elevational position by long-lasting human impacts (anthropogenic treelines). If these treelines are moving upslope, recent land abandonment or declining human impacts are the dominant drivers whereas climate change plays a subordinate role. Substantial treeline advances or shrub encroachments of alpine meadows in recent decades, reported in some studies (e.g. Baker and Moseley 2007; Brandt et al. 2013; Singh et al. 2018), have to be mainly attributed to effects of land use change. Bold statements in remote sensing studies about exceptional short-term climate warming-induced treeline advances (e.g. Mohapatra et al. 2019; Singh et al. 2020) must be viewed with extreme caution, in particular if they are not backed up by field data. Climate change, however, is a more significant driver of near-natural treeline dynamics. Only very few near-natural treeline ecotones have persisted in remote locations, mainly on north-facing slopes where these treelines have to be categorized as krummholz treelines (Schickhoff et al. 2015, 2016b; Schwab et al. 2017; 2020); they show rather low responsiveness to climate warming (see also Liang et al. 2011; Chhetri and Cairns 2015, 2018; Gaire et al. 2017; Sigdel et al. 2020).

Studies at the near-natural treeline in the Rolwaling valley (Nepal Himalaya) showed that the dense, self-sustaining and persistent krummholz belt of *Rhododendron campanulatum* forms a very effective barrier that largely prevents the expected upslope migration of *Abies spectabilis* and *Betula utilis* and other treeline tree species. The site conditions in the krummholz belt, modified by *Rh. campanulatum* itself in particular in terms of light and nutrient deficiencies, lower soil temperatures, and allelopathic effects, severely restrict the competitiveness of other tree species, reflected inter alia in a negative correlation between abundance and density of *Rh. campanulatum* and recruitment of other tree species. The elevational position of the Rolwaling treeline can be regarded as rather stable, suggesting a certain decoupling of treeline dynamics from global warming. However, the sensitivity is clearly evident in terms of stand density, seed-based regeneration and tree growth patterns, while a treeline shift is to be expected in the medium to long term only (decades to centuries) (Schickhoff et al. 2016b, 2020; Schwab et al. 2016, 2017, 2020; Müller et al. 2016; Bürzle et al. 2018).

Increasing stand densification as well as intense tree recruitment within other Himalayan treeline ecotones indicate the potential for future treeline shifts (Lv and Zhang 2012; Gaire et al. 2014, 2017; Shrestha et al. 2015; Wang et al. 2016; Tiwari et al. 2017a; Yadava et al. 2017; Tiwari and Jha 2018; Mainali et al. 2020; Sharma et al. 2020). Moreover, remote sensing studies indicate a general biomass increase in treeline ecotones over recent decades (Rai et al. 2013, 2019), while modelling studies support the concept of climate change-induced upslope range expansion of treeline species (Forrest et al. 2012; Joshi et al. 2012; Zomer et al. 2014; Rashid et al. 2015; Manish et al. 2016; Bobrowski et al. 2017; Lamsal et al. 2017; Chhetri et al. 2018; Gilani et al. 2020). Recent dendroecological studies at HKH treelines indicate enhanced tree growth at some high-elevation sites (Fan et al. 2009; He et al. 2013; Huang et al. 2017; Thapa et al. 2017; Shi et al. 2020), and a widespread strong sensitivity of tree growth to pre-monsoon temperature and humidity conditions (Fig. 1.29) (Dawadi et al. 2013; Liang et al. 2014; Ram and Borgaonkar 2014; Bräuning et al. 2016; Kharal et al. 2017; Panthi et al. 2017; Sohar et al. 2017; Tiwari et al. 2017b; Schwab et al. 2018; Singh SP et al. 2019). Warming-induced higher evapotranspiration and soil moisture deficits during dry spring months adversely affect tree growth in particular on sites which are prone to drought stress. Moisture supply in the pre-monsoon

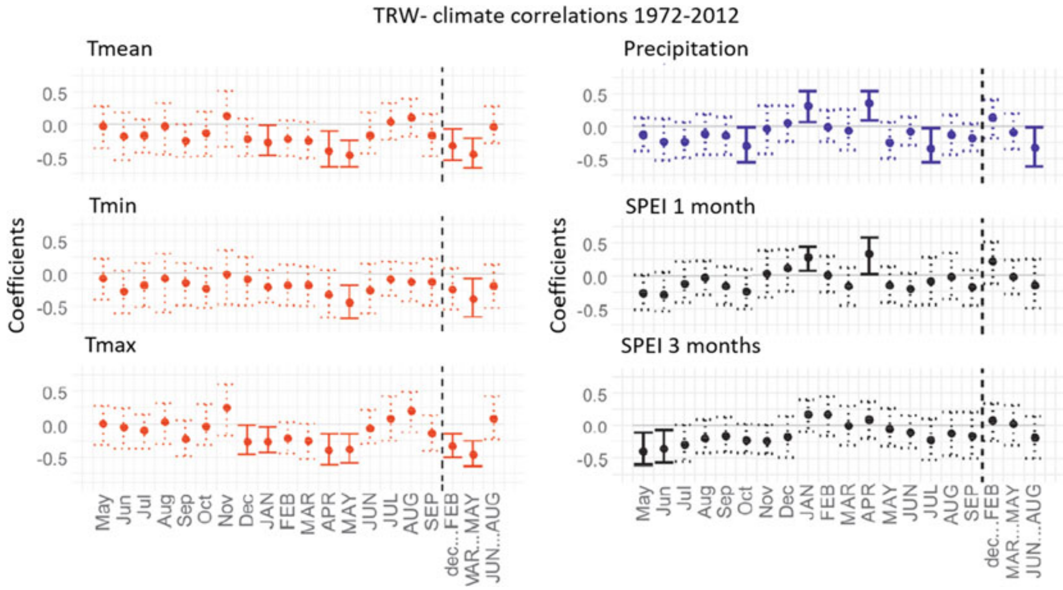


Fig. 1.29 Static correlations (1972–2012) of the tree-ring width chronology with temperature, precipitation and drought indices (SPEI) for current and previous year's months and current year seasons; solid bars indicate significant correlations ($p < 0.05$). (Modified from Schwab et al. 2020)

season might become an effective control of future treeline dynamics (Schickhoff et al. 2016b; Mishra and Mainali 2017; Sigdel et al. 2018; Lyu et al. 2019; Schwab et al. 2020). Correlations between ring width and winter temperatures in treeline ecotones were found to be largely positive (e.g. Bhattacharyya et al. 2006). However, increasing winter temperatures can be detrimental to the growth of *Rhododendron* shrub species above the treeline (Bi et al. 2017).

At elevations below the treeline, climate change is as well a major threat for the integrity of ecosystems (Chakraborty et al. 2018; Chettri et al. 2020). Montane and subalpine forest ecosystems in the HKH region are very critical for biodiversity, watershed protection and livelihoods of forest-dependent communities. Impacts of climate change on species distribution patterns, species composition of forest communities, and ecosystem functioning might degrade the capacity to maintain the provision of ecosystem services. Recently, a considerable upward migration of alien invasive species into the Himalaya was observed (Bajracharya et al. 2015; Negi and Rawal 2019), which is projected to continue (Lamsal et al. 2018). The spread of exotic species such as *Ageratina adenophora* (up to 2800 m) or *Lantana camara* (beyond 1500 m) over vast stretches of lower elevational zones alters species composition and ecosystem services of native plant communities. Higher native species richness obviously facilitates the invasibility of habitats (Bhattarai et al. 2014).

Similar trends of climate change-induced biotic responses, summarized above for the HKH alpine regions, prevail in other Asian and Australasian mountain systems. With regard to phenological changes, results of a meta-analysis across 145 sites in China demonstrated that more than 90% of the spring/summer phenophases time series show earlier trends and 69% of the autumn phenophases records show later trends (Ge et al. 2015). Recent positive trends in vegetation growth and productivity have been detected for the mountain regions of the Chinese landmass (Xu et al. 2014; Fang et al. 2016) and for some Mongolian mountain ranges (Kappas et al. 2020), including partially substantial treeline advances (Du et al. 2018) and upslope expanding distribution ranges (Zong et al. 2016).

Considerable shifts of upper altitudinal limits of mountain plant distributions were assessed in the Central Mountain Range of Taiwan over the last century, in parallel with rising temperatures in the region (Jump et al. 2012). Treeline advance on Mt. Fuji, Japan, is enhanced by climate warming (Sakio and Masuzawa 2012). Encroachment of subalpine bamboo species into alpine meadows, resulting in declining plant species richness, was reported from the Taisetsu Mountains, Hokkaido, northern Japan (Kudo et al. 2011). Warming-induced vegetation dynamics in the Altai and Mongolian mountains and in the Tien Shan and Pamir will be largely controlled by moisture conditions (Dulamsuren et al. 2010a, b; Poulter et al. 2013; Bao et al. 2015; Seim et al. 2016; Yin et al. 2016; Jiang et al. 2017; Dubovyk 2018), which also drives the extent to which forests will be transformed to forest-steppes and steppes in southern Siberian mountains in the next decades (Tchebakova et al. 2016). As temperatures in Inner Asia have increased substantially since the mid-twentieth century, tree growth has declined in many areas of the forest steppe (Dulamsuren et al. 2010b, 2011, 2013; Liang et al. 2016). At treeline elevations in the Chinese and Mongolian Altai, Chen et al. (2012) and Dulamsuren et al. (2014) assessed positive correlations of tree growth and growing season temperature, and no drought-induced growth limitation. Kirdyanov et al. (2012) assessed a densification of formerly open forests and an upslope shift of the treeline of approximately 50 m over the last century in the Putorana Mountains, northern Siberia, corroborating the large-scale treeline advances and tree growth enhancements found over much of Siberia by Esper and Schweingruber (2004), Soja et al. (2007), Kharuk et al. (2010), Petrov et al. (2015), Shevtsova et al. (2020) and others.

In the Russian Altai, a treeline shift of 150 m during the past 50 years was reported, with the rate of upslope movement having accelerated until recently (Gatti et al. 2019). In the North Urals, the upper limits of tree stands with different degrees of canopy closure have risen by about 100 m of elevation since the mid-nineteenth century (Moiseev et al. 2010; Hagedorn et al. 2014). Accelerated forest growth in the treeline ecotone has been detected in the Tien Shan under conditions of rising temperatures and sufficient precipitation (Qi et al. 2015). Elevational belt shifts are expected in the Tien Shan, while shifts of phenological dates have already been observed (Dimeyeva et al. 2015; Imanberdieva et al. 2018). A study of long-term vegetation dynamics of alpine communities in the Caucasus confirmed an upward shift of the upper limit of species distributions and an increasing abundance of species in upper alpine zones (Elumeeva et al. 2013; see also Gigauri et al. 2013). Treeline tree species in the Caucasus (*Betula litwinowii*) expand their range to higher elevations as well, caused by combined effects of land use change (reduced grazing pressure) and climate change (Akatov 2009; Hansen et al. 2018). Climate change-induced migrations will most likely result in northward and upward shifts of subalpine plant species in the mountains around the Iran Plateau (Shamsabad et al. 2018), treeline advances are expected in Pontic Mountains (Kurt et al. 2015).

Preliminary observations in alpine zones of SE Asian mountains point to phenological changes and to subalpine/alpine grasslands affected by shrub and tree encroachment (Hope 2014). Overall trends towards a longer duration of the growing period were detected in different study areas of the Australian and New Zealand Alps (e.g. Thompson and Paull 2017). The average advance of the timing of spring events, based on long-term datasets of c. 350 species, was calculated to be c. 4 days per decade (Chambers et al. 2013). Shifts in species' distributions are predicted for many taxa in the mountains of Australia and New Zealand, with suitable habitats shifting and/or contracting as the climate changes (Cabrelli et al. 2015). A temperature rise of 3 °C may lead to a loss of c. 80% of existing alpine lands in New Zealand and to a loss of up to 50% of vascular plant taxa (Halloy and Mark 2003). The New Zealand *Nothofagus* treelines are relatively unresponsive to recent climate warming, however, and show only little evidence of treeline advance (McGlone et al. 2010; Harsch et al. 2012). Population dynamics at the alpine treeline in SE Australia were found to be responsive to climate change, reflected in a recent short-distance treeline advance, while treeline dynamics is largely controlled by fire (Naccarella et al. 2020). It needs to be highlighted that there are still tremendous knowledge deficits with regard to climate change-induced biotic responses in mountain life zones of Asia and Australasia which need to be reduced in order to develop appropriate management strategies aiming at the maintenance of mountain ecosystem integrity and the continuous provision of essential goods and services.

Europe

Long-term greening trends prevail at higher elevations in representative regions of the European Alps. Significant increases of peak NDVI are widespread over recent decades (Julien et al. 2006; Carlson et al. 2017a, b; Filippa et al. 2019). Accelerated greening of above treeline habitats coincides with a pronounced increase in the amount of snow-free growing degree-days. Remote sensing studies confirm the observed recent colonization of previously glaciated/non-vegetated areas at higher elevations as well as shrub/tree encroachment due to the abandonment of agricultural practices, and highlight the interplay of climate and land use change in controlling greening dynamics in the Alps (Filippa et al. 2019). Reduced human activities also play a major role in recent biomass increases in Scandinavian mountains where regional case studies suggest that climate warming is of subordinate importance (Tømmervik et al. 2019). Growth responses to climate are complex and spatio-temporally unstable (Hofgaard et al. 2019), however, the largely increasing trend in radial growth of trees and in productivity of northern vegetation over recent decades is considered to be climate change-induced (Lopatin et al. 2008; Park et al. 2016). Positive trends in maximum NDVI detected in Arctic mountains are positively correlated with mean summer temperature (Vickers et al. 2016). However, high latitude greening is complex and browning drivers such as extreme winter warming and loss of freeze tolerance, drought stress, thermokarst development, fire, defoliating insects, or rust fungi may temporarily reduce greenness and productivity in different parts of mountain landscapes (Buermann et al. 2014; Phoenix and Bjerke 2016; Tei et al. 2017; Myers-Smith et al. 2020). Detected greening trends in treeline ecotones of the Scandes are attributed to expanding shrub vegetation and densification of previously sparse vegetation cover (Franke et al. 2019). Increased greening was also observed in the Pyrenees as long as productivity of alpine grasslands is not compromised by high stocking rates (Gartzia et al. 2016). Complex interrelationships between climate and land use change determine productivity and biomass in Mediterranean mountains, with the current balance being still towards greening since land abandonment is still buffering the browning drivers (Pausas and Millán 2019; Vicente-Serrano et al. 2020).

The emergence of longer and warmer growing seasons is not only associated with high-elevation plant communities producing more biomass, but also with plants and animals dramatically altering their phenology (Fig. 1.30) (Menzel et al. 2006; Amano et al. 2010; Fu et al. 2014; Garonna et al. 2014). Vitasse et al. (2009) showed for tree species in Pyrenean mountain forests that leaf unfolding is the major driver of extending the growing season with increasing temperature. Spring phenological phases, such as budburst and flowering, occur 20 days earlier at low elevations and 15 days earlier above 1000 m in the Swiss Alps than 50 years before (Defila et al. 2016). Considering the duration of the vegetation period at both elevations, the advance is much more pronounced at high elevation in the Alps (Güsewell et al. 2017). Xie et al. (2017) and Asam et al. (2018) detected correlations with interannual differences in snow cover duration. Climate warming not only affects the timing of phenological events but also the underlying patterns in phenology along environmental gradients. Vitasse et al. (2018) highlighted stronger phenological advance at higher elevations and showed that the elevation-induced shift in the time of leaf-out in four common tree species in the Swiss Alps between low and high elevation has contracted by 35% from the 1960s until

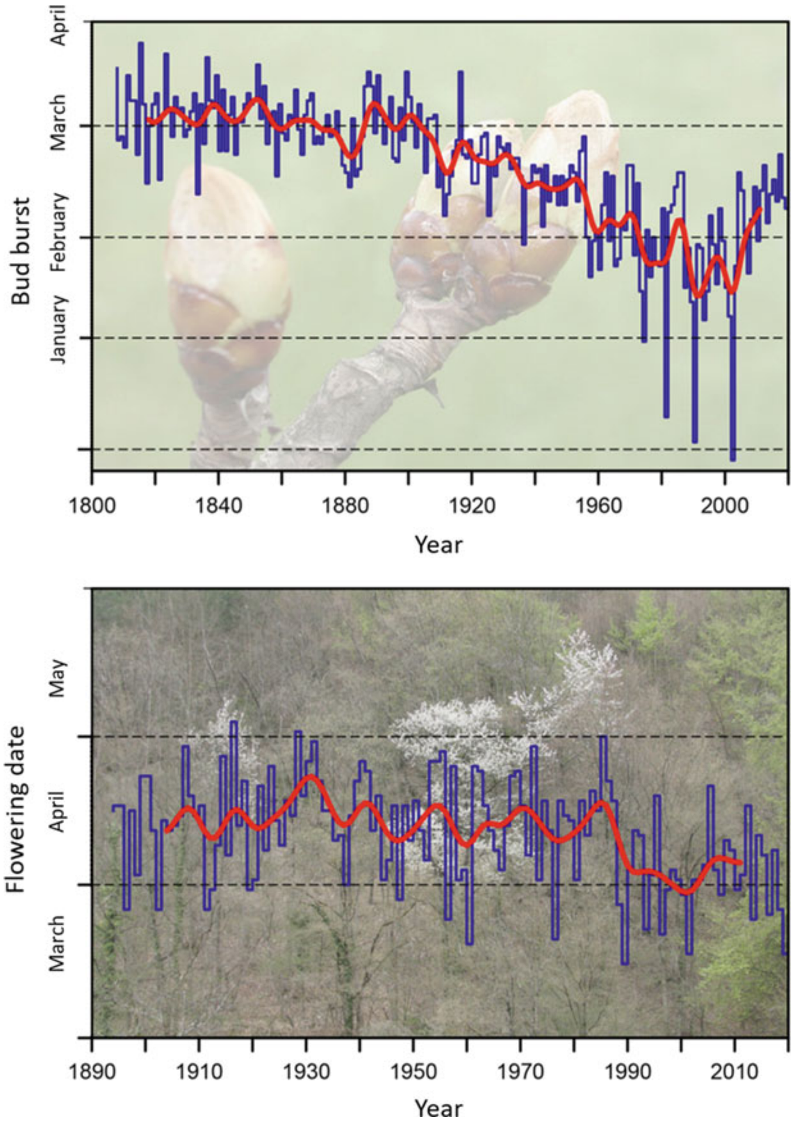


Fig. 1.30 Top: Bud burst of the horse chestnut (*Aesculus hippocastanum*) in Geneva, Switzerland, 1808–2020. Bottom: Flowering of the cherry tree (*Prunus avium*) in Liestal, Switzerland, 1894–2020; the red lines show the respective 20-year weighted average. (Modified from www.meteoschweiz.admin.ch)

today (Fig. 1.31). This increase in the rate of progression of spring leaf-out with elevation is mainly attributed to an increasingly insufficient number of chilling days at low elevations during warmer winters (days with mean temperature of 0–8 °C between November and mean leaf-out date), resulting in less pronounced phenological shifts (see also Asse et al. 2018). Thus, lowland trees are not keeping up with the pace of phenological advance of their conspecifics at higher elevations. The results of Vitasse et al. (2018) are of far-reaching significance in that they suggest that global warming has altered 'Hopkins' bioclimatic law' which specifies the progressive delay in tree leaf-out with increasing latitude, longitude, and elevation. Vandvik et al. (2018) analysed the alteration of this law at other elevation and environmental gradients across Europe and concluded that a change of this law occurs at broader scales, suggesting far-reaching consequences for species, communities, and ecosystems since community composition, trophic interactions, biochemical cycling and the like are affected. A distinct advance in spring phenophases has been observed over much of southern Fennoscandia during recent decades while high mountains areas and northern Fennoscandia showed a delay due to higher winter precipitation and longer snow cover in spring (Pudas et al. 2008; Wielgolaski and Inouye 2013). In the Mediterranean region, warm and dry springs have resulted in advances in flowering, leaf unfolding and fruiting dates, and in lengthening the growing season (Peñuelas et al. 2002; Gordo and Sanz 2010). However, severe drought conditions may reduce the length of the growing season and affect flowering phenology (Spano et al. 2013).

The anticipated shifts in climatic zones in Europe within the next decades (Jylhä et al. 2010) will be associated with further shifts of species distribution ranges and accelerated transformations of montane and alpine vegetation. To date, climate-induced shifts in biodiversity patterns including upward migration of plant species and transformations of plant communities have nowhere been studied in greater detail than in the European Alps. Long-term vegetation monitoring series are available from summit areas in the Alps, including detailed surveys dating back to the nineteenth century. First extensive resurveys of summit sites in the alpine-nival ecotone in the 1990s and 2000s provided compelling evidence of increasing vascular plant species richness on most of the summits and a general trend of upward migration in the range of up to more than 100 elevational metres per century, given that appropriate migration corridors are available (Grabherr et al. 1995; Pauli et al. 2001; Bahn and Körner 2003; Grabherr 2003; Walther et al. 2005; Holzinger et al. 2008; Parolo and Rossi 2008; Vittoz et al. 2008a; Stöckli et al. 2011; Wipf et al. 2013). Bergamini et al. (2009) reported a significant upslope migration also for bryophytes (24 m per decade). Based on a large dataset, Frei et al. (2010) confirmed a strong trend towards increasing species richness per summit and found many plant species at an elevation higher than any reported occurrence in the region one century ago. Their results also pointed to a more heterogeneous response at lower range limits, suggesting species-specific differences in response patterns. Increasing species richness of alpine plant communities, albeit without distinct upward shift processes, was reported in a resampling study (1953–2003) by Cannone and Pignatti (2014).

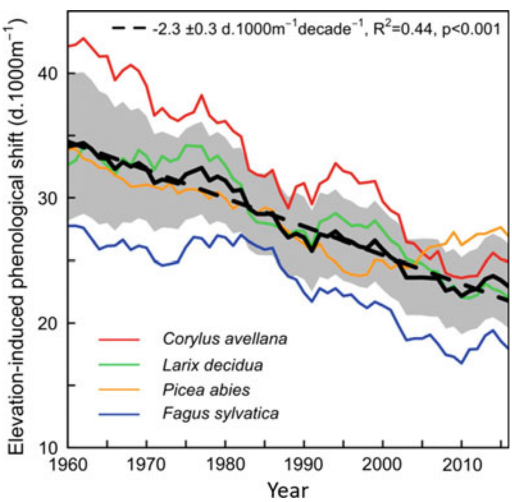


Fig. 1.31 Changes of the elevation-induced phenological shift for four tree species over the period 1960–2016 in Switzerland; eleven-years moving averages are represented (black line) and slope of the linear regression (dashed line) and SD (grey area) across species is also shown. (Modified from Vitasse et al. 2018)

After establishing the GLORIA network in the mountains of Europe, increasingly comprehensive and detailed studies on vegetation dynamics in the alpine-nival ecotone of the Alps have been conducted. It became evident that distinct increases of alpine pioneer species are accompanied by significant declines of subnival-nival plant species, suggesting range contractions at their rear edge (Pauli et al. 2007, 2012). The pattern of expansions of more thermophilic species to higher elevations and concurrent declines of cold-adapted, long-established species of the upper alpine and subnival belt was corroborated in the first pan-European GLORIA resurvey study which substantiated a widespread thermophilization process in alpine vegetation after a period of only seven years (Gottfried et al. 2012). The most compelling evidence of a continent-wide acceleration in the rate of

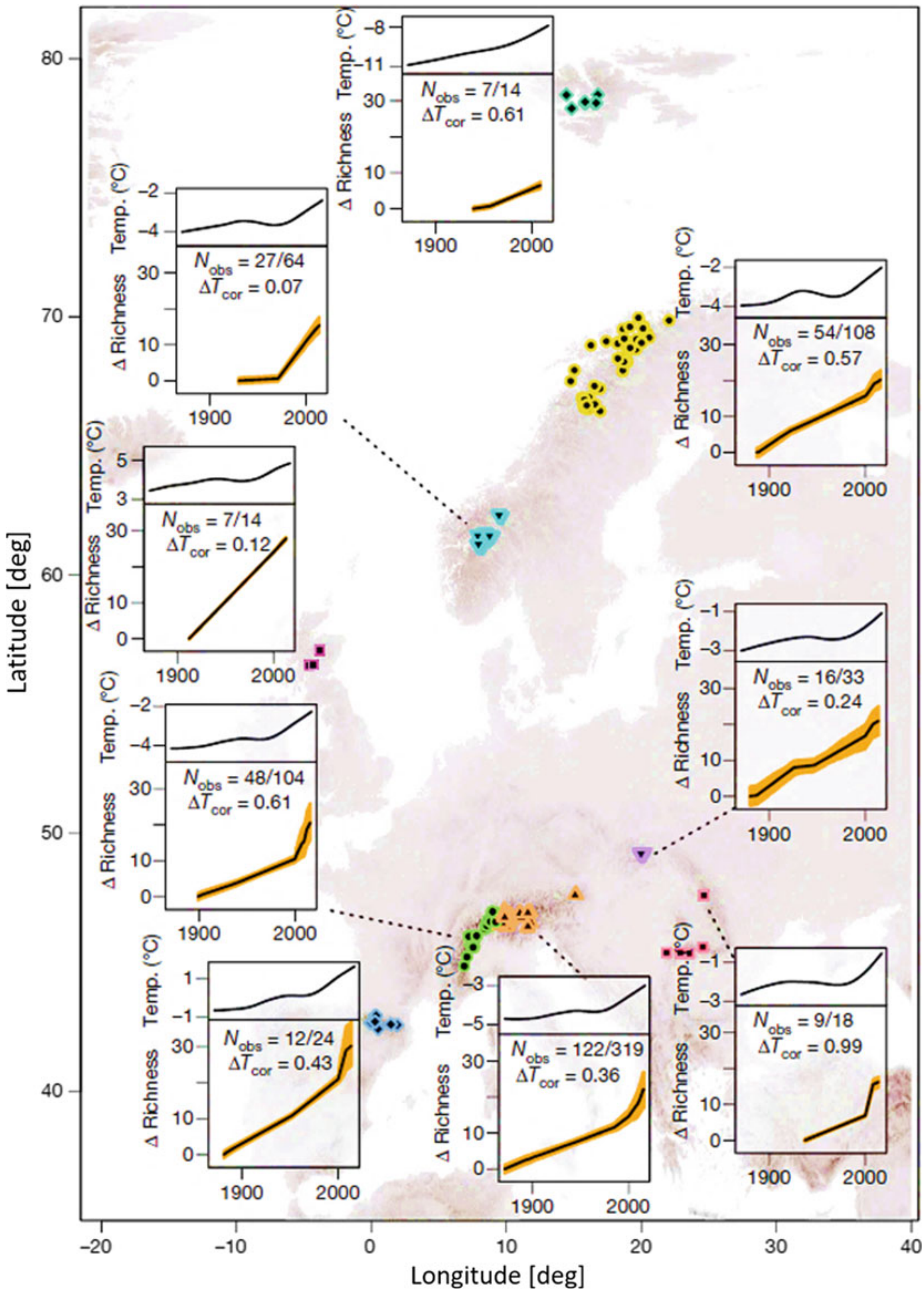


Fig. 1.32 Average species richness change on European mountain summits over time (lower parts of inset panels) compared to mean annual temperature over time (upper part of inset panels); N_{obs}, number of summits/surveys within the mountain region providing data for the panel; correlation between rate of change in species richness and rate of change in temperature (ΔT_{cor}) is positive for all mountain regions. (Modified from Steinbauer et al. 2018)

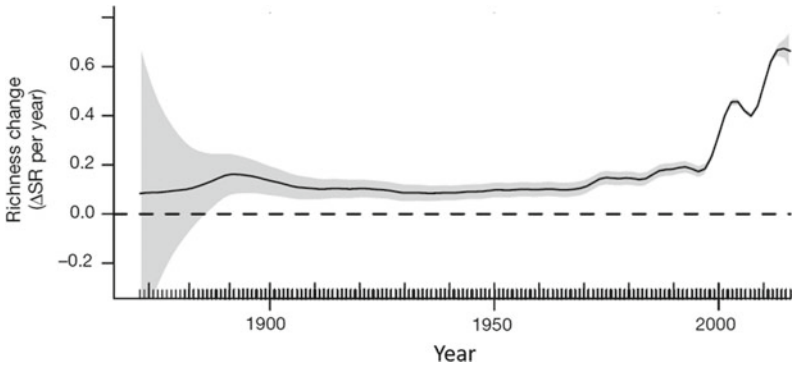


Fig. 1.33 Rate of change in species richness on European mountain summits over time (mean, black line); positive values indicate an increase in species richness on summits and negative values indicate a decrease; rates (ΔSR per year = $(SRt2 - SRt1)/(t2 - t1)$ where SR is species richness and t is time) were averaged across all summits. (Modified from Steinbauer et al. 2018)

increase in plant species richness at high elevations was provided by Steinbauer et al. (2018), who evaluated a dataset of repeated plant surveys from 302 mountain summits across Europe, spanning 145 years of observation. Species enrichment between 2007 and 2016 was five times higher than fifty years ago and found to be strikingly synchronized with accelerated global warming (Figs. 1.32, 1.33).

A recent resurvey of the largest alpine to nival permanent GLORIA plot site in the Alps after two decades showed increasing vascular plant species richness over the entire period while vegetation cover decreased due to the decline of cryophilic species. The increase in richness was reduced in the second decade when disappearance events of more cryophilic species became more numerous, suggesting an accelerating transformation towards more thermophilic and more drought-adapted vegetation (Lamprecht et al. 2018). European GLORIA data also showed larger increases in species richness and higher numbers of newly established species on the warmest slopes of summit zones (east- and south-facing slopes) (Winkler et al. 2016). In the context of increasing maladaptation to warmer habitat conditions and a successive trailing-edge decline of cryophilic species and a leading-edge expansion of more thermophilic species, Steinbauer et al. (2020) highlighted that cryophilic species declines preceded the onset of strong competition pressure from advancing species, suggesting physiological constraints of cold-adapted specialists in adapting to a warmer temperature regime having greater significance than competitive displacement. Another comprehensive resurvey of more than 1500 vegetation plots confirmed that elevational ranges of cold-adapted species tended to contract, while those of thermophilic species which showed a marked increase in species abundance tended to expand (Rumpf et al. 2018). The results of this study suggest that 'losers' of recent range dynamics are overrepresented among high-elevation, cryophilic species with low nutrient demands, and that these species face the risk of displacement by novel, superior competitors that move up faster than they themselves can escape to even higher elevations. The extinction risk of high-elevation plants is alleviated, on the other hand, by topographic complexity and the high diversity of microhabitats, facilitative neighbour interactions, and the longevity of many mountain plants (Scherrer and Körner 2011; Rixen and Wipf 2017; Graae et al. 2018). Nevertheless, long-term warming effects will increase mountain-top extinctions, in particular among endemics, once the accumulated extinction debt will be paid off (Dullinger et al. 2012).

Range shifts, species enrichment on mountain summits, and plant community thermophilization are pan-European phenomena, documented also in Scandinavia (Klanderud and Birks 2003; Kullman 2007a; Odland et al. 2010; Michelsen et al. 2011; Felde et al. 2012), the Carpathians (Czortek et al. 2018; Kobiv 2018), and in some Mediterranean mountain environments (Molero Mesa and Fernández Calzado 2010; Pérez-García et al. 2013; Evangelista et al. 2016; Stanisci et al. 2016; Frate et al. 2018). Grytnes et al. (2014) confirmed the widespread upward shifting of species in their pan-European survey and found elevational shifts in range limits not as clearly related to climatic warming as latitudinal shifts. The thermophilization process on Mediterranean mountain summits is largely characterized by declining species richness, with the loss of high-elevation species, often endemics (Kazakis et al. 2007), outweighing the new appearance of more widespread species. This shift in species composition is attributed to combined effects of increasing temperature and decreasing precipitation in spring and summer (García-Romero et al. 2010; Pauli et al. 2012; Fernández Calzado and Molero Mesa 2013; Jiménez-Alfaro et al. 2014; Giménez-Benavides et al. 2018). Among alpine habitats, snowbeds experience substantial changes and a general homogenization in species composition due to strongly modified snow cover and soil moisture conditions, with the invasion of shrubs and generalists from surrounding grasslands, and increasing species richness and plant cover (Virtanen et al. 2003; Kullman 2007b; Matteodo et al. 2016; Liberati et al. 2019). The ongoing glacier retreat in European mountains extends the surface area of recently deglaciated terrain which is already colonized by first bryophytes and vascular plants after one to three years (Cannone et al. 2008; Burga et al. 2010). Climate warming enhances the establishment of plants on glacier forelands, favouring also other than true pioneer species, and accelerates successional stages. Successful establishment depends in particular on the grain size of the substrate, the associated water capacity, the available gene pool, and on the distance to the seed source (Erschbamer et al. 2008; Erschbamer and Caccianiga 2016; Schumann et al. 2016; Franzén et al. 2019). Compared to vegetation dynamics in glacier forelands 100 years ago, Fickert et al. (2017) assessed an accelerated colonization and more species involved in early colonization. Examples in the Alps (Goldbergkees, Jamtalferner) also show that after 100 years of primary succession roughly 80% of the ground is covered by plants while the number of species (vascular plants) increases to 40–50 per 10 m^2 sample site (Fickert and Grüninger 2018; Fischer et al. 2019).

Enhanced tree growth, intense regeneration and infilling of gaps are common trends in European treeline ecotones (Rolland et al. 1998; Batllori and Gutierrez 2008; Vittoz et al. 2008b; Hofgaard et al. 2009; Holtmeier and Broll 2011; Vitasse et al. 2012; Mathisen et al. 2014; Camarero et al. 2015, 2017; Kaczka et al. 2015; Hedenås et al. 2016; Jochner et al. 2017, 2018; Malfasi and Cannone 2020). Positive climate-growth relationships were also found for shrubs above treeline in most studies, suggesting densification of shrub stands and further expansion (Hallinger et al. 2010; Rundqvist et al. 2011; Francon et al. 2017; Vowles et al. 2017; Weijers et al. 2018). Most alpine treelines have advanced to higher elevations over the past century (Fig. 1.34). Some studies documented substantial treeline shifts, with gains in elevation of 70–100 m or more (Meshinev et al. 2000; Peñuelas and Boada 2003; Cudlin et al. 2017). A recent remote sensing-based study indicated widespread strong treeline advances from the western Pyrenees to the eastern Carpathians over the last 40 years, with eastern European mountains showing the most remarkable changes (Fig. 1.35) (Dinca et al. 2017). In the Swedish Scandes, treeline shifts to even more than 200 m were assessed (Kullman 2007b, 2018, 2019; Kullman and Öberg 2009) as well as upward migration of thermophilic tree species such as *Betula pendula* and *Alnus glutinosa* and of true temperate tree species (*Quercus robur*, *Ulmus glabra*, *Acer platanoides*) into treeline ecotones (Kullman 2008). Many authors refer to correlations of advancing treelines with increases in mean temperatures. The effects of declining land use intensity, however, are certainly often involved, and appear to explain most cases of particularly significant treeline shifts, at least in temperate and southern European mountains (Gehrig-Fasel et al. 2007; Chauchard et al. 2010; Kulakowski et al. 2016; Treml et al. 2016; Cudlin et al. 2017; Kyriazopoulos et al. 2017; Wielgolaski et al. 2017; Wieser et al. 2019). It is evident, for instance, that the cessation of land use has been the most important driver of the large-scale forest expansion at higher elevations in the Alps over the past century (Mietkiewicz et al. 2017). Land use legacies are also considered the major drivers of stand densification processes and treeline advances at anthropogenic Mediterranean treelines (Palombo et al. 2013; Ameztegui et al. 2016; Vitali et al. 2019). Likewise, recruitment patterns in treeline ecotones and treeline advances in northern Europe are not infrequently correlated with impacts of reduced reindeer grazing or other abandoned human disturbances (Bryn 2008; van Bogaert et al. 2011; Aakala et al. 2014; Potthoff 2017).

Biotic responses are pervasive at mid- and lower elevations, though less obvious compared to alpine or treeline environments. General trends include upslope and northward range shifts (Lenoir et al. 2008; Amano et al. 2014), increases of lowland and thermophilic species, and decreases of cold-tolerant species of higher elevations at rear edges of their ranges at lower elevations (Lenoir et al. 2010; De Frenne et al. 2013). Significant upslope shifts over short time periods can be observed in different taxonomic groups as data from the national biodiversity monitoring programme of Switzerland show (Roth et al. 2014). Drought stress and climate-induced disturbances result in vegetation shifts, increasing forest damage and canopy mortality (Martínez-Vilalta and Lloret 2016; Senf et al. 2018). Mountain forests have responded faster over recent decades in terms of shifts in species distribution and plant community composition than lowland forests (Bertrand et al. 2011). A comprehensive analysis in western European temperate and Mediterranean mountains yielded the result of a significant upward shift in species optimum elevation over the twentieth century, averaging 29 m per decade (Lenoir et al. 2008). In Swiss forests, Küchler et al. (2015) detected a strong signal of upslope shift in the understorey

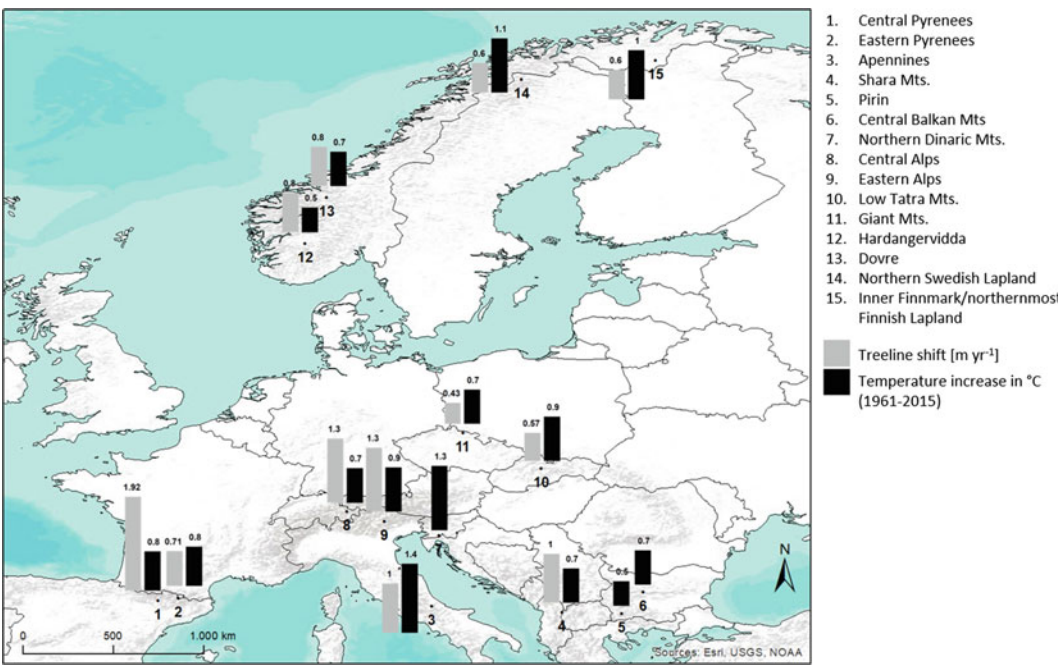


Fig. 1.34 Temperature increase in °C (1961–2015) and treeline shift in m yr^{-1} (different time periods between 1915 and 2015) for selected European mountains (after data in Cudlin et al. 2017)

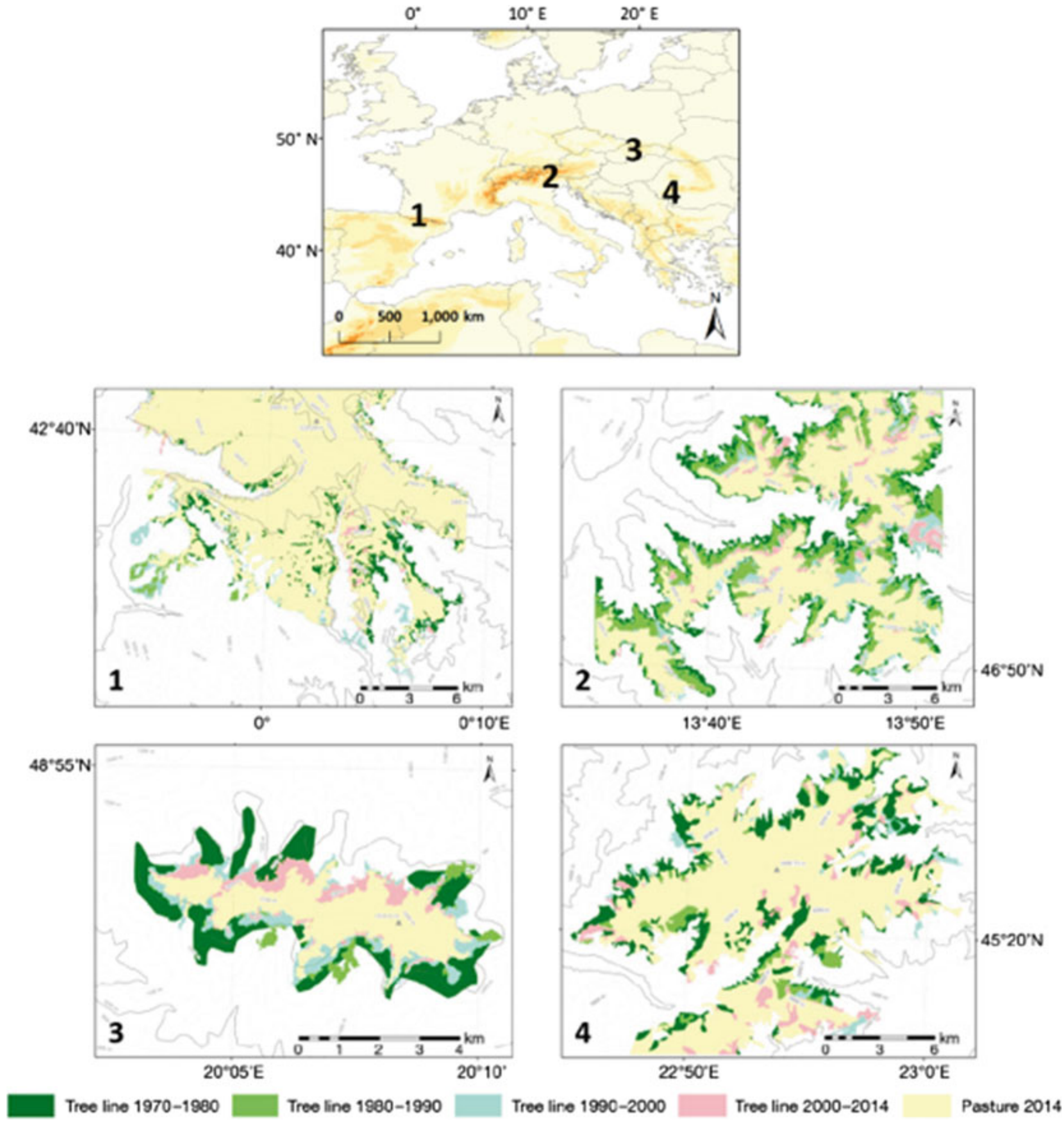


Fig. 1.35 Spatial and temporal dynamics of forest line and cover in 4 European mountain sites obtained from supervised land cover classification of Landsat satellite data between 1970 and 2015; dynamics are represented using a 10 yr time-step interval for (1) Ordesa and Monte Perdido National Park, Spain; (2) Nockberge Biopshere Park, Austria; (3) Low Tatra Park, Slovakia; and (4) Retezat National Park, Romania. (Modified from Dinca et al. 2017)

vegetation of about 10 m per decade since the mid-twentieth century. Significant upslope shifts were observed for single temperature-sensitive species. Dobbertin et al. (2005) resurveyed pine mistletoe (*Viscum album* ssp. *austriacum*) occurrences in pine forests of the European Alps and showed that the current upper limit is roughly 200 m above the limit found 100 years ago. Some evidence suggests that elevational shifts in European forest belts below the treeline are only partly driven by climate warming, and that forest successional changes such as the

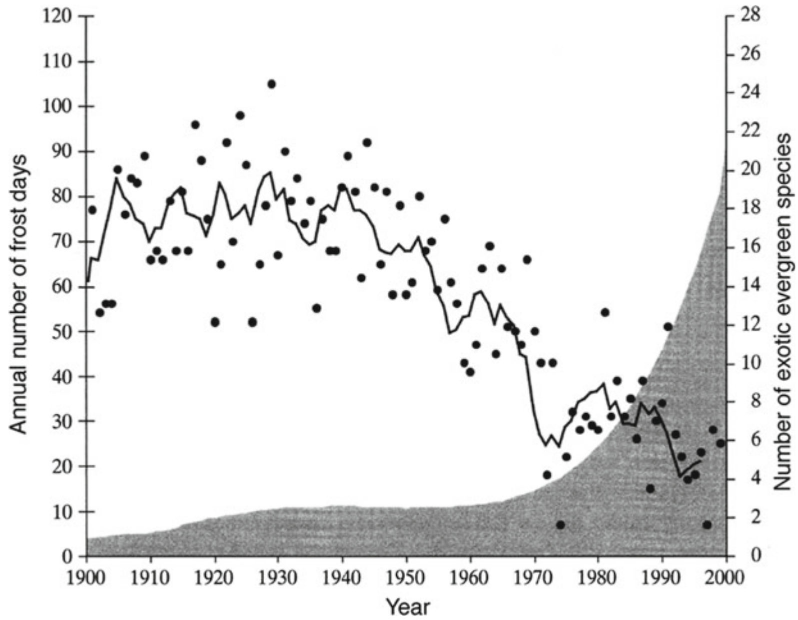


Fig. 1.36 Increase of exotic evergreen species under increasingly milder winter temperatures on the lower south slope of the European Alps; dots: annual number of frost days; solid line: smoothed 5-year averages. (Modified from Walther et al. 2002)

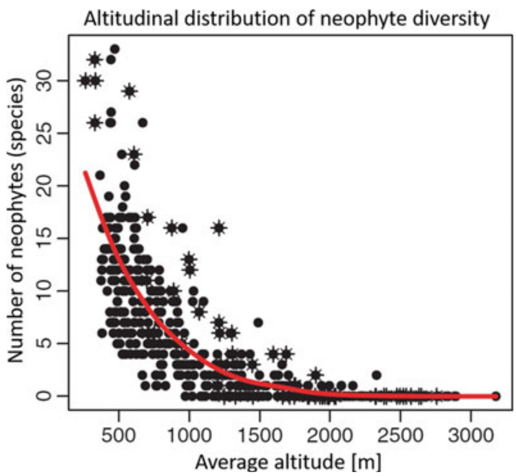


Fig. 1.37 Altitudinal distribution of neophytes in Switzerland based on data from the Swiss Biodiversity Monitoring Programme. (Modified from Nobis 2008)

closure and maturation of forest stands, associated with agricultural land abandonment, play a major role (Bodin et al. 2013).

A well-known example of thermophilization of temperate forests in the southern Alps is the increase in abundance and frequency of indigenous evergreen broadleaved (laurophyllous) species which become increasingly competitive with lengthening of the growing season and decreasing number of frost days (Fig. 1.36) (Walther 2001). Even exotic evergreen species including dwarf palms (*Trachycarpus fortunei*) have succeeded in colonizing these forests, driven by mild winter temperatures and reduced frost occurrence (Walther et al. 2007). Meanwhile, *Trachycarpus fortunei* is regularly recorded and locally established in northern Switzerland and further north (Essl 2019). Shifts in species composition of communities and species richness patterns are increasingly altered by such non-native species invading European mountains. Non-native species affect native species richness and community dissimilarity, resulting in biotic homogenization (Haider et al. 2018). In addition, invasive species affect trophic levels, biotic interaction networks and other ecosystem properties (Gallien et al. 2017). With increasing elevation, however, non-native species decline in probability of occurrence (Fig. 1.37), and their high-elevation range limits do expand, but not rapidly (Becker et al. 2005; Pyšek et al. 2011; Seipel et al. 2016; Siniscalco and Barni 2018). Thermophilic species are prevalent in the alien species pool in the European Alps which has only a small number of mountain specialists (Dainese et al. 2014). In northern European highlands and mountain ranges, an increased risk of non-native plant colonization was assessed, mainly driven by human-mediated dispersal (Wasowicz 2016).

Critical transitions of forest ecosystems in the Alps with potentially severe consequences for ecosystem services may already occur at warming levels of around +2 °C (Elkin et al. 2013; Albrich et al. 2020). Such substantial transitions, for instance, the progressive replacement of cold-temperate ecosystems (*Fagus sylvatica* forests) by Mediterranean ecosystems (*Quercus ilex* forests) from lower elevations during the twentieth century were reported from Mediterranean mountains (Peñuelas and Boada 2003). Rear-edge replacement of Mediterranean fir species (*Abies pinsapo*, *A. cephalonica*) by more drought-resistant pine species (*Pinus halepensis*) also indicate a drought-induced shift in dominance patterns of woodland vegetation (Linares et al. 2009; Sarris et al. 2011). Increasing duration and intensity of drought periods have negative impacts on Mediterranean forests, resulting inter alia in declining tree growth trends, crown condition decline, and increasing tree mortality rates (Carnicer et al. 2011; Linares et al. 2011; Galván et al. 2014).

America

In high latitudes of North America, remote sensing data provide evidence for heterogeneous greenness changes. While the long-term satellite record (1982–2019) in the Arctic indicates greening, the interannual variability in greenness has increased in recent years and browning trends in some regions are increasingly observed (Phoenix and Bjerke 2016; Lara et al. 2018; Frost et al. 2020; Myers-Smith et al. 2020). NDVI analyses in the boreal zone show that areas with productivity decreases have gained predominance in recent decades. While in maritime regions with sufficient precipitation a general greening trend as a response to rapid warming prevails (Ju and Masek 2016), also in alpine treeline ecotones in the Boreal Cordillera ecozone (Bolton et al. 2018), the positive effect of increased temperatures in many dry continental regions is meanwhile offset or even exceeded by the disadvantage of increased evapotranspiration due to temperature rise. The areal fraction exhibiting browning trends in recent years is associated with high winter temperatures and frost drought, fire, or drought limitations (Beck et al. 2011; Beck and Goetz 2012). Tree-ring analyses corroborate drought-induced growth declines in boreal forests of the western Canadian interior (Hogg et al. 2017). While vegetation productivity in high latitude mountain regions still shows a strong dependency on growing season temperature, temperature-induced drought stress has become an important limiting factor in interior mountain regions unless the ongoing warming is accompanied by a significant increase in precipitation (Verbyla and Kurkowski 2019). Dendroecological studies in high-elevation forests and at alpine treelines in Alaska and Yukon point to complex growth responses to continued warming and small-scale differences in climate-growth relationships, with soil moisture often mediating the sensitivity to warm temperatures and affecting productivity (Wilmking et al. 2004; D’Arrigo et al. 2008; Ohse et al. 2012; Wolken et al. 2016; Sherriff et al. 2017; Tei et al. 2017; Dearborn and Danby 2018; Lange et al. 2020). NDVI increases prevail in the Canadian Rocky Mountains (Jiang et al. 2016). However, remote sensing-based studies across the Rocky Mountains and the western US also found water limitation, in particular early summer drought conditions, to impose critical controls on vegetation productivity under continued atmospheric warming (Sloat et al. 2015; Berkelhammer et al. 2017; Berner et al. 2017; Wainwright et al. 2020). In the southwest region of the US, NDVI increases at higher elevations in the southern Rocky Mountains and the Sierra Nevada contrast with drought-induced decreases at lower elevations and in the south of California and the Four Corner States, with recent drought periods accentuating the elevational transition from water-limited to temperature-limited ecosystems (Herrmann et al. 2016; El-Vilaly et al. 2018; Dong et al. 2019). Recent prolonged drought periods facilitated fire severity and extensive tree dieback at low and mid-elevations

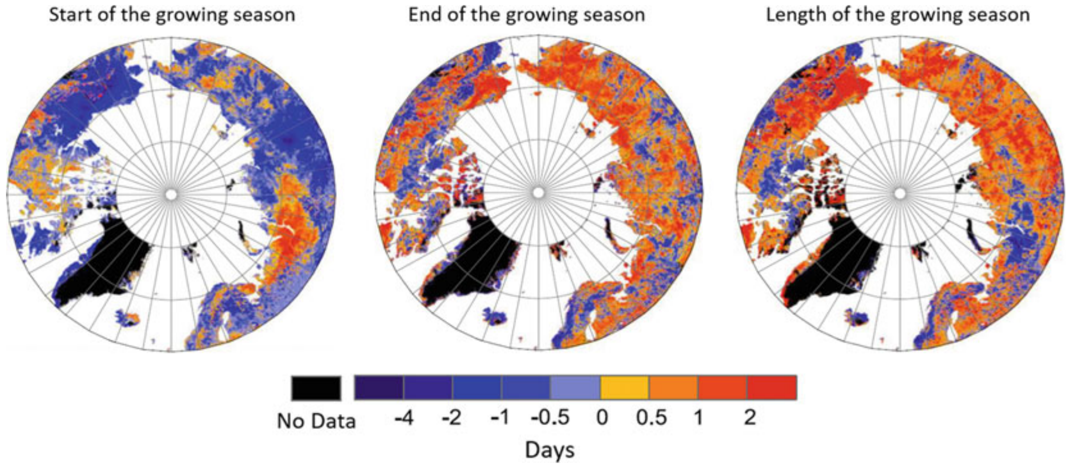


Fig. 1.38 Spatial patterns of the linear trends in SOS (start of the growing season), EOS (end of the growing season) and LOS (length of the growing season) from 2000 to 2010 based on MODIS data; positive values (warm colors) indicate later onset (SOS), later finish (EOS) and longer duration (LOS) of the growing season. (Modified from Zeng et al. 2011)

(Byer and Jin 2017; Potter 2017; Crockett and Westerling 2018).

The thermal potential growing season in temperate and high northern latitudes has lengthened over recent decades (Barichivich et al. 2013; Melaas et al. 2018). This trend is increasing and regionally accelerating. According to MODIS data, the growing season length in the North American Arctic increased by about 14 days between 2000 and 2010, with a significantly earlier start of the growing season of c. 11.5 days (Fig. 1.38) (Zeng et al. 2011). Species-level phenological shifts result in a substantial reshaping of various temporal components of entire plant communities, affecting patterns of temporal overlap among (mutualistic) species and interactions within trophic levels and beyond (phenological mismatch). Notwithstanding the recognition that photoperiod constrains spring plant phenology in alpine regions and the extent to which the growing season can lengthen is limited (Ernakovich et al. 2014), considerable phenological shifts have been assessed at higher elevations. Using a unique long-term record of flowering phenology from the Colorado Rocky Mountains, CaraDonna et al. (2014) showed that the diversity of species-level phenological shifts contributed to altered coflowering patterns within meadow communities, a redistribution of floral abundance across the season, and an expansion of the flowering season by more than one month between 1974 and 2012 (Fig. 1.39). Large shifts in the phenology of rare Colorado Rocky Mountain plants were found by Munson and Sher (2015), who assessed an acceleration of flowering dates by more than 40 days since the late 1800s. With regard to plants of alpine habitats, they found high spring temperatures explaining the accelerated phenology. Correspondingly, flowering initiation in alpine plants of the Colorado Front Range was observed to occur earlier with earlier snowmelt (Inouye and Wielgolaski 2013; Winkler et al. 2018), potentially generating a mismatch in the seasonal timing of interacting organisms, e.g. plants and pollinators (Forrest and Thomson 2011). In the Catalina Mountains of south-central Arizona, precipitation appears to play a much larger role for flowering patterns in spring and summer than further north (Crimmins et al. 2013). Shifts of morphological and physiological phenophases of trees in drier habitats seem to be less pronounced (Hallman and Arnott 2015), despite a considerable lengthening of the growing season (Tang et al. 2015). Climate warming-induced advance in the timing of spring onset is consistent across

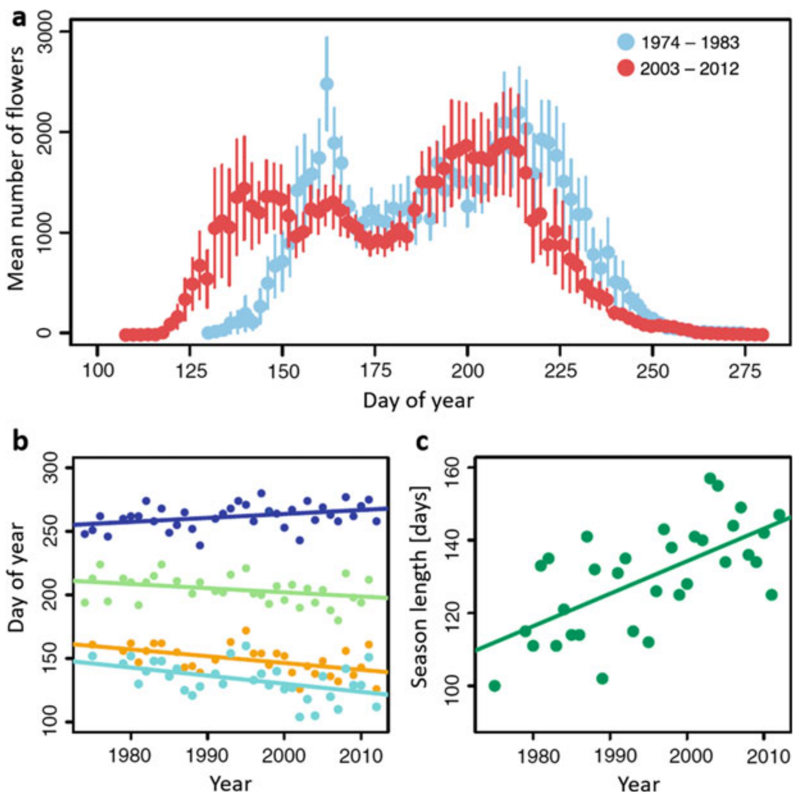


Fig. 1.39 Aggregate community-level shifts in flowering phenology; **a** comparison of the season-wide flowering curves for the first and last 10 years of the dataset; each dot is the 10-y mean number of flowers; **b** phenological shifts through time for first flowering of the community (cyan), last flowering for the community (dark blue), and timing of community-level spring peak (orange) and summer peak (green); each dot represents a community-level phenological measure in 1 y; **c** change in the length of the flowering season; each dot represents the total number of days on which open flowers were present in each year. (Modified from CaraDonna et al. 2014)

the mountain regions of the western and north-eastern US (Ault et al. 2011; Schwartz et al. 2013).

As presented for Europe, upward migration of plant species and transformation of montane to alpine plant communities is pervasive across North American mountain ranges as long as the expansion of distribution ranges is not constrained by a decreased water balance and drought stress or other non-thermal drivers (cf. Rapacciuolo et al. 2014). Elmendorf et al. (2015) analysed changes in plant community composition from repeat sampling and experimental warming studies in varied arctic and alpine habitats and found a general increase in the relative abundance of species with a warmer thermal niche. Over vast areas of arctic mountain ranges, climate warming-induced significant changes in plant community composition have occurred (Danby et al. 2011), in accordance with a strong trend towards subarctic forest-tundra ecotone advance which, however, is rarely capable to keep pace with climate change within the twenty-first century (Rees et al. 2020). The velocity of latitudinal tree migration which is predominantly northward is also lower than the velocity of climate warming in temperate and boreal forests in eastern Canada and the eastern US, suggesting a constrained capacity to track climate warming (Boisvert-Marsh et al. 2014; Fei et al. 2017; Sittaro et al. 2017).

Upward range expansion of species, induced or facilitated by climate warming, appears to be a common change pattern across the Rocky

Mountains (Landhäusser et al. 2010; Sproull et al. 2015), while climate change effects on the abundance and distribution of tree species are mediated in particular by ecological disturbances such as wildfires and insect outbreaks (Keane et al. 2018; see also Littell et al. 2013 for the Cascade and Coast ranges). A thermophilization of montane to alpine plant communities is reflected in the results of a resurvey in the Colorado Rocky Mountains (2600 to 4100 m) after 65 years: Zorio et al. (2016) detected significant changes in species composition and dominance, with an upward shift of species' mean elevation of 41 m. Many species from lower elevations, in particular graminoids and shrubs, expanded their ranges into new communities. A study on shrub encroachment into alpine tundra in the Colorado Front Range showed a shrub cover (*Salix planifolia*, *Salix glauca*) expansion by 441% over 62 years (1946–2008) on a 18 ha study site (Formica et al. 2014). Here, data from other long-term monitoring plots (20 + years) showed increasing species and functional diversity (Spasojevic et al. 2013). Most resurvey studies in North American mountain ranges reveal thermophilization processes of plant communities. Examples include shifts in herb community composition in the Klamath-Siskiyou Mountains (California/Oregon) over more than 50 years (Damschen et al. 2010), expansion of subalpine species into alpine plant communities in California's White Mountains over 49 years (Kopp and Cleland 2014), and shifts in plant distributions with elevation in southern California's Santa Rosa Mountains over 30 years (Kelly and Goulden 2008). The average elevation of dominant plant species was found to have shifted upslope by c. 65 m as a consequence of changes in the regional climate in the latter study. Increased dominance of evergreen oaks in foothill woodland and montane hardwood forest of the Sierra Nevada also suggests thermophilization under warmer and drier conditions (Dolanc et al. 2014). Changes in the geographic distributions of species in the US Southwest mountain ranges, strongly associated with observed changes in climate, were highlighted in general by Fleishman et al. (2013). Range shifts are documented for diverse groups of animals as well (Moritz et al. 2008; Forister et al. 2010), including pathogens, thus increasing the risk of forest infestations at higher elevations (Bentz et al. 2010).

Corresponding to recent results from the European Alps, Lesica (2014) found plant species restricted to highest elevations in the Montana Rocky Mountains to decline in abundance, while lower-elevations species expand their range upslope with climatic warming. In accordance with these declines, long-term monitoring (1988–2014) of arctic-alpine and boreal plant species at their southern range limit in the Rocky Mountains revealed overall declining population trends (Lesica and Crone 2017). In the Santa Catalina Mountains of southern Arizona, montane plant species showed significant upward shifts of lower elevation range boundaries and elevational range contractions over the past five decades, attributed to the conditions of decreasing precipitation and increasing temperatures (Brusca et al. 2013). Warming-mediated drought stress is also driving upslope retreat of *Pinus ponderosa* in the Sierra Nevada, where low-elevation ponderosa pine forests have been replaced by montane hardwood forests and annual grasslands (Field et al. 2016). Range shifts in montane forests were reported as well from eastern US and Canadian mountain ranges. Beckage et al. (2008) found a rapid upward movement of the northern hardwood-boreal forest ecotone in the Green Mountains (Vermont) from 1964 to 2004, while Savage and Vellend (2015) detected significantly increasing mean elevations of species distributions (9 m/decade) on Mont Mégantic (southern Québec) between 1970 and 2012 (Fig. 1.40), associated with biotic homogenization at higher elevations.

Upslope elevational range shifts have also been assessed for tree species at alpine treelines. Accordingly, observational studies in many mountain ranges detected a treeline advance. Climatic treelines which still show persistence are expected to shift to higher elevations in the mid- or long term, unless non-thermal site factors do not prevent advances. In particular, limitations to seedling recruitment with warming can

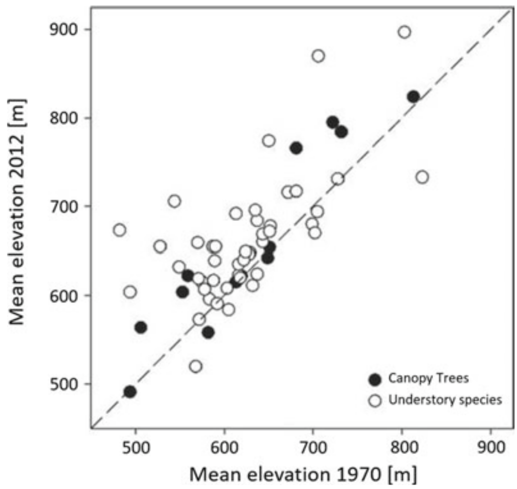


Fig. 1.40 Mean abundance-weighted elevation of species distributions in 1970 and 2012; each point represents a single species ($n = 45$ understorey species, $n = 13$ canopy trees); distributions of species above the diagonal 1:1 line increased in mean elevation, and vice versa. (Modified from Savage and Vellend 2015)

constrain the pace of tree range shifts at treelines (Conlisk et al. 2017; Elliott 2017; Kueppers et al. 2017). Rapid upward advance of woody vegetation over the past 60 years (Dial et al. 2007, 2016; Terskaia et al. 2020), and significant increases in treeline elevation and stand density over the past 100-plus years were detected at several boreal-subarctic alpine treelines in Alaska and Yukon (Lloyd and Fastie 2003; Danby and Hik 2007; Stueve et al. 2011). Other alpine treelines at higher latitudes indicate moderate upslope shifts (de Lafontaine and Payette 2012; Trant and Hermanutz 2014), or show ongoing treeline dynamics, for instance by stand infilling, but more or less stagnating elevational positions (Mamet and Kershaw 2012). A recent study, covering nine alpine treeline ecotones in the Canadian Rocky Mountains, revealed a widespread increase in radial growth, establishment frequency, and stand density since the mid-twentieth century, and a concurrent treeline advance at all sites, averaging 40–50 m (Davis et al. 2020). Empirical evidence of increases in tree density and treeline advance since 1950 across a latitudinal gradient of 1100 km in the Rocky Mountains was provided by Elliott and Kipfmueller (2011), Elliott (2012), and Elliott and Petruccelli (2018), with treeline advance ranging between 39 and 140 m. As elsewhere, however, treeline dynamics in the Rocky Mountains is complex, with site- and species-specific responses modifying the general trend of treeline advance (Malanson et al. 2007, 2009; Holtmeier 2009; Elliott 2011; Holtmeier and Broll 2010, 2012, 2017b; Davis and Gedalof 2018; Elliott et al. 2020). Across five mountain ranges of the Great Basin, Smithers et al. (2018) found a mean vertical treeline elevation shift of c. 20 m since 1950, associated with upslope expanding ranges of *Pinus longaeva* and *Pinus flexilis*, whose recruitment and radial growth is controlled by water limitations that complexly interact with temperature (Millar et al. 2015). Millar et al. (2004) documented expansion of subalpine conifers in the central Sierra Nevada, reflected in snowfield and subalpine meadow invasion, branch elongation, and vertical branch release. Here, a resampling-based study revealed a densification of high-elevation forests over the past 75 years with widespread, multiple-species increases in density of young trees, with interactions between water balance and disturbance factors playing a crucial role in future shifts in vegetation composition and structure (Dolanc et al. 2013).

As a result of the colonization from Europe, non-native plant species richness is highest in New World regions, with the US having the highest number of recorded invasive alien species globally (Seipel et al. 2012; Turbelin et al. 2017). In North American mountain ranges, as elsewhere, the abundance of non-native plant species declines with increasing elevation, while their invasibility is facilitated by climate warming. Relative to lowland ecosystems, alpine environments host few non-native plants (Alexander et al. 2016; Malanson 2020). The density of non-native plant species is related to the density of native plant species (Stohlgren et al. 2005), suggesting an increased invasion risk in national parks and other protected areas with high native species richness and high percentage of threatened and endangered plants (Allen et al. 2009). Increasing rates of exotic species introductions are expected in the boreal zone as a result of human activities and climate change (Sanderson et al. 2012). In the Rocky Mountains, dominant exotic species comprise intentionally introduced Eurasian grasses (e.g. *Phleum pratense*, *Poa pratensis*, *Bromus tectorum*, *Bromus inermis*) and herbs (e.g. *Melilotus*, *Medicago*, *Trifolium*, *Verbascum*, *Taraxacum* spp.) which particularly occur along roadways and invade disturbed sites primarily in montane steppes and open forests (Weaver et al. 2001; Pollnac et al. 2012). In the southern Sierra Nevada, non-native species have their main range of elevational occurrence between 1500 and 2000 m, only a few alien species have been ecologically successful invaders in subalpine/alpine ecosystems (Rundel and Keeley 2016). Invasive grasses such as *Bromus tectorum* occur in subalpine forests (Brooks et al. 2016), but mainly invade lower elevations, in particular grazing- and fire-affected sites, causing significant changes in ecosystem composition, structure, and function (Blumler 2011; Grüninger 2015; Millar and Rundel 2016). The distribution of the most common exotic invasive species in California, *Centaurea solstitialis*, is mainly confined to elevations below 1200 m (Pitcairn et al. 2006). Non-native species are a prominent vegetation component on the tropical island of Hawai'i where these species are in an upward niche expansion phase. Exotic species showed a significant upward shift in both their upper and lower elevation limits, by 126.4 and 81.6 m, respectively, between 1970 and 2010 while native species shifted significantly upward in their lower elevation limit only (by 94.1 m), resulting in a drought stress-related range contraction (Koide et al. 2017).

The number of studies on biotic responses to climate change in Central and South American mountain ranges is still comparatively limited. In the Trans-Mexican Volcanic Belt, considerable upward shifts in species distribution ranges are projected, suggesting a high vulnerability of species due to limited habitat space available at higher elevations (Villers-Ruiz and Castañeda-Aguado 2013). Current geographic distributions of temperate/montane pines and oaks in Mexico will most likely decrease significantly (Gómez-Mendoza and Arriaga 2007). Climate change is also threatening montane cloud forests in Mexico. Ponce-Reyes et al. (2012) showed that climatically suitable areas will get lost for more than 90% of protected cloud forests, and that almost three quarters of the entire cloud forests could vanish by 2080. Concurrently, the respective area of suitable habitat for cloud forest species, e.g. small mammals, will be substantially reduced (Lorenzo et al. 2019). Analysing tree species composition in annually censused plots along an altitudinal gradient (70–2800 m) in Costa Rica, Feeley et al. (2013) observed directional compositional shifts, with increased relative abundance of lowland species in 90% of plots caused by disproportionate mortality of highland species. The results point to the significance of successful migrations in order to persist under future warming.

Spatio-temporal patterns of vegetation productivity and phenology along the Andes are highly heterogeneous, affected to a large extent by the moistening trend in the inner tropics and the drying trend in the subtropical Andes, by the precipitation and temperature anomaly patterns associated with ENSO, and by the steep W-E precipitation gradient in the southernmost Andes. South of 9° S, NDVI-based monitoring (1981–2011) alongside the Andes showed positive trends in productivity for temperate forests in Chile and subhumid/humid areas in Peru, Bolivia and Brazil, while arid/semiarid and subhumid vegetation types across Argentina, northern Chile and SE Bolivia showed negative trends (van Leeuwen et al. 2013). A reversal from greening to browning trends around the mid-1990s was assessed by Krishnaswamy et al. (2014). A longer growing season was indicated in southern Chile and southern Argentina. Bianchi et al. (2020) confirmed positive NDVI-temperature relationships over temperate forests in western N Patagonia, while these relationships are weaker east of the Andes and biome-specific. A NDVI analysis in Patagonia covering the period 2001–2016 revealed a greening trend over the western zone, and a drying trend over the eastern zone (Olivares-Contreras et al. 2019).

Tree-ring growth of *Nothofagus pumilio* in northern Patagonia is positively related to growing season temperature and negatively to precipitation at mesic and humid treelines, while at xeric treelines the opposite is observed (Lavergne et al. 2015). A study on the productivity dynamics of high Central Andean peatlands in the semiarid Chilean Altiplano over the past three decades (1986–2017) found more or less stable peatland productivity and a recent regional greening trend over the last seven years (Chávez et al. 2019). In the semiarid region of Chile, Glade et al. (2016) detected negative trends of vegetation productivity below 2000 m and positive trends for higher elevations, associated with an earlier start of the growing period in mountainous ecosystems. On the other hand, high-elevation East Andean ecosystems (>4400 m) in N Argentina and S Bolivia showed decreasing plant productivity over recent decades (radial growth of *Polylepis tarapacana*), attributed to increased aridity (Carilla et al. 2013).

Upward range expansions of species in the Andes under climate warming are predicted (Anderson et al. 2011; Larsen et al. 2011; Ramirez-Villegas et al. 2014), however, only a few observational studies documenting range shifts are available. Nevertheless, the results show more or less consistent patterns of upward species migrations and thermophilization effects throughout elevational gradients, even though wetter biomes and dry biomes may show heterogeneous responses to climate change (Tovar et al. 2013a; Cuesta et al. 2019). In the tropical Andes, Morueta-Holme et al. (2015) revisited the Chimborazo volcano in Ecuador 210 years after an expedition by Alexander von Humboldt and found the limit of plant growth having been strongly pushed upslope (Fig. 1.41). Here, distinct upward shifts in the distribution of vegetation zones are associated with increases in maximum elevation limits of individual plant taxa of >500 m on average. Duque et al. (2015) detected thermophilization effects in N Andean montane forests and adjacent lowlands in NW Colombia, reflected in directionally changing tree communities through time to include relatively more thermophilic species, with compositional shifts occurring primarily via range retractions (high tree mortality at lower elevations). Repeated censuses of forest inventory plots spanning an elevational gradient from 950 to 3400 m in SE Peru showed that most tropical Andean tree genera shifted their mean distributions upslope over the study period (2003/04–2007/08), while the observed mean rate of change was less than predicted from the temperature increases for the region, suggesting a limited ability to respond to increased temperatures and an increased extinction risks with further climate change (Feeley et al. 2011). Widespread thermophilization patterns in Andean forests were confirmed in a recent study based on almost 200 forest plots between 360 and 3360 m spread throughout the tropical and subtropical Andes (Fadrique et al. 2018). The results showed directional shifts in species composition towards having greater relative abundances of species from lower, warmer elevations, while the rates of thermophilization were heterogeneous throughout the elevation gradient, with negative or non-significant rates at highest (treeline) and mid-elevations (cloud base at the transition from montane to cloud forests). A repeated resurvey of permanent plots on four high Andean summits (4040–4740 m) in NW Argentina revealed high rates of plant community turnover and generally decreasing, but temporally fluctuating trends of plant cover, species richness, and diversity, related to the ENSO-influenced short-term temperature and precipitation variability (Carilla et al. 2018). Analysing chronosequences (38 years) in recently deglaciated terrain at high elevations (4700–4900 m) in the Central Andes, Zimmer et al. (2018) observed an overall increase in species richness, abundance, and plant cover and showed that colonization lags behind the velocity of warming and associated glacier retreat, and leads to no-analogue plant communities. As elsewhere, upslope range shifts have also been assessed for diverse groups of animals in the Andes (e.g. Moret et al. 2016; Seimon et al. 2017).

Climate warming-induced treeline dynamics is primarily reflected in tree growth (Lavergne et al. 2015) and increased recruitment above

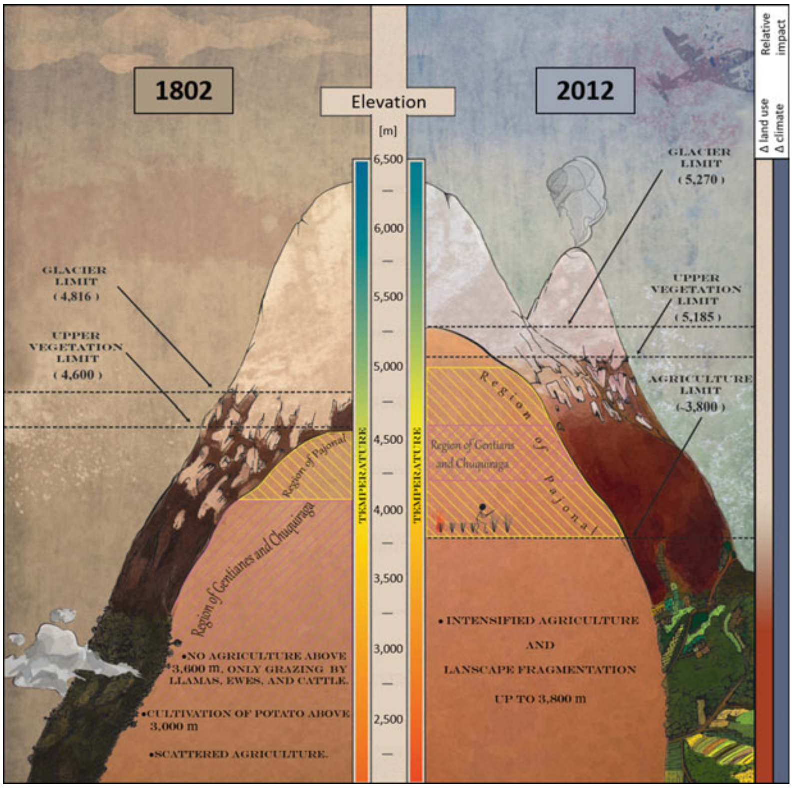


Fig. 1.41 An update of Humboldt's classic study of 1802, showing major changes in overall vegetation limit, average glacier limit, and shifts in topmost vegetation regions on Chimborazo from 1802 to 2012; the major drivers of change, climate, and land use change are represented by the bars to the right: a constant impact of climate change—in particular, increased temperature—the stronger relative impact of land use at the lower sites, mainly through intensified agriculture, and the effect of grass harvesting and local burning. (Modified from Morueta-Holme et al. 2015)

treeline in some places, but not (yet) in distinct treeline shifts. Based on a 42-year span of aerial photographs and high resolution satellite imagery in the high Peruvian Andes, Lutz et al. (2013) found only minor treeline shifts, with migration rates in protected areas being only 2.3% of the rates needed to stay in equilibrium with projected climate by 2100. In the semiarid Peruvian Andes and also in the case of cloud forests in the tropical Andes, initially stationary treelines suggest that other factors (topographic controls, high temperature variation, extreme cold events, water stress, high levels of solar radiation, low seed dispersal, competition with grasses, human impact) override the influence of increasing mean temperatures and may prevent cloud forest tree species from shifting their leading range edges upslope in response to climate warming (Bader et al. 2007; Rehm and Feeley 2015, 2016; Toivonen et al. 2018). Nevertheless, the results of Kintz et al. (2006) and Young et al. (2017) provide landscape-scale evidence of woody plant encroachment, upward treeline shifts, increasing shrubland areas, and increases in the number, size, and connectivity of forest patches at anthropogenic treelines in the Peruvian Andes. At *Nothofagus pumilio* treelines in Patagonia, Fajardo and McIntire (2012) found treelines moving uphill in abrupt pulses until at least 40–70 years ago, but declining tree growth in recent decades. The complexity of treeline dynamics in northern Patagonia was already highlighted by Daniels and Veblen (2004), who stressed the importance of moisture availability for seedling establishment of *Nothofagus pumilio*, and the small-scale differing und unstable relationships of radial growth and seedling demography with climate and ENSO over the late twentieth century (see also Srur et al. 2016). In southern Patagonia, Aravena et al. (2002) found positive correlations between *Nothofagus pumilio* tree growth and temperature at treelines, but a strong influence of local site factors. Srur et al. (2018) corroborated the sensitivity of abrupt *Nothofagus pumilio* treelines to changes in climate variations in the southern Patagonian Andes and found the rate of seedling establishment to be strongly modulated by the interaction between temperature increase and variations in precipitation.

As elsewhere, few non-native plant species have established in higher elevation habitats of the Andes. Alien species are largely restricted to disturbed sites, yet even protected mountain areas have been invaded (Speziale and Ezcurra 2011; Barros and Pickering 2014). Potential impacts of introduced species, e.g. competition for pollination, vary with their density (Muñoz and Cavieres 2008). Currently, the invasive nature of the common gorse (*Ulex europaeus*) causes serious problems in Colombian high Andean forests and paramos. The dense, compact, and homogeneous colonies of this invasive species impoverish or even eliminate native plant communities (Osorio-Castiblanco et al. 2020).

Africa

The increased warming trend across the African continent implies substantial impacts on ecosystems and has triggered similar biotic responses in mountains and highlands as reviewed above for other continents. Remote sensing studies in the Atlas Mountains suggest slightly positive land productivity trends and increases in montane forest cover and density (Del Barrio et al. 2016; Barakat et al. 2018), however, productivity and phenology are strongly controlled by precipitation variability (Otto et al. 2016; Missaoui et al. 2020), and effects of land use changes are pervasive (Mohajane et al. 2018). Positive correlations of radial growth of main tree species and interannual NDVI values in the Ethiopian Highlands suggest that precipitation variability controls landscape-level patterns of vegetation productivity (Siyum et al. 2018). However, increased pressure of human activities often overrides the effects of climatic variables. In the NW Ethiopian Highlands, for instance, monitoring of long-term NDVI changes (2000–2014) revealed a decline in vegetation productivity despite a significant positive trend of annual precipitation (Zewdie et al. 2017). The pattern of positive correlations between rainfall and NDVI and negative correlations between temperature and NDVI is widespread, while the start of the growing season in the highland ecoregions has advanced and the length has extended over recent decades (Workie and Debella 2018; Liou and Mulualem 2019). Significant NDVI declines in dry highland ecoregions suggest an increased risk of land degradation, to be attributed to interacting climate change and land use effects (Gebru et al. 2020). Patterns of vegetation productivity decline are reported for large tracts of land in eastern Africa (Landmann and Dubovyk 2014; Kalisa et al. 2019), largely explained by temperature-induced moisture stress (Krishnaswamy et al. 2014). This does not apply for most of the upper mountain regions of Mt. Kilimanjaro which have undergone a long-term (1982–2011) increase in vegetative signal (‘greening up’), to be mainly attributed to vegetation recovery after disastrous fires during the outgoing twentieth century, while the seasonal vegetation activity strongly responds to ENSO and IOD (Indian Ocean Dipole) teleconnections (Torbick et al. 2009; Detsch et al. 2016). Positive trends of recent NDVI values (2002–2017) were also assessed in the Drakensberg Mountains of South Africa (Mukwada and Manatsa 2018).

Very few observational studies on warming-induced changes in plant species distribution patterns and range shifts are available for African mountains and highlands. Modelling studies in the Atlas Mountains suggest that forest species such as *Cedrus atlantica* and *Quercus suber* will disappear from many localities and shift their distribution ranges, which become more contracted and fragmented, to higher elevations (Vessella et al. 2017; Bouahmed et al. 2019). In Algerian mountain forests, fire is considered the most important driver of forest degradation, with fire occurrence being linked to increasing aridity (Djema and Messaoudene 2009). In tropical African highlands, range shifts are mainly driven by anthropogenic pressure and fire as well (Wesche et al. 2000; Wesche 2002), and it is just as difficult to disentangle the role of climate change from the impacts of other drivers. Jacob et al. (2015a) pointed out for treeline environments in tropical African mountain ranges that treeline dynamics cannot be used as a proxy of climate change since treelines are strongly disturbed and have lowered due to high human and livestock pressure. In case studies in the northern Ethiopian highlands and in the Simien Mountains, Jacob et al. (2015b, 2017) provided evidence that treelines tend to shift upslope once anthropogenic pressure is decreasing, suggesting that the strong impact of land use outweighs climate change effects. Notwithstanding, a shift of 150 m of an almost inaccessible *Erica arborea* treeline in the Simien Mountains between 1905 and 2004 indicates involvement of rising temperatures (Jacob et al. 2017). Predicting advances of tropical treelines is, however, a difficult task given the multi-faceted constraints on tree regeneration above the uppermost forest stands (Wesche et al. 2008a).

Nevertheless, climate change and the interaction between climate drivers and land use change have additional effects, causing far-reaching alterations in Africa's mountain ecosystems (Niang et al. 2014). Future suitable habitats of *Juniperus procera*, the endangered and most preferred tree in the northern Ethiopian Highlands, are predicted to shrink by 80–90% (RCP 2.6 and 8.5) by the mid-century (Abrha et al. 2018). Growth patterns of *Juniperus procera* are strongly related to the amount of precipitation, suggesting high sensitivity to future drought periods (Couralet et al. 2007). Studies on *Erica arborea* tree-rings in North Ethiopia showed that tree growth is significantly and positively correlated with minimum temperature in the growing season, but negatively with minimum temperatures in the rainy season in spring (Jacob et al. 2020). In the southern highlands, upward range shifts will most likely create strong potential risks in terms of lowland attrition and range-shift gaps and lead to decreasing population sizes and a higher extinction risk (Kreyling et al. 2010; Kidane et al. 2019). Mekasha et al. (2013) showed that projected warming could significantly affect grassland herbaceous plant communities and that successful migrations of species are essential to mitigate range contraction and habitat losses with range-shift gaps. This also applies to diverse groups of animal species in African highlands (e.g. Raxworthy et al. 2008). Specialized high-alpine giant rosette plants are likely to face very high risk of extinction following climate warming (Chala et al. 2016).

Recurrent fires with climate change-induced higher frequency and intensity have resulted in substantial shrinkage of upper montane forests on Mt. Kilimanjaro, downward shift of the treeline, and in a biotic homogenization between the subalpine and alpine belts (Hemp 2005a, 2009). Increasing isolation of East African mountain ecosystems due to anthropogenic impact increases the threats to diversity and endemism under climate change (Hemp and Hemp 2018). Patterns in plant–pollinator specialization along elevational gradients on Mt. Kilimanjaro suggest that rising temperatures may destabilize pollination networks (Classen et al. 2020). Changes in East African highland ecosystems also include upslope range shifts of malaria vector species. Warmer temperatures at higher elevations facilitate range expansions and the creation of suitable vector habitats in the highlands (Ermert et al. 2012; Kulkarni et al. 2016). Regarding South Africa and Lesotho's mountainous regions of high biodiversity, substantial contractions in species' ranges towards higher elevations are predicted, decreasing the potential regions of occurrence of montane species (Bentley et al. 2019).

Invasive alien species in African highlands sometimes generate conflicts of interest between local communities and governments. On the one hand, they may provide benefits to local people as in the case of Mimosa (*Acacia dealbata*) in the Highlands of Madagascar or Mesquito (*Prosopis juliflora*) in East Africa (Kull et al. 2007; Mwangi and Swallow 2008). On the other, they adversely affect biodiversity and ecosystem services and their control incurs enormous costs across Africa each year (Boy and Witt 2013). At higher elevations, non-native plant species decrease in number and are largely confined to anthropogenic vegetation along roadsides or climbing routes, as exemplified by *Poa annua* on Mt. Kilimanjaro (Hemp 2008).

1.3 Effects of Land Use Changes in Major Mountain Systems of the World

1.3.1 General Overview

Humans have influenced and reshaped much of the world's mountain environments for millennia. In particular, highlands in Africa, Asia and Europe have been subjected to long-lasting land use and anthropogenic landscape transformation (Walsh and Giguet-Covex 2020). For instance, the onset of pastoralism in the Tibetan highlands dates back at least to 8000–9000 years BP (Miehe et al. 2009a, b, 2014, 2019). In many Old World mountain systems, the foundation of permanent settlements and the development of associated land use systems date back at least to the mid-Holocene. In adaptation to the challenges and constraints of harsh high mountain environments, mountain dwellers have developed over many generations sophisticated, complex resource utilization strategies for their sustenance, including a wide spectrum of farming and pastoral practices. Initially, mountain nomadism evolved as a strategy to sustain mountain-related livelihoods, often complementing or replacing subsistence hunting and gathering. It is characterized by animal husbandry as the predominant base for economic and labour activities of mobile communities conducting large-scale seasonal migrations between lowlands and highlands. After the establishment of permanent settlements and village lands, the combination of crop-farming and livestock-keeping evolved as the dominant basis of high mountain agriculture. Pastoral practices in alpine life zones have been increasingly integrated into more complex land use systems including *Alpwirtschaft* (combined or mixed mountain agriculture) and transhumance. However, nomadic pastoralism is still practised in Old World mountain regions, for instance in North and East Africa, Siberia and Mongolia, in the Altai, Tien Shan, Pamir, in Tibet, the HKH region, in the Zagros, and in parts of North and South Europe (Rhoades and Thompson 1975; Grötzbach 1980; Ehlers and Kreutzmann 2000; Kreutzmann 2012; Cunha and Price 2013; Price 2015).

In mountain regions already settled in prehistoric times, combined mountain agriculture has become the most widespread form of traditional land use. The combination of crop cultivation and livestock-keeping reflected the need to incorporate essential natural environmental resources of various altitudinal zones (forests, pastures) and different seasons into the land use system. Developing sophisticated practices of combined mountain agriculture involved interferences in mountain forests which have been increasingly converted to croplands. It also involved encroachments on alpine treelines which have been shifted downslope, often by several hundreds of metres, in order to enlarge alpine grazing lands. However, as long as mountain regions had been sparsely settled, overall impacts remained limited for many generations, and remote mountain ranges probably relatively undisturbed. In previous centuries, mountains provided a degree of isolation from the outside world for their permanent inhabitants and were often characterized by distinct inaccessibility resulting in more or less independent subsistence economies with limited trade and exchange relations with the plains or other mountain regions (Schickhoff 2011).

In some Old World mountain regions, far-reaching transformations of mountain environments are associated with the colonial history. Unlike Europe, where the growing demand for cultivable and pasture land as well as for timber and firewood led to an extensive clearing of mountain forests since the Middle Ages or even much earlier (e.g. in Mediterranean mountains), a significant number of Asian mountains experienced a considerable increase in mountain populations and the concurrent intensification of land use in the course of the past two centuries, encompassing the arrival of colonialism in mountain regions. Nevertheless, cultural landscapes associated with traditional land management also evolved in mountains of Asia over long time periods. In many mountain ranges, however, significant intensifications of agricultural land use took place at a later stage. For instance, rapid landscape transformations in the Himalaya, i.e. large-scale deforestation and substantial changes in the distribution of forests and agricultural lands, occurred only after the British annexation of Himalayan regions in the first half of the nineteenth century. In many Asian mountain ranges, the nineteenth and the twentieth century was a crucial period in the course of cultural landscape evolution and saw a considerable intensification of land use at higher elevations (Schickhoff 1995, 2007, 2011).

During the twentieth century, mountain regions in the Global South were largely characterized by high population growth, poverty, lack of economic opportunities, increased land use pressure, and increased integration into the economy of the lowlands. The primary sector had still been growing in importance, and local mountain farmers were often forced to intensify land use in response to internal drivers, e.g. population growth, and effects of economic globalization, for instance the cultivation of cash crops. Alpine zones were subjected to increased grazing pressure, adversely affecting highland integrity and biodiversity. Heavy grazing implies potentially dramatic losses of biological richness, soil degradation and erosion, and reduced site productivity. Increasing livestock populations, the transformation of traditional pastoral production systems, and inappropriate management practices initiated a general downward spiral in the productivity of many alpine grazing lands and resulted in a loss of biodiversity as well as an increased marginalization of pastoral people (Miller 1997; Schickhoff 2011). At the same time, even the most distant and remote mountain regions were influenced by effects of globalization, and mountains in general have been affected by far-reaching socio-economic transformation processes, notably in the second half of the twentieth century.

In mountain regions of Europe, livelihood diversification has started to gain momentum in the nineteenth century. In the course of the twentieth century, these transformations have eventually led to the extensification of traditional land use and to land abandonment as well as to the concurrent exploitation of mountain environments for tourism, mining, power generation or industrial-scale farming in favourable areas. Traditional forms of agricultural use have been abandoned and mountain farmers were increasingly absorbed in the tourist economy, particularly in winter tourism. The substantial shift from the primary to the tertiary sector has significant environmental implications, e.g. the development of winter mass tourism has neglected many environmental issues. Traditional land use on a moderate level appears to be a key driver for sustaining high levels of biodiversity, both at the ecosystem and landscape scale. Both intensification and abandonment reduce plant species richness relative to traditional land use patterns (Schickhoff 2011). In mountain regions of the Global South, the replacement of farming and herding by the tourism industry as the new economic mainstay has not yet progressed so far as in the European Alps, but the tourism industry has greatly expanded, as evident, for instance, from the mountain tourism in the Nepal Himalaya.

Recently, globalization effects and socio-economic integration into the larger world enhanced modernization trends in mountain agriculture in the Global South. Mountain farmers seek to improve their livelihood by combining alternative farming systems (e.g. agroforestry, cash crops), non-agrarian income (e.g. tourism), and migrant labour remittances, while taking full advantage of the well-established access to lowland markets, provided by the tremendously reinforced road construction. Another intensifying trend is the migration of mountain people from remote locations to surrounding lowlands which could already be observed in the late twentieth century. Impoverished and marginalized mountain people, especially those which are young, energetic and economically active, are increasingly attracted by more diverse and favourable education, job and income opportunities in urban centres of the lowland. Highland-lowland migration, sometimes also stimulated by environmental or political crises (Hugo and Bardsley 2014), often alleviates the population pressure on the scant resource base and leads to a reduced land use intensity at higher elevations. Decreasing population numbers and reduced human pressure may allow ecosystem and biodiversity recovery, where alpine grazing lands had been degraded by previous overuse. It also facilitates the imposition of new forms of land tenure, for instance the establishment of national parks and other protected areas whose number has considerably increased in recent decades. While conservation of most terrestrial ecosystems is not trending towards sustainability, any progress in protecting biodiversity and ecosystems in mountain regions is a vital support for achieving the land degradation-related UN Sustainable Development Goals (UN 2020).

1.3.2 Regional Overview

Asia and Australasia

In the vast HKH region, pastoral strategies are still critically important for sustaining livelihoods of a large human population (Kreutzmann 2012; Dong SK et al. 2016). Livestock grazing in the framework of combined mountain agriculture or by mobile pastoral communities is the predominant land use strategy in the alpine life zone (Fig. 1.42). Alpine grasslands cover more than half of the total land area (including the Tibetan Plateau) and are currently expanding at the expense of snow/glacier cover (cf. Wu et al. 2013; Paudel et al. 2016), thus representing a substantial resource base for animal husbandry. However, as elsewhere, alpine pastoralism is highly susceptible to ongoing social, economic and cultural transformations, resulting in a significant decrease in the importance attached to highland livestock strategies and in a decline of grazing intensity. Labour outmigration is the most important driver of reduced alpine land use intensity. In Nepal, for instance, the migrant population is steadily increasing. Almost 500,000 workers left Nepal in 2014 to work in India, Malaysia, the Gulf countries and other destinations, and remittances have exponentially increased in recent years and already contribute more than 30% to the country's gross domestic product (Fig. 1.43) (Shrestha 2017; Siddiqui et al. 2019). Rural–urban migration within Nepal has also reached high levels and resulted in a largely uncontrolled urbanization process in Kathmandu, leading, inter alia, to severe environmental degradation (Schickhoff 2019). A general decline in pastoral lifestyle and in the number of pastoralists has been assessed for the eastern, central, and western HKH region (Afghanistan might still be an exception), where transformation processes, commercialization of pastoral lands, youth migration and labour shortage, inadequate policy support and institutional arrangements, the decline of trans-Himalayan exchanges (Fig. 1.44), and also the establishment of parks and protected areas aggravate maintaining accustomed pastoral strategies (Nüsser and Gerwin 2008; Bhasin 2011; Schmidt-Vogt and Miehe 2015; Gentle and Thwaites 2016). The livestock sector in the HKH region is characterized by a general decline in the cattle population, while land abandonment and the decrease of traditional agricultural practices due to labour shortage are apparently more pronounced at higher elevations (Chidi 2017; Wang et al. 2019). Whereas the decline in grazing intensity in the Himalaya mainly results from modified pastoral strategies adopted by pastoralists themselves (e.g. Bergmann et al. 2012), reduced high-elevation pasture utilization on the Tibetan Plateau as well as in high mountain ranges of E and S China is caused by external interventions, i.e. state programmes in order to transform the pastoral sector such as resettlement schemes and sedentarization measures aiming at modernization and at reducing grazing pressure and ecological degradation (Ptackova 2012; Hua et al. 2013; Kreutzmann 2013; Qiu 2016).

Over the past few decades, overgrazing by livestock was a major stressor on alpine ecosystems, livestock-environmental interactions had resulted in degradation of alpine grazing lands across the entire HKH region, in particular in drier parts and on the Tibetan Plateau (Harris 2010; Paudel and Andersen 2010; Wu et al. 2013; Baranova et al. 2016; Miehe et al. 2019; Niu et al. 2019; Breckle and Rafiqpoor 2020). In

Fig. 1.42 Pastoralism is declining, but still the predominant land use strategy in the alpine High Asia, exemplified by Ladakhi pastoralists with their goats and sheep at Lake Tso Moriri (4522 m). (*Photo* © Udo Schickhoff, September 23, 2018)

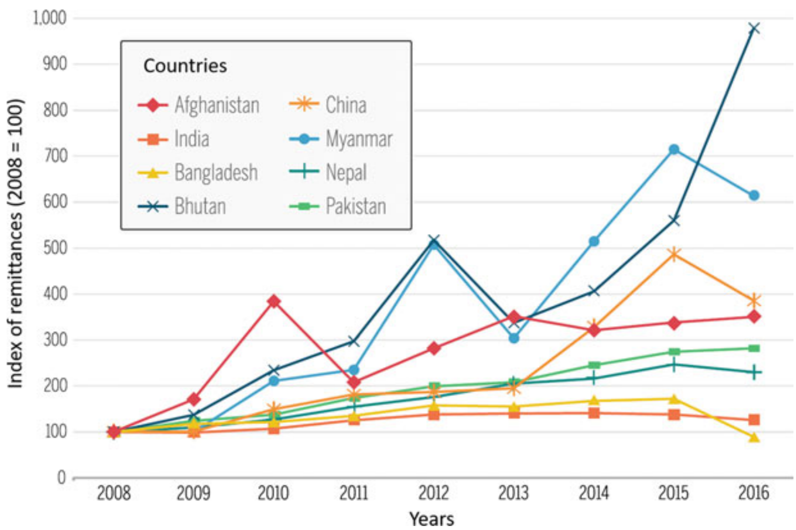


Fig. 1.43 Changes in the index of international remittances received by HKH countries in 2016 (2008 = 100). (Modified from Siddiqui et al. 2019)

quite a few locations, however, local herders have developed effective indigenous rangeland management systems using effective grazing and conservation practices (Dong SK et al. 2007, 2016; Aryal et al. 2014). The current extensification of alpine pastoralism (e.g. Dangwal 2009a) gives grounds for cautious optimism that pasturelands will no longer be grazed beyond their carrying capacity, that formerly degraded rangelands will recover, and livestock grazing will sustain biodiversity and ecosystem services (Cai et al. 2015).

In addition to transformations of high-elevation grasslands, significant land use/land cover changes in the HKH region over recent decades include the conversion of forest to other land uses, mainly farmland, at lower elevations (Wang et al. 2019). However, the (pre)historical dimension of land use/land cover change and deforestation may not be disregarded. As indicated by palaeoecological studies, humans have changed forest environments and transformed forests into replacement communities at least since the mid-Holocene (Miehe et al. 2009a, b;

Fig. 1.44 Desertion of settlements and abandonment of terraced fields after the closure of the Trans-Himalayan trade as exemplified by repeat photographs (1956/2004) of the summer village Milam (3440 m) in Uttarakhand, Indian Himalaya, also symbolizing the recent decrease in land use intensity and the development of a new periphery (upper photo by Bhup Singh Negi; lower photo by Marcus Nüsser; photos courtesy of Marcus Nüsser). (Modified from Nüsser 2006)

Byers 2017), albeit with human interferences and forest clearings having commenced at considerably different times in various Himalayan valleys (Jacobsen and Schickhoff 1995; Beug and Miehe 1999; Schlütz and Zech 2004). It needs to be highlighted that the basic patterns of the present-day cultural landscape in Himalayan valleys are not much different from those of the late nineteenth century (Schickhoff 1995, 2007, 2012). Even though the forests of the Himalaya were considered to be more or less untouched and inexhaustible in pre-colonial times, human impact must have transformed the landscape in many valleys for many centuries, in particular in fertile basins such as Kathmandu or Kashmir Valley which had been inhabited in early times. For instance, the difference between the current upper limit of forests and the potential alpine treeline may be up to 500 m, on south-facing slopes even more, resulting from long-lasting human impact (Miehe 1997; Schickhoff 2005; Miehe et al. 2015; Schickhoff et al. 2015). The expansion of agriculture and trade after the British occupation of Himalayan territories in the first half of the nineteenth century resulted in first significant reductions of forest cover in colonial times. Severe overexploitation of Himalayan forests occurred during the railway building era in the following decades which prompted the constitution of the Imperial Forest Department by the then British India government in 1864. Despite the introduction of 'scientific forestry', unsustainable use in large tracts of mountain forests continued, while the protective influence of silvicultural management was more or less confined to less extensive forest stands, demarcated as 'Reserved Forests' (Schickhoff 1995; Dangwal 2009b). Another phase of massive deforestation arose during World War II and the subsequent struggle for independence.

The first decades of the post-colonial era were characterized by the extensive failure of centralized forest management systems, ultimately resulting in a paradigm shift in forestry (Schickhoff 2014). Continued depletion and degradation of forest resources constituted a threat to rural livelihoods and environmental sustainability and gave rise to the generation of environmental initiatives such as the 'Chipko' movement and to revised forest policies in the 1970s and 1980s, characterized by the introduction of participatory forest management approaches. During this phase, disaster scenarios were fabricated, based on simplified relationships between population growth, deforestation, overgrazing, soil erosion, and floods in the lowland, assuming that the Himalaya was approaching a complete loss of forest cover and catastrophic levels of environmental degradation. Ives and Messerli (1989) clarified that much of this 'Theory of Himalayan Environmental Degradation' is nothing but scaremongering, and encouraged subsequent studies that clearly disproved the theoretical construct (see Ives 2004,

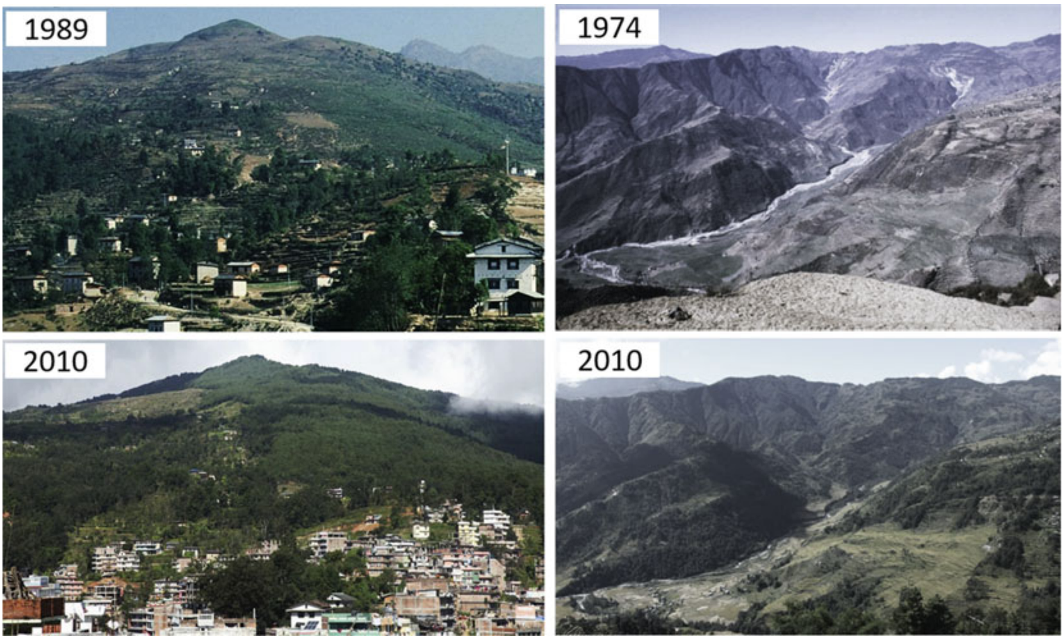


Fig. 1.45 Repeat photographs of Charikot (1989/2010) and Serabesi (1974/2010), Bhimeshwor cluster, indicating the success of community-based forest management in forest restoration. (Nepal Swiss Community Forestry Project; modified from Pokharel et al. 2011; Niraula et al. 2013)

2013). First positive outcomes of participatory and community-based management practices were reflected in an increase of forest areas in c. 25% of all Himalayan districts between 1960 and 1990, while c. 35% reported forest loss (Zurick and Pacheco 2006; Schickhoff 2007). A substantial loss of forest cover was observed in the Karakoram and in the outer Himalayan ranges (Schickhoff 2002, 2006, 2009). In recent decades, decentralized management systems following the 'Community Forestry' approach have been successfully established across the HKH region and have gained relevance for the cultural landscape, in particular in Nepal (Figs. 1.45, 1.46) (Schickhoff 2014). To date, more than 18,500 community forest user groups are managing almost 2 million ha of Nepal's forest, corresponding to c. 30% of the total forest cover (Xu et al. 2019). Remote sensing data show that only 12% of Nepal's districts experienced a loss of forest cover between 1990 and 2013, while 68% showed an increase (Figs. 1.47, 1.48) (Nebelung 2016). Among the national-level forest assessments in Nepal since the 1970s, the latest forest resource assessment 2010–2014 detected the largest forest cover (40%) in Nepal (Fig. 1.49) (DFRS 2015).

In spite of multiple challenges and some limitations and shortcomings such as inequitable benefit-sharing and the exclusion of poor and marginalized groups, the adoption of community-based management approaches has resulted in positive ecological, economic, and social impacts, and most user groups succeeded in regenerating areas of degraded forests and reversing the trend towards forest degradation and deforestation (Gurung et al. 2013; Pathak et al. 2017; Luintel et al. 2018). This also applies to mountain forests in Bhutan, Tibet, India, and to some extent in Myanmar, while Pakistan and Afghanistan are still concerned to achieving a visible impact from community forestry (Xu et al. 2019). On the other hand, the success of community-based forest management should not obscure the fact that forest degradation and deforestation is still an issue at various locations across the HKH (Nüsser 2000; Pandit et al. 2007, 2014; Qasim et al. 2013; Schmidt-Vogt and

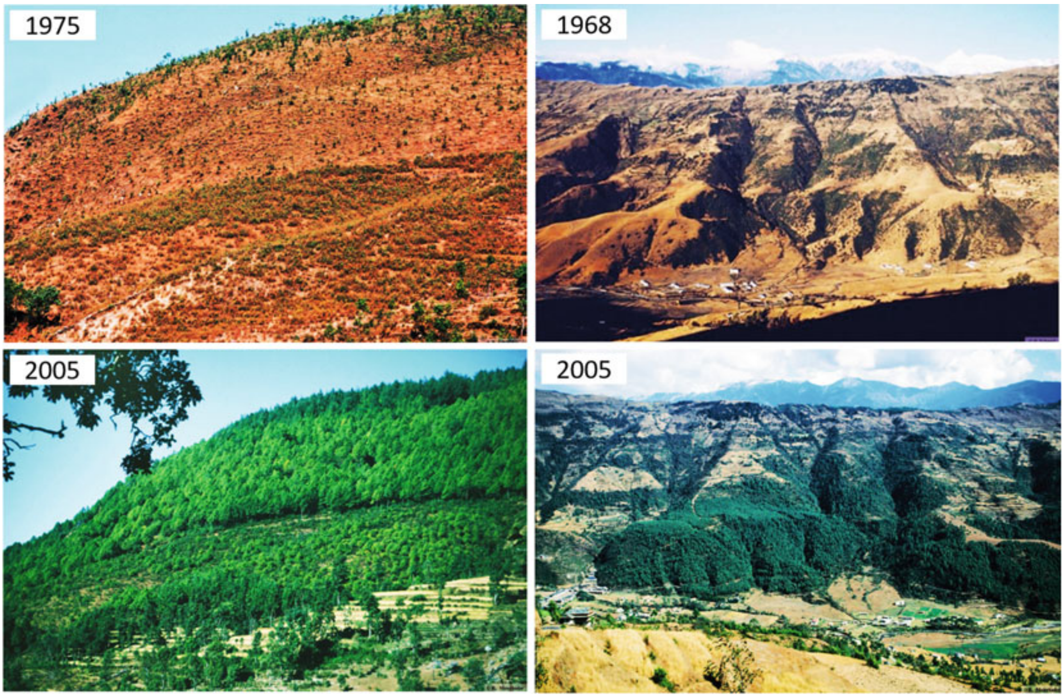


Fig. 1.46 Repeat photographs of Dandapakhar, Sindhupalchowk district (1975/2005) and Jiri, Dolakha district (1968/2005), indicating the increase in forest cover caused by the community forestry approach (Nepal Swiss Community Forestry Project. (Modified from Pokharel et al. 2011)

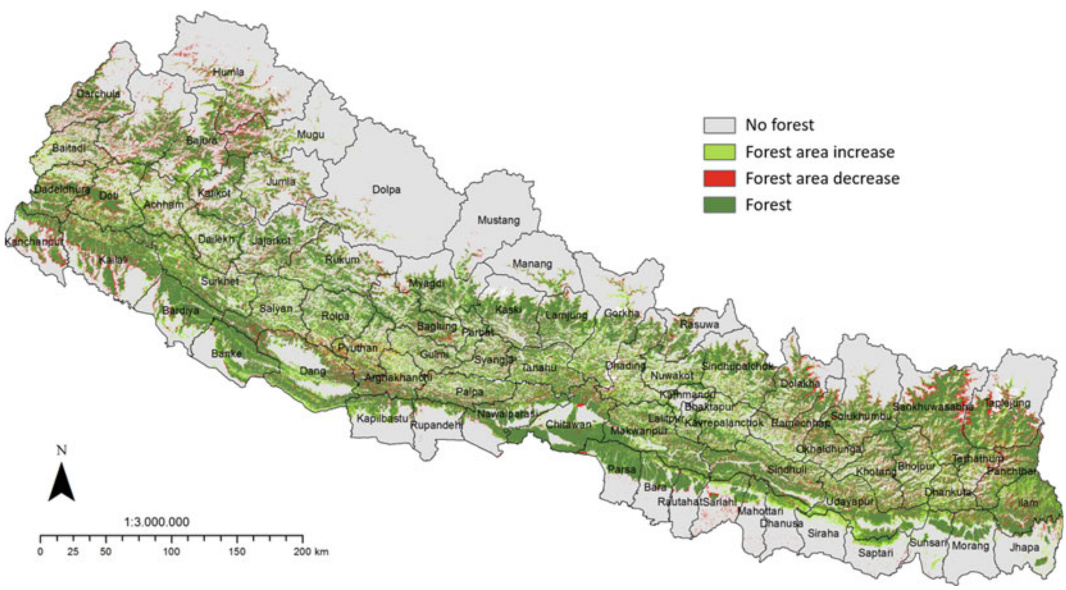


Fig. 1.47 Forest cover change in Nepal 1990–2013. (Modified from Nebelung 2016)

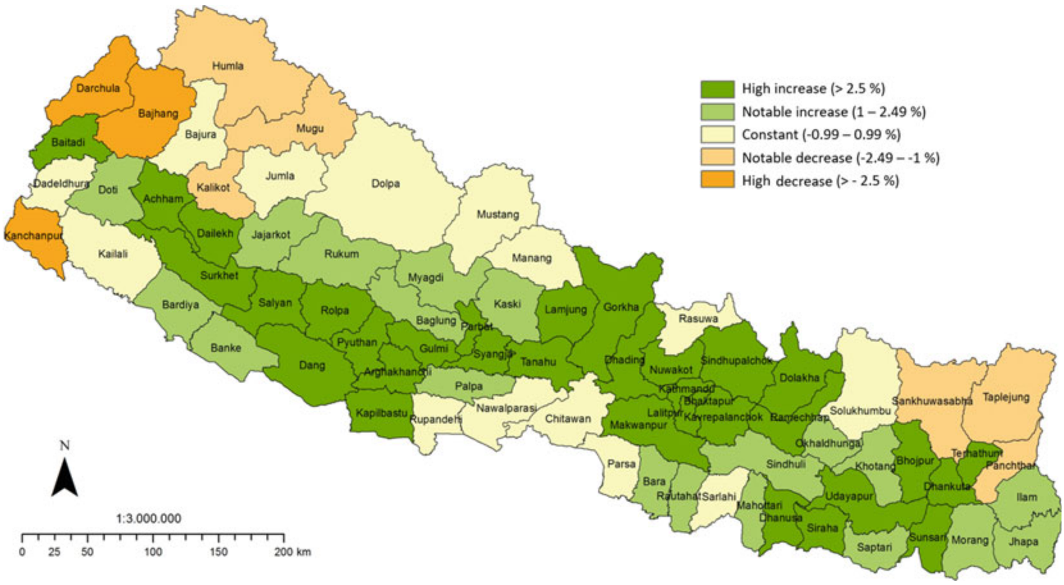
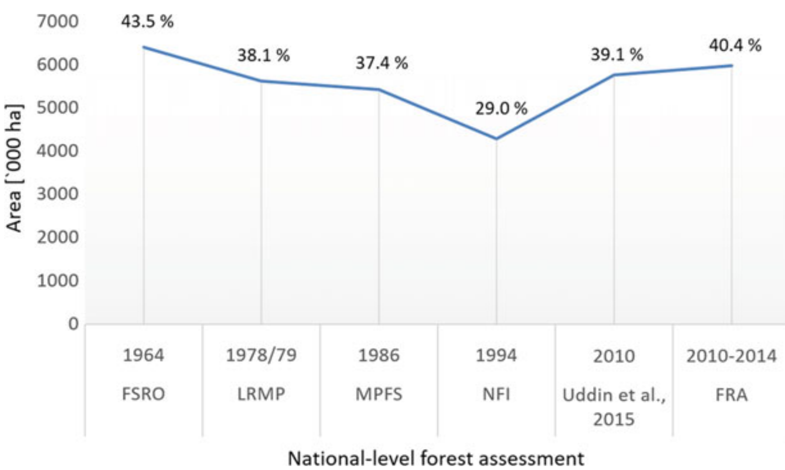


Fig. 1.48 District-wise percentage change of forest cover in Nepal 1990–2013. (Modified from Nebelung 2016)

Fig. 1.49 Extent of forest cover in Nepal 1964–2010/14 according to national-level forest assessments (after data in Paudel et al. 2016; DFRS 2015)



Miehe 2015; Uddin et al. 2015; Garrard et al. 2016; Qamer et al. 2016; Nüsser and Schmidt 2017; Kanade and John 2018; Reddy et al. 2019). It also needs to be highlighted that forest area statistics have little meaning for the qualitative condition of mountain forests. The loss of structural complexity, shifts in species composition, decreasing species richness, erosion of humus horizons and adversely affected ecosystem functions are widespread side effects of forest utilization in recent decades (Schickhoff 2002, 2009, 2012).

As tourism is one of the fastest growing sectors in the world, it has become a significant contributor to the national economy in developing mountain economies. In High Asia, Nepal stands out as a particularly popular destination for international tourism in recent decades, receiving more than one million visitors in 2018. The rapid development of tourism has transformed Nepal's economy, society and environment. While the positive impacts of tourism on local economic growth are widely acknowledged, social and cultural impacts of tourism are viewed critically due to observed changes in local norms, values and behaviour (Shakya 2016). It was feared that the environmental carrying capacity of tourism in the Nepal Himalaya could be exceeded, e.g. the growing demand for firewood and timber was intermittently an object of concern (Byers 2005). In the meantime, tourism is better integrated with environmental conservation, not least through the involvement of locally based institutions and enhanced local participation (Anup et al. 2015). A major development impulse for remote mountain regions is triggered by the expanding rural road network that facilitates the adoption of mobility as an adaptive livelihood strategy. Beazley and Lassoie (2017) recently examined the wide variety of influences on environmental, socio-economic, and sociocultural spheres in the Nepal Himalaya. Human and environmental systems in formerly secluded mountain regions have been tremendously impacted by road construction, as evident from the case of the Karakoram Highway (Kreutzmann 1991; Stellrecht and Winiger 1997; Stellrecht 1998; Schickhoff 2009).

Pastoralism has clearly predominated land use systems at higher elevations in the Pamir, Tien Shan, and Altai, playing a crucial role in Central Asian economies, societies, and cultures since time immemorial. In the former Soviet Central Asian Republics, pastoral traditions and strategies have undergone tremendous changes in the course of the twentieth century, to be attributed to strong external interventions. The first decades of the Soviet era were characterized by forced sedentarization and collectivization campaigns, resulting in a considerable intensification of pastoral land use and its integration into socialist agro-industrial production (Dörre and Borchardt 2012). The pastoral strategy of Soviet times was based on pastoral brigades and herding collectives in the framework of *kolkhozes* (collective farms) and *sovkhozes* (state farms) as well as on permanent high-elevation grazing with short-distance migrations only. This 'detached mountain pastoralism' (Kreutzmann 2011) entailed overuse of grazing resources and related degradation problems that were addressed with pasture irrigation, fertilization, and rotational grazing. The disintegration of the Soviet Union in 1991 and the subsequent political and economic transformation required once again fundamental adaptations of pastoral strategies, now based on private ownership of livestock, subsistence farming, and low state interference in grazing activities. Deindustrialization, the initial decline of national economies, and the disappearance of social securities have led, inter alia, to an increased dependency on grazing land resources. After three decades of post-Soviet transformation, an increased scope and diversity of pasture-related socio-ecological challenges can be observed including conflicts about access to pasture resources, utilization rivalries, insufficient management practices, and degradation processes (Borchardt et al. 2011, 2013; Dörre 2012; Vanselow et al. 2012a), in spite of efforts to decentralize governance and to establish community-based pasture management (Shigaeva et al. 2016). The spatial pattern of pasture degradation has changed in recent decades: Grazing intensity on remote summer pastures at higher elevations has declined due to abandoned seasonal livestock migration (Fig. 1.50), while winter pastures, located close to settlements, have been subjected to more intense grazing pressure with adverse effects on vegetation, plant functional traits, and soils such as lower species richness and diversity, lower biomass, decreased plant height and specific leaf area, lower organic matter content, and higher soil pH values (Akhmadov et al. 2006; Vanselow et al. 2012b; Hoppe et al. 2016a, b, 2018; Mirzabaev et al. 2016; cf. also Liu and Watanabe 2016).

The relative proportion of land covered by mountain forests in Central Asia is rather low. Nevertheless, the natural resources of the forested zones have been an essential component of local land use systems since time immemorial. Forests have been subjected to grazing use and to intensive use of timber and non-timber products (timber, firewood, nuts, fruits, herbs, hay,

Fig. 1.50 The decline in seasonal livestock migration has reduced grazing intensity on remote summer pastures as in Suusamyr valley, Kyrgyzstan, Tien Shan. (*Photo* © Udo Schickhoff, September 26, 2004)

mushrooms, etc.) ever since, resulting in fragmentation, degradation, and transformation. For instance, the extensive walnut forests in the western Tien Shan (Kyrgyzstan) are most likely of anthropogenic origin. Most of these forests replaced mixed juniper-deciduous forests and were established 1,000–1,500 years ago, when fire was used for agricultural purposes and planting of walnut trees was promoted (Beer et al. 2008). The walnut-fruit forests are of high economic value and of essential importance for sustaining the livelihoods of a large population living in the forest area, however, they are characterized by impoverished stand structures, regressive successions, and insufficient regeneration (Borchardt et al. 2010). The deteriorated state of the mountain forests in Central Asia results from the legacy of silviculture practised in the Soviet period and intensified, sometimes unregulated forest utilization in the post-Soviet phase when economic recession increased the pressure on forest resources. Centralized and formal forest management had started with the Russian occupation in the nineteenth century and was strengthened after establishing the planned economy of the USSR. The recent transformation process initiated by the collapse of the Soviet Union and globalization effects have resulted in intensified exploitation and degradation of mountain forests, facilitated by the local population's insecure economic situation, the erosion of managing institutions and institutional weakness with unsustainable and inconsistent management practices, and the appearance of new actors (Schmidt 2005, 2012). Accordingly, the area covered by walnut forests has decreased considerably in recent years (Hardy et al. 2018), adding to the general negative trend of forest cover in the Asian Dryland Belt (Chen et al. 2020).

The history of mobile pastoralists' land use strategies and livelihoods in Mongolian mountain ranges in the twentieth century has many similarities to the former Soviet Central Asian Republics. The system of traditional land use has undergone significant and to some extent dramatic changes, characterized by sedentarization and collectivization during the period of the People's Republic, and by the revival of pastoral nomadism in the early 1990s after the transition to a democracy and market economy. The return of Mongolian nomadism resulted in rapidly growing livestock populations, shifts in herd composition, and widespread degradation of rangelands, also at higher elevations (Fernandez-Gimenez 2002; Janzen 2005; Schickhoff et al.

2007; Zemmrich et al. 2010; Hilker et al. 2014). Reduced livestock mobility, a lack of institutions governing pasture use, and increased poverty among herders are among the challenges to manage rangeland sustainably. The ongoing establishment of community-based rangeland management—over 2000 formally organized herder groups formed since 1999—is a promising institutional innovation which should support implementing strategies towards sustainable pastoral land use (Fernandez-Gimenez et al. 2015). Uncontrolled grazing in mountain forests, fire, and logging are primary drivers of forest degradation and forest depletion and have resulted in substantial annual forest loss in the post-socialist era (Tsogtbaatar 2013). The major industrial sector in Mongolia is mining, accounting for a higher share of the GDP than nomadic animal husbandry. Exploitation of mineral resources has caused severe environmental problems in Mongolia's mountain ranges including devastated rivers and decreasing water resources (Suzuki 2013).

Land use patterns and livelihoods of pastoralists in Russian mountain ranges (Siberia, the Urals) have been affected by the implementation of post-socialist land policy in a similarly fundamental way, subjecting herders to socio-ecological challenges such as unequal allocation of grazing land and localized high grazing pressure (Intigrinova 2010; Istomin and Habeck 2016). In the Caucasus, post-socialist land reforms have reshaped land use patterns meanwhile to that extent that subalpine and alpine zones are currently characterized by outmigration, land abandonment, and increasing recreational activities (Belonovskaya et al. 2016; Gunya 2017). High mountain ranges of Iran have been subjected to intense grazing since ancient times, reflected in the dominance of thorn-cushion formations. Recently, alpine ecosystems are increasingly threatened by reinforced grazing impact, even in protected areas (Noroozi et al. 2008, 2020). Overgrazing has also caused severe pasture degradation in the Pontic Mountains (Curebal et al. 2015). Land use impacts on alpine life zones in New Guinea are considered to be relatively low, exceptions include mining impacts on Mt. Jaya and recently developing ecotourism on Mt. Wilhelm (Hope 2014). However, the mosaic of subalpine forests and grasslands and the fragmentation of the treeline in some mountain areas originated from forest clearings by fire over previous decades and centuries (Hope 2020). Increasingly adverse tourism impacts on the alpine environment have also been assessed on Mt. Kinabalu, Borneo (Latip et al. 2016).

Land use changes in New Zealand's mountain ranges are inextricably linked to the introduction of a large number of non-native species, to which unique island ecosystem biota are particularly vulnerable. New Zealand is one of the most invaded places in the world, many alien species are considered to be invasive pests. Polynesian settlement of New Zealand c. 800 years ago resulted in the clearance of vegetation and in the extinction of 27 bird species, including all moa genera (flightless birds), not least through the introduction of the Pacific rat (*Rattus exulans*) (Bellingham et al. 2010). But only after the late eighteenth century arrival of the Europeans reinforced exploitation of mountain environments (logging, grazing, mining, quarrying) commenced, resulting in large-scale deforestation and substantial landscape transformation (Pawson and Brooking 2013). The period of exploitative pastoralism in montane and alpine grasslands was associated with the depletion of palatable native grasses and herbs that was countered since the 1950s by widespread oversowing with introduced grasses and legumes, leading to the spread of pastoral weeds (Lord 2020). Recently, marginal pastoral high country has been reverted to shrubland and forest. However, indigenous forest, shrubland and grassland vegetation showed a declining trend between 1996/97 and 2012/13, with the exception of subalpine shrubland (Dymond et al. 2017). Numerous non-native plant and animal species have been introduced by the Europeans, some of them such as the brushtail possum (*Trichosurus vulpecula*) and the red deer (*Cervus elaphus*) constitute an important threat due to the damage caused in mountain forests by trampling and browsing (Allan and Lee 2006). As tourism is New Zealand's fastest growing industry, alpine areas are heavily used for sight-seeing, hiking and skiing, placing considerable pressure on higher elevations (Lord 2020). In the Australian Alps, Aboriginal peoples already burned vegetation, however, vegetation physiognomy has undergone more changes during the 200-plus years of Anglo-Australian settlement, inter alia, through the introduction of exotic grasses and weeds. Currently, recurrent disturbance by fire overrides other impacts regarding landscape-scale changes (Collins et al. 2019).

Europe

In many respects, the European Alps can be considered a role model for recent development processes in mountain regions worldwide (Perlik 2019). Over the past 150 years, the Alps have witnessed the process of a profound structural change from an agrarian society to a post-industrial service-based economy, associated with an advanced transformation from a rural to an urban society (Bätzing 2015). Accordingly, land use systems have been reshaped, with modified type and intensity of land use having far-reaching consequences for Alpine landscapes and ecosystems. After-effects of early land use are still visible in the modern Alpine landscape. Neolithic herdsmen already started to expand grazing lands by slash and burn practices in parts of the Alps about 7500 years BP (Conedera et al. 2017). Many centuries of forest clearing have lowered the alpine treeline by 300 m on average, in places by 600 m or more, a process which is of landscape relevance until today. Several waves of increase of the human population and human migration into the Alps, notably between 5,000 and 3,500 and between 1,200 and 700 years BP, entailed the foundation of permanent settlements at higher elevations, leading to widespread human impacts on mountain forests and to large-scale deforestation (Bebi et al. 2017). Complex livelihood systems evolved based on the combination of subsistence agriculture and animal husbandry (*Alpwirtschaft*) in order to make maximum use of resource extraction from multiple altitudinal belts. After the deforestation phase of the Middle Ages, intensive exploitation of mountain forests for energy (in particular for salt processing) and construction materials continued, only slowing down in the aftermath of the Black Death. Renewed population growth and increased demand of wood resources due to the beginning industrialization resulted in another phase of accelerated deforestation across most of the Alps in the late eighteenth and early nineteenth century (Bätzing 2015; Mathieu 2015). Over the centuries the traditional cultural landscape of the Alps had been created, considered to be of high aesthetic value, of vitally contributing to human well-being, and to be the basis for destination marketing of the tourism industry (Schirpke et al. 2019).

In the early nineteenth century, the Alps constituted a less developed region, with Alpine inhabitants facing relative poverty, malnutrition, starvation, and waves of out-migration. The introduction of the potato in the early 1800s and the building of roads and railways in the following decades allowed for some partial mitigation of poverty and hunger. At the same time, the beginning of the industrial revolution led to a successively reduced importance of farming, crafts and mining, prompting the commencement of tourism in the *Belle Epoque* towards the end of the nineteenth century. Livelihood diversification with the decline of traditional farming, the rise of industry and commercial agriculture, and increased economic activities related to tourism has pushed the fundamental structural change that is still unfolding today (Bätzing 2015). The transformation of landscape patterns resulting from the decline of the traditional cultural landscape became most notable after the Second World War, in particular with the initiation of mass tourism in the 1960s and the investment in large-scale winter sports and winter tourist facilities. The former agricultural society has transformed itself into a leisure society (Lichtenberger 1988), not least indicated by the fact that in many Alpine regions income from tourism has become more important than economic returns from farming. Following a stagnation phase (1985–2003), recent trends in Alpine tourism are characterized by the redevelopment of tourism centres and new major projects,

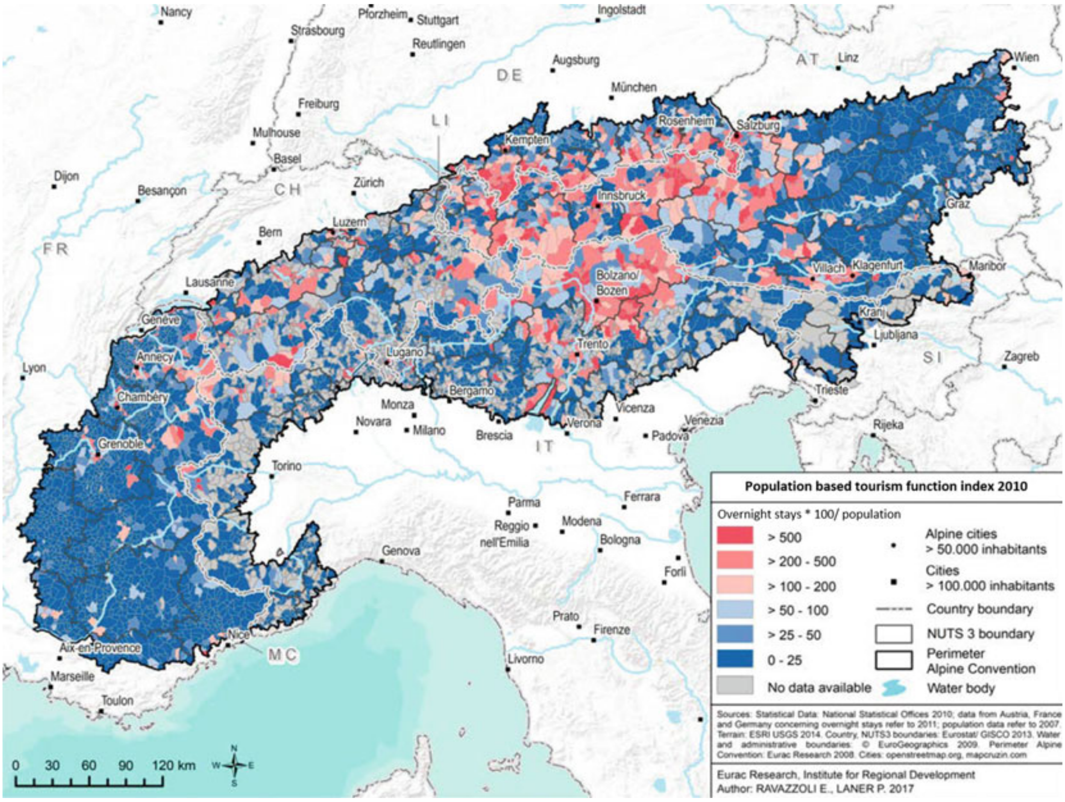


Fig. 1.51 Spatial pattern of tourism intensity in the European Alps, based on the ratio between overnight stays and population. (Modified from Elmi et al. 2018)

associated with a strong centralization in fewer tourism municipalities in more favoured areas with a higher number of touristic beds and overnight stays (Fig. 1.51) (Bätzing 2018). Nowadays, the tourism industry contributes significantly to the Alpine economy, even though the number of jobs directly or indirectly linked to tourism is less than 20%.

Apart from the expansion of touristic infrastructure, the abandonment of agriculturally used areas and the subsequent regeneration of forests has been the essential process of land cover change across the European Alps over the past 150 years. Agricultural land has almost halved between 1850 and 2005, while forest areas have increased by about half and settlement areas quadrupled (Egarter Vigl et al. 2016). In some places, the cessation of land use encompasses as much as 70% of previously used land areas (Tasser et al. 2005). Agriculture in less accessible and marginal areas, in particular on alpine pastures, has tended to become more extensive or has even been abandoned, whereas a trend towards intensifying production can be observed in easily accessible prime locations where much of arable land has been converted to grassland (Tasser et al. 2009; Zimmermann et al. 2010). The observed abandonment of farms is particularly striking in Italy and parts of France and Switzerland (Fig. 1.52). The total number of Alpine farms decreased from 570,000 in 1980 to 260,000 in 2010 (Elmi et al. 2018). The decreasing significance of the agricultural sector is also reflected in the low share of employees in agriculture which was as high as 75% in 1850, but accounts for only 2.5% of total employment in the western Italian Alps and for only 2.3% in the French Alps today (Permanent Secretariat of the Alpine Convention 2015). While mountain agriculture is generally becoming less and less

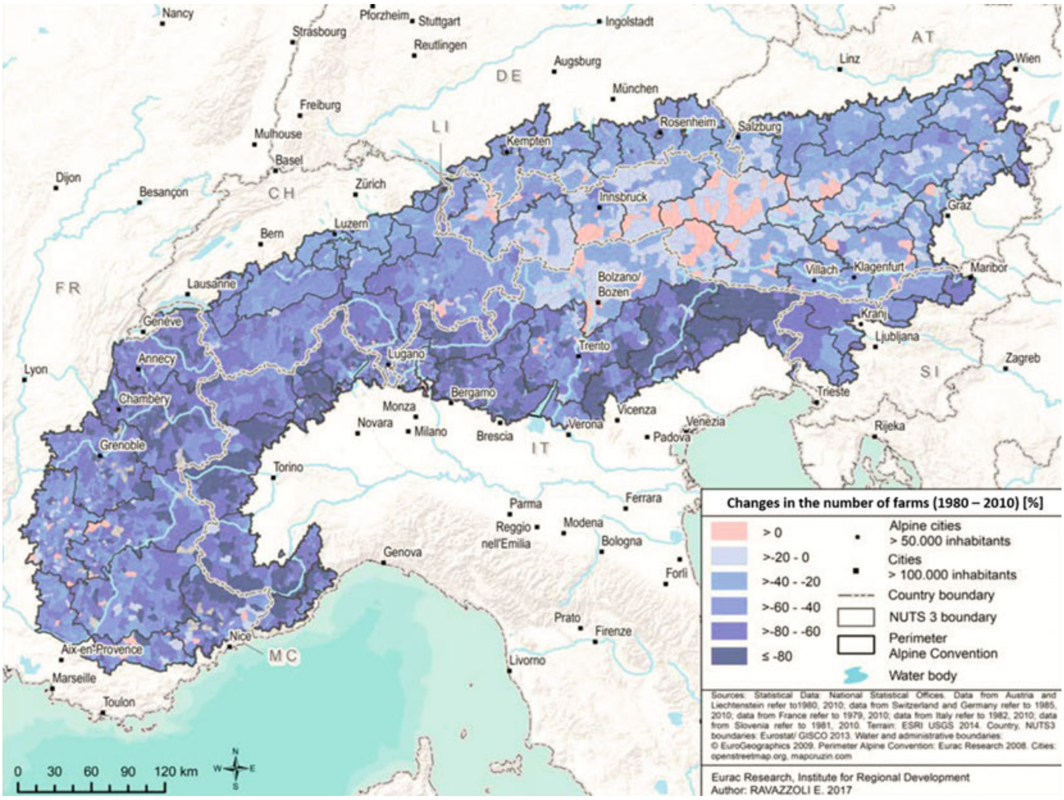


Fig. 1.52 Spatial pattern of the abandonment of farms in the European Alps. (Modified from Elmi et al. 2018)

competitive under economic globalization, it is still highly relevant for maintaining landscape patterns in the Alps. Agriculture still plays a larger role in the northern, German-speaking Alpine countries, facilitated by mountain farming subsidies and the practice of part-time farming (Borsdorf et al. 2015). Forest cover has increased across the entire Alps (Fig. 1.53), with average rates recently accelerating from +3.7% per decade since 1930 to 4.3% per decade since 1990 (Bebi et al. 2017). Secondary forests mainly established on former agricultural land by natural reforestation (Borsdorf and Bender 2007; Tasser et al. 2007). Free succession on abandoned areas inevitably leads to the establishment of new forest areas. Over the past decades, the most rapid increase in forest cover has been observed in the Italian Alps, in the southern Swiss Alps, and in the Austrian province of Salzburg (Bebi et al. 2017). The increase in forestland is a conspicuous effect at landscape scale (Fig. 1.54), associated with a trend towards more monotonous landscapes with reduced structural diversity.

Land abandonment as well as land use intensification results in changes in biodiversity, biogeochemical cycles, climatic and hydrological processes, and related feedback effects on, inter alia, erosion rates, magnitude of floods, snow gliding, and avalanches. Observed biodiversity changes in montane, subalpine and alpine grasslands of the Alps were found to be mainly driven by land management, suggesting that land use change rather than climate change appears to be the most prominent pressure acting on Alpine biodiversity (Vittoz et al. 2009; Dullinger et al. 2020). While land use had a facilitating impact on species and habitat diversity in previous centuries, the transition towards modern high intensity agriculture and the abandoning of land

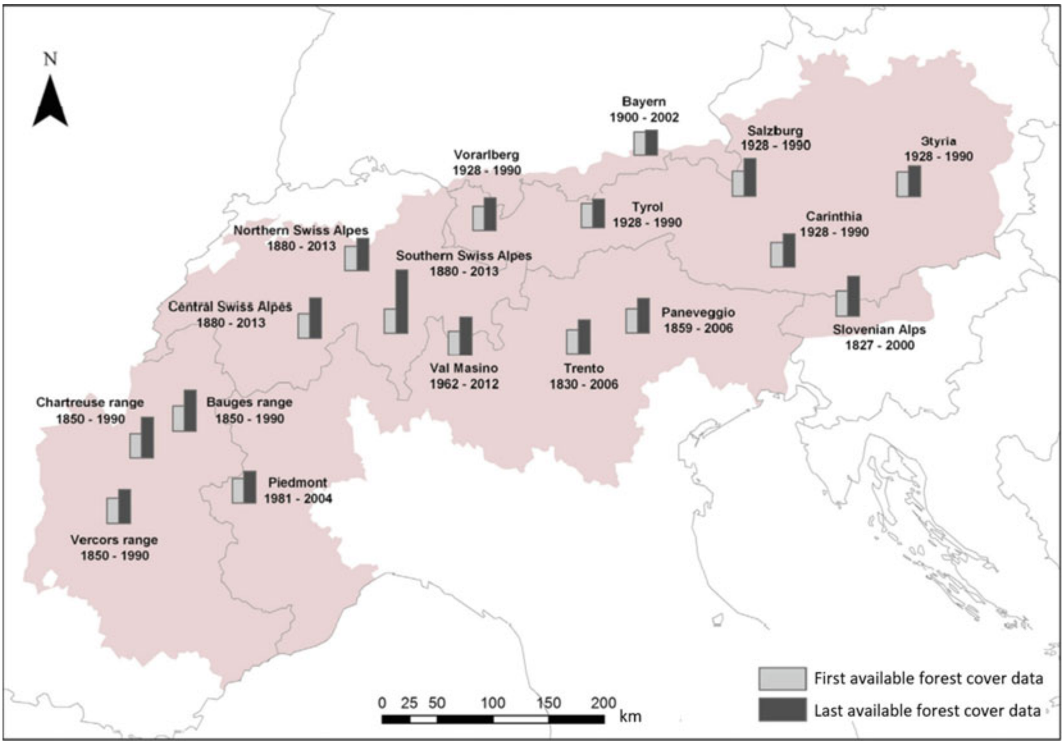


Fig. 1.53 Forest cover changes in different regions of the European Alps, indicating the omnipresent expansion of forest areas. (Modified from Bebi et al. 2017)

use on marginal areas after the Second World War has had the reverse effect (Stöcklin et al. 2007). Resampling of subalpine/alpine grasslands in the northern calcareous Alps revealed a significant long-term decline of plant species richness following land abandonment (Dullinger et al. 2003). On the other hand, high land use intensity has a negative effect on biodiversity on agricultural land (Schmitzberger et al. 2005; Niedrist et al. 2009). It is evident from several studies that both intensification and abandonment change species composition and reduce plant species richness relative to traditional land use patterns (Tasser and Tappeiner 2002; Tasser et al. 2005; Spiegelberger et al. 2006). The loss of biodiversity affects major ecosystem services and ecosystem processes and may lead, in the long term, to decreases in nitrogen mineralization, decomposition rates, nutrient availability, and soil respiration (Tasser et al. 2005). It can be concluded that the goal of sustaining high levels of biodiversity and preserving the diversity of habitats and landscapes can best be achieved by maintaining a wide range of land use types with moderate management intensity (Maurer et al. 2006; Stöcklin et al. 2007; Fischer et al. 2008; Rudmann-Maurer et al. 2008; Strebel and Bühler 2015). Moderate agricultural management intensity also consolidates vegetation cover and soil properties, thus reducing the vulnerability of Alpine ecosystems to landslides, hillslope erosion, and snow gliding processes (Tasser et al. 2003).

The initiation of mass tourism in the Alps, in particular the development of winter sport resorts (Fig. 1.55), has caused severe changes of Alpine landscapes and ecosystems. Winter tourism requires much more extensive technical infrastructures than summer tourism, and ski resorts, ski runs, chairlifts and cableways, and snowmaking facilities are constantly being expanded. Since the 1970s ski runs have been extended to

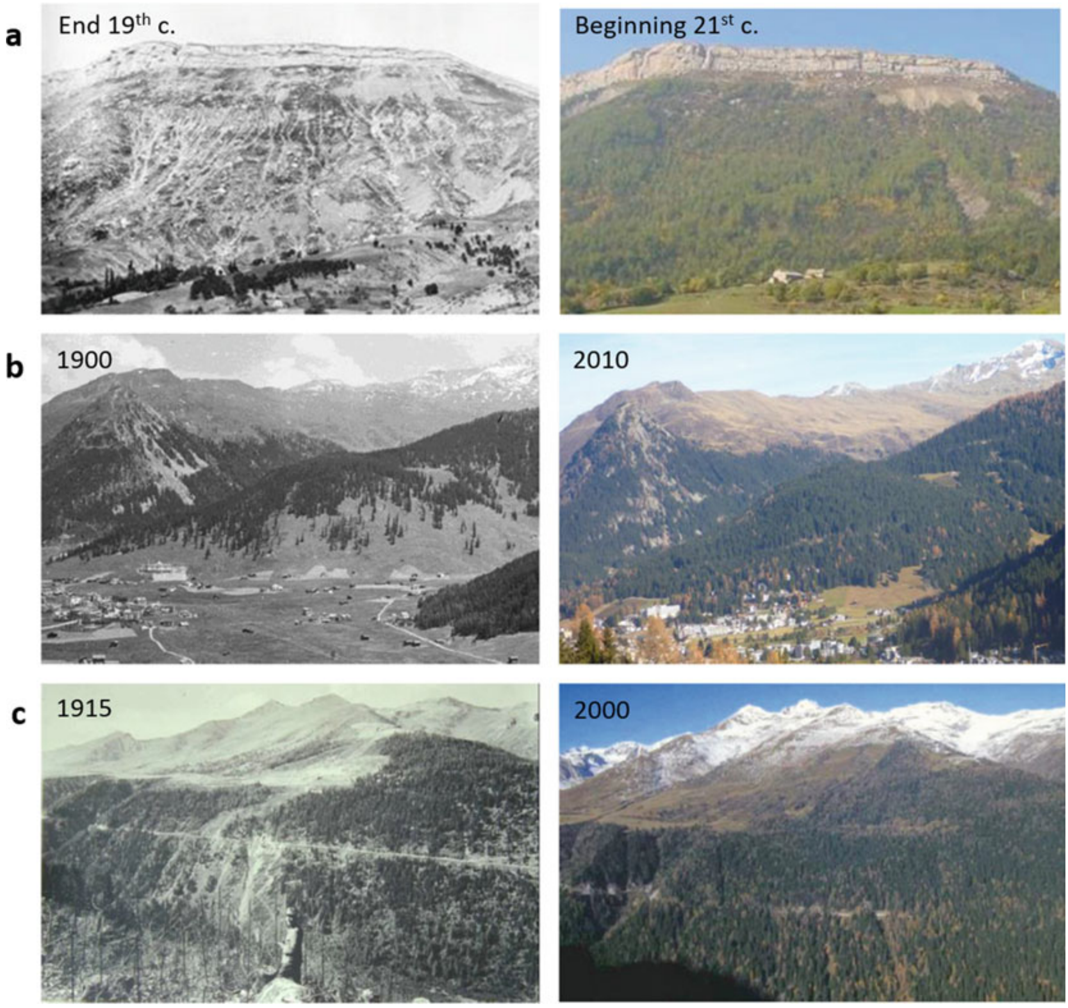


Fig. 1.54 Forest cover changes in the European Alps since the nineteenth century visualized by repeat photographs: **a** Ceüse, southern French Alps; **b** Davos, Central Swiss Alps; **c** Vermiglio, Trentino, Italian Alps. (*Source* Trento Autonomous Province Archive; modified from Bebi et al. 2017)

form wide ski highways, since the 1980s enormous skiing areas have been created, since the 1990s artificial snowmaking has been introduced, and since the 2000s even entire ridge and summit zones in skiing areas are covered with artificial snow, requiring the building of large reservoirs at high elevations (Bätzing 2018). Currently, the Alps capture 43% of total skier visits worldwide and host 80% of the major global ski areas and 38% of the global ski lifts (Vanat 2020). More than 10,000 ski lifts are located in the Alps, covering c. 28,500 km of ski runs that are distributed over ski slopes with high density per massif, pointing to the high pressure exerted by ski activities on mountain territories (Pintaldi et al. 2017). Most winter sports areas in the Alps have caused landscape damage and impairment of ecosystem services that exceeds an acceptable level (Rixen and Rolando 2013; Ringler 2016). The construction of ski runs and skiing has severe impacts on soils in alpine terrain (Fig. 1.56), implying the perturbation of topsoils and the removal of weathered soil horizons as well as subsequent problems such as soil compaction and reduction of water and air permeability, depletion of organic matter, reduction of soil aggregate stability, and nutrient imbalance (Freppaz et al. 2013; Pintaldi et al. 2017; Bacchiocchi et al. 2019). The deterioration of physical, chemical and biological soil properties in turn impairs the establishment and development of plant communities which are also adversely affected by snow compaction and the production of artificial snow. Snowmelt on ski runs is delayed by 2–3 weeks, and soil freezing under compacted snow and snowmaking-related water, salt and ion input are additional stressors that prevent a full recovery of the vegetation (Rixen 2013). Climate warming and the decline in snow cover is an increasing challenge to the winter tourism industry. Austria and Italy bear the highest weather-induced risk of decreasing winter overnight stays related to skiing tourism in Europe (Damm et al. 2017).

Other European mountain systems show many similarities in terms of land use changes over recent decades, but also major differences in historical and political evolution. Integrated in the geo-political context of Eastern Europe, the Carpathians have experienced multiple abrupt shifts in institutions, politics and economics, related to the fall of empires, the collapse of socialism, and the accession of the EU. Recurrent dramatic political, institutional and socio-economic changes have caused several shifts in land management, with land use intensification induced by economic and institutional drivers as well as land abandonment as a result of other socio-demographic and policy changes (Munteanu et al. 2017). Cultural landscapes of the mountain regions have evolved over several thousand years, characterized by small fields, scattered settlements, and large tracts of forests, while larger-scale agriculture was confined to the lowlands. Landscape transformation due to forest clearing for agriculture and for pastures was a dominant process in the Middle Ages up to the nineteenth century. Over the past 200 years, forest cover changes in the Carpathians reflect the turbulent political history, expressed in regionally varying change patterns, while the overall long-term trend indicates an increase in forest areas (Fig. 1.57) (Munteanu et al. 2014): Throughout the nineteenth century until the end of the Austro-Hungarian Empire (1918), forest cover was reported to be stable or slightly decreasing, with a decline in forest cover in the Ukrainian, Romanian and Slovakian Carpathians. The following time periods are characterized by generally increasing forest cover, but regional deviations. During the Interwar and Socialist period, forest cover increases prevailed in the northern Carpathians, while cases of forest loss occurred in Romania and Slovakia. After 1990, forest cover increased with higher mean annual rates, with notable exceptions in parts of the Romanian Carpathians. As in other Carpathian countries, Romania saw a substantial decline of mixed and coniferous forests between 1985 and 2010. Simultaneously, a large-scale successional encroachment of deciduous tree species onto abandoned land has commenced, leading to a net increase in forest cover since the mid-1980s (Griffiths et al. 2014; Vanonckelen and Van Rompaey 2015). Clear-cutting activities (both legal and illegal logging) and widespread natural disturbances, related to an increasing vulnerability of spruce plantations to pests and pathogens, point to regionally highly dynamic forest cover changes. Disturbance patterns in Romanian and Ukrainian forests were attributed to loopholes in national forest laws and illegal harvesting, causing severe damage in valuable old-growth forests (Kuemmerle et al. 2009; Knorn et al. 2013).

In general, forest cover increases in the Carpathians have been synonymous to decreases of agricultural land. Agricultural abandonment on marginal lands and on large tracts of land previously used by state farms accelerated after 1990 due to lack of agricultural subsidies, decreased profitability and migration of labour to western Europe (Kozak 2010; Baumann et al. 2011; Kuemmerle et al. 2011; Bucala-Hrabia 2018). Much cropland has also been converted to grassland. A significant decline of transhumance during the twentieth century has caused considerable forest regrowth at treeline elevations (Shandra et al. 2013; Weisberg et al. 2013). Declining livestock numbers and widespread

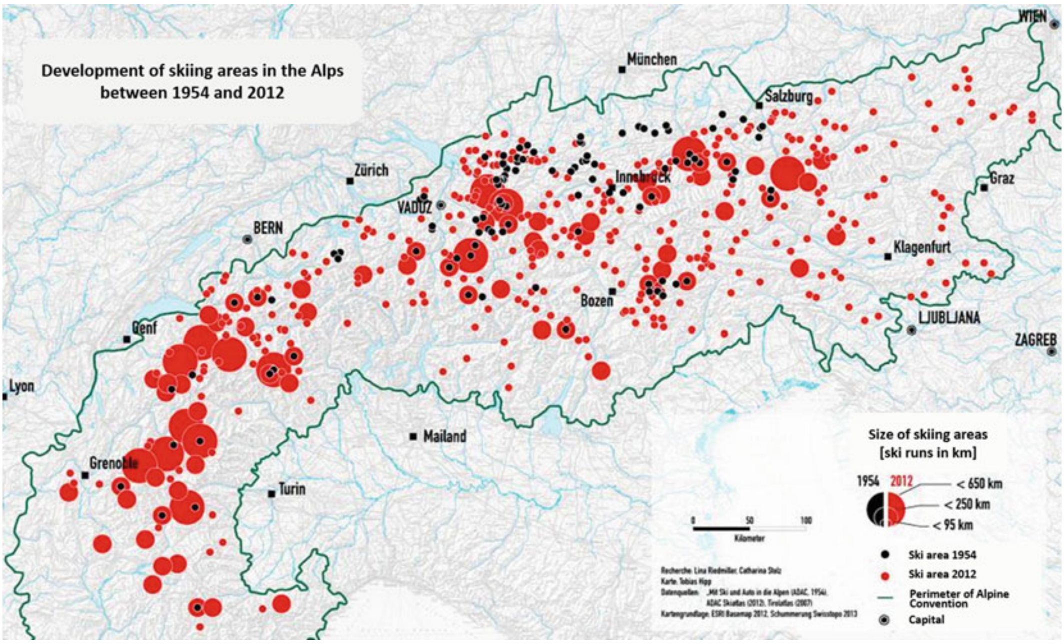


Fig. 1.55 Evolution of the number and size of skiing areas in the European Alps between 1954 (black) and 2012 (red). (Modified from www.alpenverein.de)

Fig. 1.56 Large tracts of land in the European Alps were reshaped in order to develop the ski industry as here above Ischgl, Austria, one of the top winter sport destinations in the Alps. (*Photo* © Udo Schickhoff, August 11, 2011)

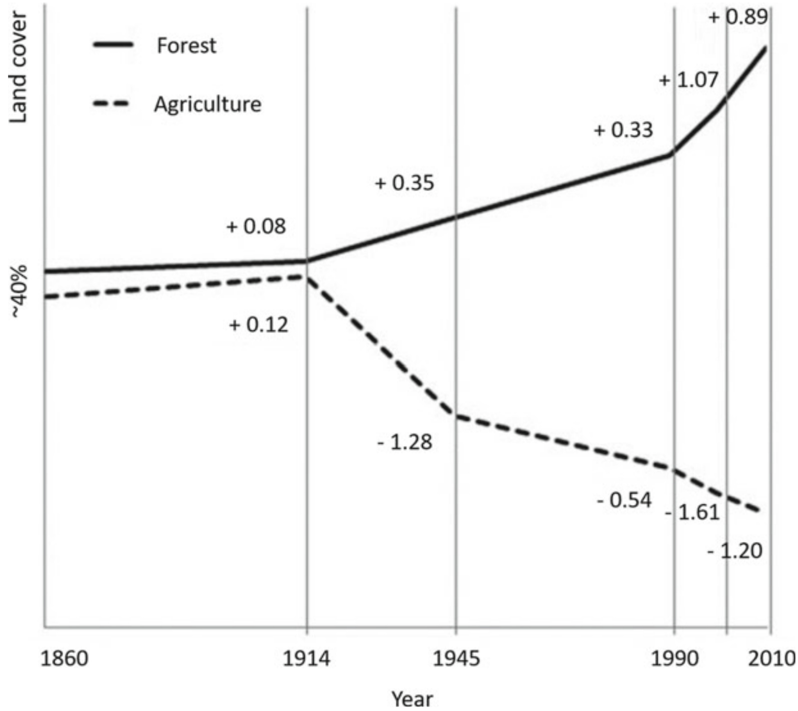


Fig. 1.57 Conceptual reconstruction of the long-term forest and agricultural land cover dynamics in the Carpathians; mean annual rates of land change based on case studies in Munteanu et al. (2014). (Modified from Munteanu et al. 2017)

forest succession have reduced the diversity and area of mountain pastures and meadows, resulting in significantly decreasing species richness and to the entire disappearance of some unique grassland communities (Bezák and Halada 2010; Kricsfalusy 2013). Another significant recent trend in land cover dynamics in the Carpathians is a considerable increase in built-up areas, related to urban sprawl in the lowlands and tourism development and the building of second homes in the mountains (Gerard et al. 2010; Mika 2013).

Mediterranean mountains have one of the longest histories of human intervention, with multiple land use/land cover changes transforming Mediterranean landscapes (Blondel 2006). However, the conspicuous degradation of mountainous environments is arguably a comparatively recent phenomenon, as evident from massive deforestation and soil erosion occurring between 1800 and 1950 (McNeill 1992). Accelerated population growth in the early nineteenth century and improved road connections, accessibility and transport facilities increased the pressure on mountain resources and the exploitation of forests. Deforestation slowed down when large-scale emigration of mountain people began in the late nineteenth century. After the Second World War, Mediterranean mountains have become largely marginal territories, predominantly characterized by rural emigration, abandonment of agricultural land, decline of transhumance of sheep and goats, cessation of grazing pressure, and reforestation of abandoned hill slopes (Papanastasis 2012). Rural depopulation, farmland abandonment and increases in shrubland and forest cover are ubiquitous in Spanish and French Mediterranean mountains including the Pyrenees since the 1950s (Tatoni et al. 2004; Lasanta-Martinez et al. 2005; Chauchard et al. 2007; Ameztegui et al. 2010). The Spanish Pyrenees stand out as one of the European hotspots of forestland increase between 1990 and 2006 (Kuemmerle et al. 2016). In recent decades, winter tourism with the construction of ski resorts has emerged as a land management alternative in the Pyrenees, albeit still with limited territorial impact as compared to the Alps (Lasanta et al. 2013). Similar land transformation processes initiated by rural emigration and manifested by abandoned agricultural land, declining pastoralism and increase in woodland are common and widespread phenomena in mountains of Italy (Torta 2004; Falcucci et al. 2007) and Greece (Papanastasis 2007, 2012). Terraced landscapes are a characteristic anthropogenic imprint on the relief of Mediterranean mountains. Agricultural terraces which are subject to land abandonment and non-maintenance pose an increased risk to gully erosion, terrace failure and landslides which is mitigated, however, in case of colonization by a dense shrub cover or by reforestation (Garcia-Ruiz and Lana-Renault 2011; Tarolli et al. 2014). Land abandonment, cessation of pasture grazing, and increased reforestation induce decreasing availability of habitats for many species of open habitats, but may have beneficial effects for forest-dwelling species (Blondel et al. 2010). While the decline in structural diversity of Mediterranean landscapes may have caused a decrease in floristic species richness in higher successional stages, a recent meta-analysis showed that the overall effect of land abandonment is a slight increase in plant and animal species richness and abundance, albeit with great differences in effect size between taxa, spatial–temporal scales, land uses, landforms, and climate (Plieninger et al. 2014).

Land use in mountains of northern Europe has a long tradition of several thousand years, regardless of harsh climatic conditions and mountain environments being remote and less densely settled. Prehistoric animal husbandry evolved during the Late Neolithic, associated most likely with mountain transhumance (Hjelle et al. 2006). In the Bronze and Iron Age, the use of mountain summer farms became established in southern Norway (Kvamme 1988). The eventful land use history includes a gradual intensification from the seventeenth century onwards, and the full development of Saami reindeer nomadism in the sixteenth and seventeenth centuries in northern Scandinavia where the use of seasonally inhabited farms by farm households has been of limited importance in most inland areas (Moen 2006; Müller-Wille et al. 2006). The high number of mountain summer farms in Norway in the mid-nineteenth century indicates a peak in land use intensity, followed by a strong decline in seasonal farms (Setten and Austrheim 2012). The transition from intensive reindeer herding to more extensive large-scale herding still practised today occurred towards the end of the nineteenth century (Lundmark 2007). Modern land use/land cover changes in Scandinavian mountain landscapes are predominated by forest succession as in most other European mountain regions (Emanuelsson 1987; Hofgaard 1997; Löffler et al. 2004; Bryn 2008; Bryn and Hemsing 2012; Potthoff 2017; Bryn and Potthoff 2018). However, the grazing regime controls establishment of shrubs and trees and treeline expansion to a comparatively greater extent, in particular in northern Scandinavia where semi-domestic reindeer husbandry still exerts a strong influence on mountain ecosystems (Moen 2006). Herd sizes have increased considerably over recent decades (Forbes and Kumpula 2009), and increased reindeer grazing pressure has caused shifts in plant species composition, declines in the cover of lichen heaths, soil erosion, a decline in carrying capacity, and a decrease in productivity, suggesting an overuse of grazing resources at least in some parts of northern Europe (Löffler 2004, 2007; Pape and Löffler 2012). In Finnish Lapland, deteriorating pasture conditions were attributed by the media to intensive Saami reindeer farming and overgrazing. Harkoma and Forbes (2020) highlighted that the underlying causes are more complex and include, inter alia, climate change, regulatory challenges, range restrictions, and other uses of the land such as forestry, infrastructure development, mining, and recreation. Reindeer grazing was also observed to counteract processes of climate-induced encroachment of tall shrubs in tundra (Ims et al. 2013; Bråthen et al. 2017). The recent development in reindeer husbandry is in contrast to the strong decrease of livestock grazing in Norwegian unimproved land since the 1950s (Austrheim et al. 2011). The decrease of livestock grazing is in line with the trend of abandonment of seasonal farming reported from all over Norway (Eiter and Potthoff 2016). Tourism has been part of the mountain economy in Scandinavia since long and has gained significance for local and regional development in recent decades with the decline in extractive industries and agriculture (Fredman and Heberlein 2005). In a warmer future, a northward shift of winter tourism is expected, however, potential ski tourism development zones frequently intersect with established protected areas (Demiroglu et al. 2019; Fredman and Chekalina 2019).

America

With regard to North American mountains and plateaus, a clear distinction must be made between a long period when Native Americans dominated land use and a period of Euro-American dominance of land use that started in the mid-nineteenth century (Vankat 2013). The importance of Native American agriculture increased after about 4,000 years BP with the erection of permanent small settlements. It is often assumed that the impact of Native Americans on mountain environments had been largely insignificant. However, to a certain degree their activity had modified forest extent and composition, created and expanded grasslands, and there is evidence of increased fire frequencies and altered fire regimes at the landscape scale (Denevan 1992; Allen 2002; Roos et al. 2010). Land use effects have clearly reached another dimension since the 1850s when the transition period from Native American to Euro-American dominance of land use ended—three centuries after the first Spanish exploring party had entered the Colorado Plateau. From the Rocky Mountains to the coastal ranges, permanent Euro-American settlements were established, mainly driven by extractive industries such as logging, mining, and grazing, and facilitated by the development of transportation routes, in particular railroads (Wyckoff and Dilsaver 1995; Wildeman and Brock 2000). In the Colorado Front Range, a gold find in 1858 resulted in a gold rush that marked the beginning of permanent Euro-American settlement in the Rocky Mountains (Veblen and Lorenz 1991). In California's mountains, mining emerged as the dominant form of land use after the discovery of gold in 1848 that ignited processes of economic development, settlement, environmental modification, and political adaptation still relevant for California's landscape of today (Dilsaver et al. 2000). In its initial stage, the California gold rush had a comparatively minor effect on landscapes and ecosystems. This changed drastically, however, after the introduction of hydraulic mining, associated with enormous water consumption, the use of dangerous chemicals, the practice of dumping mining debris into mountain rivers, and vast sediment loads. Following the legal ban on hydraulic mining in 1884, the mining industry declined, mining boomtowns became depopulated, and agriculture in the Central Valley became the driving force of California's economy (Alagona et al. 2016). The transformation of the semiarid Central Valley into one of the most productive agricultural regions in the US has required the extraction and diversion of vast amounts of water, primarily from the Sierra Nevada (Ives et al. 1992a). In general, mining was the principal industrial activity that attracted people and brought systematic settlement to the western mountains in Canada and the US from 1850 to 1930 (Harris 1997).

Other significant anthropogenic disturbances of the early phase of Euro-American land use dominance included logging, grazing, and fire management. The nineteenth-century mining booms resulted in heavy demands on timber resources for town construction, fuel, and mine props, and led to large-scale logging of montane forests. In many mining areas, nearly all the timber was cut, causing erosion and soil depletion. Even beyond the immediate mining areas, a large percentage of forests had been logged for fuel and construction purposes towards the end of the nineteenth century, as in the present Rocky Mountain National Park (Veblen and Lorenz 1991). Widespread logging of ponderosa pine (*Pinus ponderosa*) forests commenced in the 1870s when logging became a major industry on the Colorado Plateau, occasionally even at high elevations (Vankat 2013). In the Sierra Nevada, a large timber industry developed to exploit sugar pine (*Pinus lambertiana*) and sequoia

(*Sequoiadendron giganteum*) forests (Alagona et al. 2016). After three decades of heavy logging, the California Board of Forestry estimated in 1886 that already one-third of the Sierra Nevada's timber had been harvested (Beesley 2004). During this deforestation phase, the foothills treeline in the Sierra Nevada was raised by up to 600 m (Dilsaver et al. 2000).

Livestock grazing was the first broad-scale impact on the vegetation of mountains and plateaus in the American West after European colonization. Livestock numbers and grazing impact reached another order of magnitude in the second half of the nineteenth century, when rapid increases in large, commercial ranching operations were supported by the completion of the transcontinental railroad, the final subjugation of the nomadic Native American groups, expanding markets, as well as by the entrepreneurial resource utilization ethic that focused on maximum harvest for maximum profit (Raish 2004). The intensification of the nineteenth-century open-land sheep and cattle ranching also affected higher elevations as evident from severe grazing damage on the Colorado Plateau by high populations of these introduced herbivores (Cole et al. 1997), while seasonal migrations to and pressure on alpine grazing lands remained much less significant compared to Old World mountain regions (Bock et al. 1995). The commercial livestock industry declined at the turn of the century, mainly due to overstocked ranges, droughts, and brutal winters (Huntsinger et al. 2010). As early as 1864, increasing concerns about depleted forests, polluted waterways and degraded rangelands resulted in the designation of California's Yosemite Valley and nearby Mariposa Grove as a nature reserve for conservation and recreation (Beesley 2004). This first groundbreaking success of the American environmental movement was followed eight years later by the establishment of Yellowstone National Park, the world's first national park, and by the establishment of federal forest reserves/national forests around the turn of the century.

The alteration of fire regimes was an important effect of early Euro-American land use that had a lasting impact on vegetation and landscape, still unfolding today (Vankat 2013). Initially, livestock herders set fires to clear vegetation and stimulate forage growth, and many fires were set by mining and other human activities, either accidentally or intentionally. Post-fire stands often showed a shifted dominance of tree species, and post-fire recovery processes still influence recent forest cover changes (Rodman et al. 2019). However, fire frequency decreased in areas where livestock grazing expanded since the shrub and herb layers in open forests and meadows were greatly reduced. This culminated in extensive and effective fire suppression, an important legacy of public land management, becoming widespread in the early twentieth century since fires were then viewed as unnatural events from which vegetation should be protected. For many decades, forests remained unburnt, causing again changes in species composition, structure and dynamics. For instance, pyrophytic species such as the giant sequoia have not been exposed to fire for almost a century and did not regenerate (Harvey et al. 1980). Fire exclusion resulted in an unnatural level of fuel load and increased tree densities in mountain forests, leading to landscape-scale crown fires (Fig. 1.58). In subalpine forests with longer natural fire intervals, fire suppression had less serious implications. In recent decades, fire management practices such as prescribed burning have been developed that are only partially successful in countering the effects of long-term fire exclusion, and bear as well the risk of exceptionally large and intensive crown fires (Fulé and Laughlin 2007; North et al. 2015; Thompson et al. 2018).

Land use management in North American mountain regions remained to be driven by extractive industries during the twentieth century, albeit to a lesser extent, while the environmental movement has strengthened, and tourism, recreation, and residential development have become increasingly important. Mining declined after the global economic depression in the 1930s, but retained its importance as major employer and source of adverse environmental impacts (Fox 1997; Gardner et al. 2013). The migratory sheep

Fig. 1.58 Fire suppression for decades has increased the risk of landscape-scale crown fires, with recent climate change adding to this risk (Yosemite National Park, Sierra Nevada, USA). (*Photo* © Udo Schickhoff, August 23, 2019)

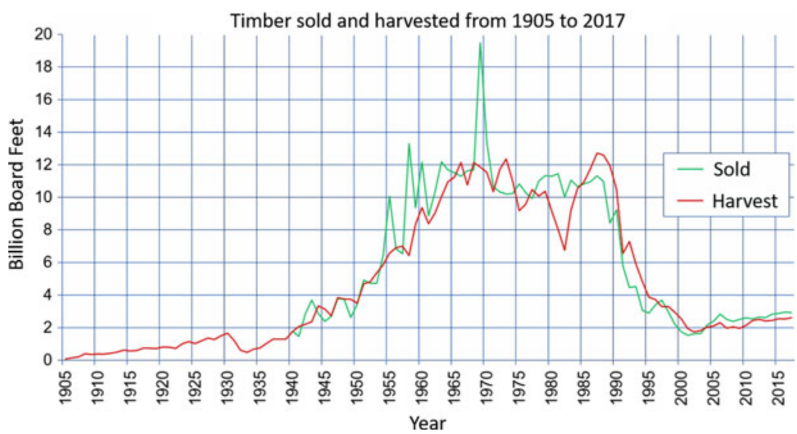


Fig. 1.59 Timber sold and harvested in the national forest system of the United States from 1905 to 2017 (sold data not available before 1940). (modified from www.fs.fed.us)

and cattle industry rapidly declined since the 1930s. The Taylor Grazing Act ended open range grazing in the western US in 1934, and herd movements were further restricted by environmental laws and the addition of more national parks. The percentage of total land area used for grazing in the Rocky Mountains decreased significantly (Cline 2013). Today, transhumance is no longer economically significant in North American mountain regions (Cunha and Price 2013). At lower elevations, however, the influence of the western range livestock industry is still strong and grazing-induced ecological changes have long been debated (Donahue 2005). One of the responses to widespread forest depletion was the establishment of the US Forest Service in 1906, who tried to restore degraded lands, severely limited grazing and regulated logging during the twentieth century (Dilsaver et al. 2000). Henceforth, national forests have been managed under a multiple-use, sustained-yield mandate, combining extractive uses as well as recreation and conservation. The balance among these uses has been spatio-temporally differentiated (Alagona et al. 2016). In the first half of the twentieth century, the focus was on conservative use and resource protection, and national forests were rarely logged (Fig. 1.59). Large-scale timber extractions were resumed in the 1950s, mainly triggered by the postwar housing boom which was fuelled by the growing prosperity of a fast growing population. More stringent environmental legislation, rapid development of plantations (mainly in the Southeast), and foreign producers capturing the US wood supply market were major reasons why the national forest timber harvest plunged to prewar levels in the 1990s (Bosworth and Brown 2007). The sharp decrease in harvest from national forests helped to ensure that the total forest area in North America is currently roughly stable (Masek et al. 2011).

Current land use in North American mountains is characterized by an increasing dominance of conservation, recreation, residential and commercial development, while resource extraction is losing importance. In recent decades, a significant migration to mountain regions can be observed as part of a national population shift to the South and West and from urban to rural areas. Rapid growth of mountain towns and dispersed, landscape-consuming residential development in rural areas reflect emerging land use patterns created by amenity migration, as described for the Colorado and Canadian Rocky Mountains (Riebsame et al. 1996; Leinwand et al. 2010; McNicol and Glorioso 2014) and for the Sierra Nevada (Loeffler and Steinicke 2006). Amenity migrants include semipermanent residents as well as homeworkers and retired persons, establishing permanency in their mountain homes (Moss 2006). Significantly increased housing density has also been the most significant land use change on lands surrounding US national parks in recent decades (Hansen et al. 2014; Resler et al. 2020). At the same time, tourism has received a huge boost and has become an important element in the local economy. Ski area development has dramatically increased (Humphries 2020). The Rocky Mountains are now an international winter tourism destination, giving rise to controversial discussions on further expansion of ski resorts (Childers 2012). The second pillar of the large-scale two-season mass tourism is the increasing nature-based summer tourism, for which the national parks represent a major resource, and which triggers diverse recreation impacts (e.g. Willard et al. 2007). The recent transformation into amenity landscapes is associated with extensive infrastructure networks, a visible expression of contemporary land use in many mountain areas (Alagona et al. 2016).

The historical development of land use in mountain regions of Central and South America exhibits differences to that of North America in the sense that indigenous highland peoples had a much higher population density over many centuries, and had reshaped the environment to a comparatively greater extent. In the Andes, at least thirteen to fourteen thousand years of continuous human occupation had preceded European contact (Erlandson and Braje 2015). Palaeo-Indian hunters burned woodland to expand the game-rich ecotone between forests and the alpine zone and initiated the large-scale deforestation of the highlands. The transformation from woodland to grassland continued to be driven forward when agriculture and pastoralism appeared around 6–7000 BP and land for cultivation and grazing use was needed (Ellenberg 1979; Baied and Wheeler 1993; Gade 1999). This transformation resulted, inter alia, in upper treelines being truly anthropogenic (Miehe and Miehe 2000; Sarmiento and Frolich 2002). Later, advanced civilizations such as the Tiahuanaco and Inca empires developed a highly successful agriculture including the sophisticated management of raised fields and irrigated terraced slopes, and thus remodelled the landscape of whole valley systems. Agriculture extended over a considerable range of altitude from the lowlands to over 5000 m and sustained more than 10 million people in the Central Andes (Grötzbach and Stadel 1997; Borsdorf and Stadel 2015). In late Pre-Columbian times, intensive agriculture was most widespread in the northern and central Andes where, below about 3000 m, maize cultivation resulted in massive landscape transformation, and areas above 4000 m were used for hunting and camelid (llama, alpaca) grazing (Knapp 2007). Grasslands were maintained by clearing, grazing, and burning (Gade 1999). Humans have modified most forest, shrubland, grassland, and wetland vegetation types in the Andes for millennia (Young et al. 2007; Young 2009).

The Spanish conquest (AD 1532) marked the start of profound transformations of Andean society, culture, economy, and environments. Colonial rule aimed at exploiting natural and human resources and at missionizing the indigenous population, focusing on high-yield mining areas, productive agricultural areas, and generally on densely populated regions (Borsdorf and Stadel 2015). The southern Andes were relatively neglected, and have experienced to date a much lower appropriation of land for human use (Hoekstra et al. 2010; but see Inostroza et al. 2016). The Spaniards modified or destroyed traditional community organization, while indigenous agricultural techniques and land use systems largely collapsed (Grötzbach and Stadel 1997). The introduction of land tenure systems, crops, domesticated animals, tools, technologies, institutions, and peoples can be termed an early globalization, involving a variety of impacts on the mountain environment that have been massive in the long term and sometimes substantial or even devastating at a local or regional level (Knapp 2007). In the wake of the Spanish conquest, fundamental socio-economic and administrative changes were introduced, leading to severe societal disruption. The indigenous population started to decline drastically, mainly due to the spread of European diseases or forced labour (Ives et al. 1992b). Declining subsistence needs of a shrinking population resulted in the abandonment of a large number of terraces as well as of raised fields that occupied large areas in highland flats, reflecting that depopulation was associated with disintensification of land use, characteristic of the entire Andes in the 1600s and 1700s (Knapp 2007; see also Butzer and Butzer 1995 for Mexico). Nevertheless, cultivation patterns were continued incorporating a wide range of introduced European crops, which could be used particularly at higher elevations. The most important change in traditional grazing patterns was the replacement of domesticated Andean animals (llamas, alpacas) by European sheep, goats and cattle that contributed most saliently to peasant livelihoods (Gade 1992). Due to the dramatic depopulation of vast tracts of the Andes after the Spanish conquest, large land areas were available for grazing, which was less labour-intensive than traditional farming (Borsdorf and Stadel 2015). Whereas traditional Andean grazing patterns are associated with sustainable production systems, large-scale soil erosion problems and drastic changes in vegetation structure in the post-conquest era are commonly attributed to overgrazing by introduced livestock to which the native vegetation is not adapted (Browman 1974; Millones 1982). Grazing-ecological studies confirm that grazing systems with introduced cattle have a lower efficiency in the use of pastoral space, show a concentration of cattle in fewer places, and have a higher magnitude of environmental impact (Molinillo and Monasterio 2006).

Nevertheless, the colonial period with the population decline of indigenous Andean highland peoples was generally associated with environmental recovery, with the exception of impacts originating from mining activities and from the demand of wood (Denevan 1992; Knapp 2007). The demise of much woodland accelerated since greater quantities of wood were needed for diversified uses including mining activities and charcoal production, controls on wood cutting were far less strict than in Inca times, and forest grazing by introduced livestock caused severe damage. In the course of time, human agency has destroyed over 90% of native Andean forests (Gade 1999). Wood shortages are meanwhile alleviated by ecologically detrimental plantations of exotic eucalyptus and pine species that accounts for a part of the recent increases in woody cover of mid-elevation areas and highland grasslands in the tropical Andes (Balthazar et al. 2015; Aide et al. 2019; but see Restrepo et al. 2015 for the Colombian Andes). At the time of the Spanish American wars of independence in the early nineteenth century, the population started to expand again, followed by an exponential growth since the 1920s that has been attenuated in recent years. The increasing integration into the global system of trade and transfer and the high population growth have resulted in highland resources having been more intensively exploited, and European modification of the environment having accelerated. The

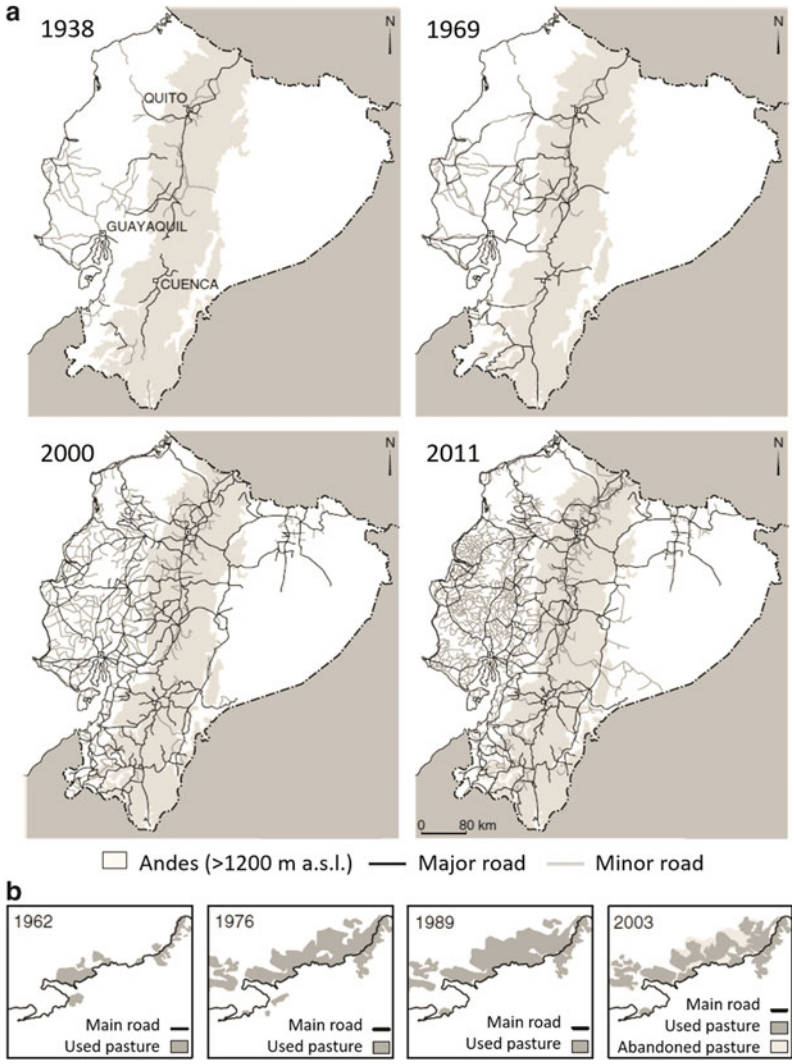


Fig. 1.60 Impacts of expanding rural road networks in the Andes; **a** road networks of Ecuador in 1938, 1969, 2000 and 2011; **b** evolution of land use forms along the main road in the upper Rio San Francisco valley over a 40-year period. (Modified from Peters et al. 2013)

colonization of the highlands was reinforced, the agricultural frontier moved up into the páramo, and the agricultural production intensified, characterized by an increased use of chemical fertilizers and pesticides, indiscriminate use of fires, overgrazing, construction of drainage systems and roads (Monasterio 1980; Hess 1990). Human disturbance still plays a primary role in shaping páramo vegetation patterns, diversity and ecosystem services (Suárez and Medina 2001; Vasquez et al. 2015; Hofstede and Llambi 2020). In general, extensive road construction played a crucial role for land reclamation, with facilitated access to remote valleys supporting deforestation and agricultural intensification (Fig. 1.60) (Peters et al. 2013; Quintero-Gallego et al. 2018).

Intensified use of highland resources applies in particular to mining-related resource extractions. Industrial-scale silver and gold mining in the Andes was already a widespread source of livelihood in Inca times and continued through the colonial and post-colonial periods. One of the largest cities in America at the beginning of the seventeenth century was Potosí, an old silver mining town at 4100 m in present-day Bolivia which had grown to 150,000 inhabitants (Borsdorf and Stadel 2015). With the globalization of economy in recent decades, the exploitation of

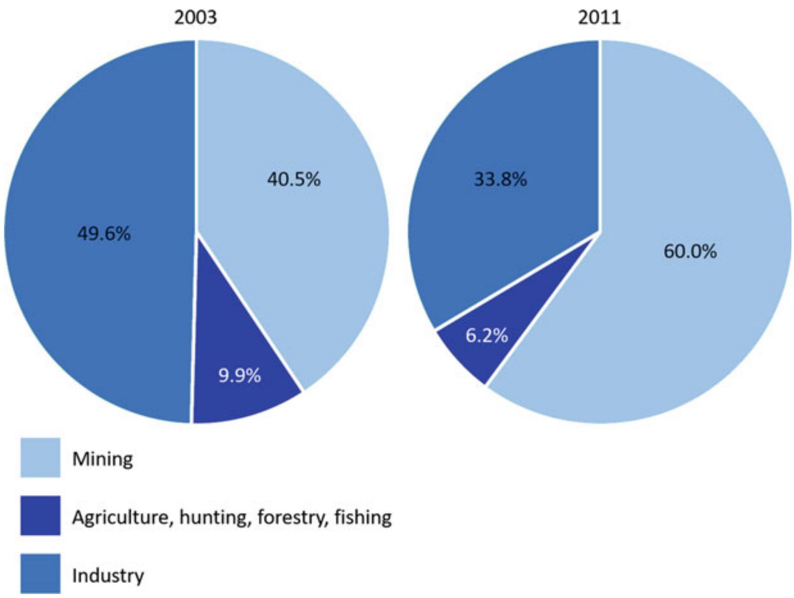


Fig. 1.61 Value share of exports by sector in Chile 2003–2011. (Modified from Simpson et al. 2014)

Andean mineral resources of global interest, such as copper, gold, zinc, tin, and molybdenum, has greatly expanded, controlled by multinational corporations. Water consumption, contamination of water and soils and other negative environmental impacts of large-scale projects are substantial, as vividly illustrated by the case of the open-pit mining project Pascua Lama at an elevation above 4000 m across the border of Argentina and Chile, strongly resisted by rural communities (Romero et al. 2009). This project, aiming at extracting gold, silver and copper, exemplifies the increasing number of conflicts created among enterprises, native ethnic groups, and residents of the lowlands who depend on highland resources such as water and wood withheld from them for the extraction of industrial minerals (Marchant 2010). The huge investment in mining in the Andes has provoked a surge in social mobilization and conflict (Bebbington et al. 2008). Unsustainable highland resource use in the wake of economic globalization is obvious from the fact that Chile, the main producer of copper and associated minerals in the world, concentrates the national mining investment in the Atacama Desert where water resources are extremely scarce and water is even imported from Bolivia (Romero et al. 2009). Mining accounts for an increasing value share of Chile's exports and a significant proportion of GDP, while the importance of the primary sector is further declining (Fig. 1.61).

A common consequence of the deep social, economic, cultural and environmental transformations that have affected the mountain regions in Central and South America was the migration of highland population to the lowland. The decline of traditional mountain economies based on agriculture and livestock had triggered conspicuous migration trends from highland to lowland and from rural to urban in the second half of the twentieth century (Escobar and Beall 1982; Ives et al. 1992b; Lauer 1993; Romero and Rivera 1996; Izquierdo et al. 2018). The huge rural exodus reflected the need to improve livelihoods through getting employment, housing, education, and health services, and resulted in an explosive population growth and slum development in expanding cities. In recent years, these migration trends and the growth of cities have weakened. Globalization and international mobility support in places the emergence of a new rurality, in which international migration patterns stimulate development by increasing the remittance income of households (Yarnall and Price 2010; Borsdorf and Stadel 2015). Recently, páramos and puna grasslands have been increasingly converted to other land uses such as more intensive agriculture and afforestation, involving higher water-demanding trees and crops (Hofstede et al. 2002; Tovar et al. 2013b; Bello et al. 2014).

Spatial patterns of rural settlements are in a process of change as well, exemplified, for instance, by increasing amenity migration, in the case of Santiago de Chile supported by the state's withdrawal from regional planning, deregulation, privatization of the land market and other factors, all of which are linked to globalization (Borsdorf and Hidalgo 2009). Other examples, caused by changing socio-ecological systems at higher elevations, include the concentration of pastoral settlements in the Peruvian puna (Charbonneau 2009) and the expansion of the permanent frontier of agriculture and dwellings to higher elevations in the Ecuadorian páramo (López-Sandoval and Maldonado 2019). The latter case study also illustrates positive conservation outcomes after the establishment of communal governance of natural resources. Unlike the Himalaya (see above), the implementation of community-based management models is less advanced in the Andes and faces diverse challenges, but needs to be promoted in order to contribute to the achievement of sustainability goals (Wilson 2016; Mathez-Stiefel et al. 2017). The view that it is imperative to protect valuable or representative natural and cultural landscapes has increasingly gained ground over recent decades, reflected in a strong increase in protected areas since the 1960s, both in number and surface area (Fig. 1.62). With extraordinary scenic and cultural diversity, the Andes have a tremendous tourism potential. Tourism is considered a key sector of the national economy and an effective strategy to counter poverty and marginalization and to contribute significantly to regional development (Borsdorf and Stadel 2015). In Peru, for example, tourism has become the second most important economic sector after mining, with strongly increasing arrivals of international tourists (more than 4 million in 2017) and annual tourism-induced foreign exchange revenues of more than USD 4 billion (Baumhackl 2019). It is a major challenge for governments and local authorities to alleviate environmental impacts resulting from the spatial concentration of tourist flows, and to make tourism compatible with the ways of living of the local population.

Africa

Since time immemorial African mountains and highlands have been more attractive for human land use than surrounding lowlands since climatic and ecological conditions for agriculture

Fig. 1.62 Number and surface area of protected areas have increased considerably in the Andes in recent decades, exemplified by the Laguna Miscanti (4140 m), Atacama Altiplano, Chile, part of the National Reserve Los Flamencos. (*Photo* © Udo Schickhoff, March 24, 2017)

and for sustaining livelihoods are much more favourable (Grosjean and Messerli 1988; Hurni et al. 1992; Messerli and Winiger 1992). Mountain regions also provided refuge areas for ethnic groups as well as better protection from severe vector-borne human and animal diseases. Accordingly, most of African highlands are areas of large population concentrations, illustrated by the case of Ethiopia where some 10% of the population only is living in areas below 1500 m (Abate 1993; Piguet and Pankhurst 2009). The Atlas Mountains are just another example of sustaining higher population densities than surrounding lowlands by providing economic resources and ecosystem services (Bencherifa 1990; Montanari 2013). Abundant natural resources of varied mountain environments may have facilitated hominid evolution in eastern and southern Africa, while notable human impact dates back to Stone Age populations that had colonized most of East Africa's biomes with very low numbers of individuals by c. 100,000 BP (Spinage 2012). The use of fire, supporting the expansion of savannahs, was the first major impact changing the ecology of Africa, followed much later by pastoralism that, in turn, paved the way for agriculture. The first appearance of goats and sheep in East Africa can be assumed for c. 5,000 BP, coinciding with the terminating African Humid Period. Cattle followed subsequently, spreading slowly from there to southern Africa, while the presence of cattle north of the Sahara is dated back to the eighth-seventh millennia BP (Gifford-Gonzalez 2000, 2017). The introduction of domesticated livestock and the expansion of pastoral communities diversified land use and marked the start of a sequence of significant land cover change (Marchant et al. 2018). After the Bantu expansion in Sub-Saharan Africa and since the Iron Age, impacts from (semi)-permanent settlements, cultivation, pastoralism, and the use of fire became a more widespread and dominating force, resulting in widespread anthropogenic degradation of vegetation and deforestation over the past 2,000–2,500 years (Spinage 2012).

Palaeoecological studies give evidence of human-induced forest clearing, often associated with soil erosion, beginning around 2,000 BP in the Atlas Mountains and in the interlacustrine highlands of East Africa (Lamb et al. 1991; Taylor 1990, 1996; Jolly et al. 1997; Marchant and Taylor 1998; Cheddadi et al. 2015). Large-scale anthropogenic forest destruction appears to have also started at higher elevations in the Rwenzoris and in the Ethiopian Highlands between 1,000 and 2,000 BP, whereas mountains in Kenya and Tanzania retained more forest cover up to modern times (Hamilton 1982; Nyssen et al. 2004; Umer et al. 2007). In East Africa, forest clearings have mainly focused on productive mid-elevation areas. Thus, very little primary forest remained between 1500 and 2500 m, and montane forest vegetation is now largely restricted to protected areas (Marchant et al. 2018). For instance, the submontane forest in the eastern Arc Mountains in Tanzania has lost more than 90% of its mid-Holocene area (Hall et al. 2009). Early deforestation in Ethiopia has been on an exceptionally large scale. The resulting environmental degradation was held partly responsible for the demise of the Axum civilization during the first millennium AD (Butzer 1981).

Over the last several centuries, population growth, migration of peoples, the introduction of new crops and technologies, effects of colonialism, and economic globalization were significant drivers of extensive and pervasive land cover change. Mountain ranges in the Maghreb countries experienced a general decline in forest cover and a matorralisation process, with most of the forests being transformed into various replacement communities including dehesa-like parklands due to high frequency of fires and the intensification of land clearance and grazing pressure (Cheddadi et al. 2015). Reconstructions of human–environment interactions show that the phase of Islamization was associated with population increase and development, including expanded pastoralism, deforestation and agriculture (McGregor et al. 2009). Another phase of agricultural intensification related to colonialism in the Atlas Mountains occurred in the late nineteenth and during the twentieth century, when the mountain ranges and intermontane valleys served as delivery systems for resources for the focal areas of development in the lowlands (Hurni et al. 1992). This function has been maintained in the post-colonial period (in Morocco since 1956), with forests and silvopastoral areas further declining in recent decades, attributed to the interaction of drivers such as drought, fire, soil erosion, and the increasing pressure on resources associated with socio-economic change (Hammi et al. 2010; Chebli et al. 2018; Kouba et al. 2018). Recent transformation processes in mountain livestock farming systems are widening the gap between the utilization of natural resources and the carrying capacity of mountain ranges, in particular in the Middle Atlas where pastoralism always played a predominant role. Socio-ecological changes include the commercialization of pastoralism and a general decline of transhumance, manifested in increasing sedentarization, the decline of traditional institutions regulating herd mobility, reduced pastoral territories and herd mobility, and increased livestock numbers, spatial concentration of herds and grazing season duration (Breuer 2007; El Aich 2018). The ongoing overuse of rangeland resources is a striking contrast to the mountain ranges in the northern Mediterranean basin. Although migration to lowland cities or abroad has a long tradition (in Morocco over the entire post-independence period), and remittance generation has considerably improved living conditions (de Haas 2009; Berriane et al. 2015), unsustainable use of economic resources has not been significantly alleviated. International migration has resulted in increasing agricultural productivity rather than in retreat from agriculture (de Haas 2006; Rössler et al. 2010), while farmers diversify their sources of income with, inter alia, tourist-related activities. Climate change will add a significant challenge to environmental and anthropogenic systems in the Atlas Mountains (Linstädter et al. 2010; Schilling et al. 2012).

The Ethiopian Highlands have been almost entirely reshaped into an anthropogenic agricultural landscape. Favourable natural resources have attracted human settlers ever since, thus deforestation is a very old phenomenon. Many centuries of land resource utilization by a growing population, mainly subsistence agriculture with crop cultivation and animal husbandry, have reduced the original forest cover of c. 80% to below 5%, with the remaining forests located in the southwestern part of the highlands (Hurni et al. 1992). The eventful history explains the spatio-temporal differentiation of land cover changes, with political stability/instability, foreign invasions, population growth, droughts, locusts, repeated famines, and economic prosperity being most important drivers of land use intensity and cultural landscape evolution. Evaluations of historical travel accounts revealed that over the past centuries phases with widespread land degradation alternated with phases of recovery, and that in many parts of the highlands closed forests were already completely absent in the early nineteenth century, in particular within the well-populated elevational belts between 1500 and 2700 m (Ritler 1997, 2003; Munro et al. 2008). A series of historical photographs of 1868 clearly shows that the status of natural resources in northern Ethiopia was already very degraded 150 years ago (Nyssen et al. 2009). After recovery from the major famine and epizootic of 1889–92 and the influenza pandemic of 1918–19, the highlands experienced steady population growth under higher political security over the following decades, resulting in local migration to agriculturally marginal zones where population pressure on land resources (fuelwood, grazing lands, new cultivation areas) increased. Cultivation expanded to steeper slopes and from the long-term mid-elevation settlement zones into lower elevations, while the upper limit of cultivation was shifted to just below the frost line (McCann 1995). An expansion of land use into higher elevations was also observed in the Bale Mountains (Miehe and Miehe 1994; Kidane et al. 2012; Hailemariam et al. 2016). Several studies confirmed a deforestation trend in favour of cultivation over the second half of the twentieth century (Kebrom and Hedlund 2000; Zeleke and Hurni 2001; Bewket 2002), continuing in places to the present day, partly driven by the government policy on land resources and land rights, and by the market-oriented production of high value crops (Lanckriet et al. 2015; Tolessa et al. 2017; Solomon et al. 2018; Strobelt and von Kocemba 2020). It also needs to be highlighted that the population has increased from 6.6 million in 1868 (Nyssen et al. 2009) to 115 million in 2020 (according to UN data), while over 90% of the population's energy requirement is still obtained mainly from biomass (Lemenih and Kassa 2014). Small remnants of the forest climax vegetation only remained in sacred groves around churches and in isolated areas (Wassie et al. 2010; Aerts et al. 2016). Even sacred church forests are threatened by human disturbance (Cardelús et al. 2019).

In general, processes of deforestation, overgrazing, and soil erosion over long time periods have resulted in tremendous land degradation (Nyssen et al. 2004, 2015). The northern highlands appear to be the most severely eroded part of Ethiopia, showing high to extremely high soil loss rates, decreased agricultural productivity (crop and livestock), and increased famine vulnerability (Hurni et al. 1992). Erosion surveys in the late 1960s prompted the initiation of nationwide soil and water conservation programmes and reforestation activities (eucalypt plantations), supported by international development aid after the disastrous drought in the early 1970s (Munro et al. 2008). Meanwhile, positive outcomes of these land rehabilitation programmes, being facilitated by the growing awareness of landholders (Fig. 1.63), are clearly visible (Fig. 1.64), reflected in new eucalypt woodlands, regeneration of indigenous trees and shrubs, and improved soil protection (Bewket 2002; Nyssen et al. 2004, 2009, 2015; de Mûelenaere et al. 2014). Recovery of vegetation was also assessed in subalpine and afro-alpine zones including treeline advance, while a decrease of areas with dense forest has occurred, on the other hand, even in some protected areas (Wondie et al. 2011; Jacob et al. 2017). A promising approach to halt the process of deforestation and forest degradation is participatory forest management which was introduced with pilot projects in the 1990s and found to provide mixed results so far in terms of livelihood and ecological benefits (Ameha et al. 2014, 2016). Even though the depletion of resources continues in places, the positive impact of improved land husbandry shows that land degradation in the Ethiopian Highlands is not principally irreversible (Nyssen et al. 2009). Land rehabilitation programmes should be supported by a rural development policy promoting livelihood strategies that are both environmentally friendly and economically sound. A promising path of rural development could be the shift from the traditionally preferred 'cereal crop-livestock mix' dominated livelihood strategy to one dominated by cash income-based activities such as off-farm business, honey production, poultry, and horticulture (Babulo et al. 2008). To relief pressure from the chronically food-insecure highlands, the government has conducted (much-criticized) resettlement programmes. Rural–urban migration continues to occur at high levels, while international migration flows out of Ethiopia are relatively small (although a much-desired possibility) as are the impacts on the local economy by remittances (Fransen and Kuschminder 2009).

According to historical accounts (compiled in Spinage 2012), mountain regions in Uganda, Rwanda, Kenya and Tanzania have lost much of its forest cover during the past 200 years, often resulting in increased soil erosion and more frequent landslides. In the highlands of Kenya, traditional land use patterns were completely transformed during colonial times, when the colonial rulers established 'white highlands', with white settlers developing export-oriented agriculture on large-scale farms. Following independence in 1963, highland areas were subdivided and transferred to indigenous small-scale farmers, resulting, inter alia, in intensified mixed farming systems, expansion into agriculturally marginal areas, economic marginalization, forest depletion, and land degradation (Hurni et al. 1992). Rapid population growth over recent decades and economic globalization has resulted in substantial agricultural expansion in the Mount Kenya area (Kiteme et al. 2008). Pressure on water and land in the foothills has more recently increased by the expansion of horticultural agribusinesses, while land use in the region remains dominated by small-scale crop and

Fig. 1.63 Land rehabilitation programmes, such as the construction of terraces for soil and water conservation, enjoy firm support from local communities and are actively supported (Tigray, northern Ethiopian Highlands). (*Photo* © Udo Schickhoff, February 21, 2020)

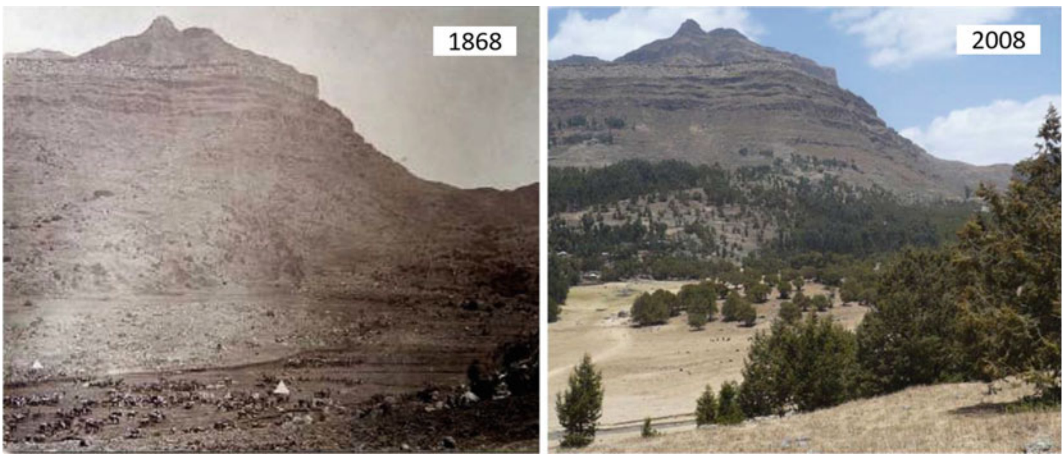


Fig. 1.64 A vivid illustration of successful land rehabilitation at Bolago, northern Ethiopian Highlands: Tree cover has much improved since 1868, afforestation started in the late 1980s. (*Photos* courtesy of Jan Nyssen; modified from Nyssen et al. 2009)

livestock farms, producing both for their own subsistence and for the local markets (Zaehringer et al. 2018). The increase of (increasingly irrigated) cropland is associated with a further decline of small forest patches, bush- and shrubland, but also with enlarged forest plantations. At higher elevations, the adoption of agroforestry systems has increased tree cover, while the land cover of protected areas including Mount Kenya National Park and National Forest remained rather stable over the past 30 years (Eckert et al. 2017). It needs to be highlighted that, even though not free from human impact, almost 100% of the afro-alpine zones in East Africa are under various forms of formal protection (Wesche et al. 2008b; Carbutt 2020).

The forest cover on the slopes of Mt. Kilimanjaro is reported to have been much more extensive in the early nineteenth century (Spinage 2012). However, logging and burning have resulted in significant land cover changes over the twentieth century. Land use pressure has increased due to enormous population growth, with the local population having multiplied 20 times since 1895 (Hemp 2005b). Recurrent fires, mainly started by humans, have played an increasingly destructive role in recent decades (Hemp 2006a). Over the past century, Mt. Kilimanjaro has lost some 300 km^2 of high altitude forests, and the upper closed forest line was lowered by 900 m because of fire (Hemp 2006b). Fire frequency is expected to increase with rising temperatures and decreasing precipitation. In addition to the impact of fire, clear-cutting of montane forests reduced the forest area by 450 km^2 since 1929, resulting in a total loss of the nineteenth-century forest cover of c. 50% (Lambrechts et al. 2002; Hemp 2006b). This forest depletion affects the fog water collection and thus the water balance of the whole mountain. Cutting of trees and illegal logging has been reduced after the introduction of stringent bylaws in 2000 (Kilungu et al. 2019). Current land use changes at lower elevations include the increasing transformation of savannah woodlands into maize fields, the emergence of commercial coffee plantations within the altitudinal zone of the traditional agricultural system of the local Chagga people (Chagga home gardens and grasslands), and the enlargement of forest plantations (Hemp 2006b; Ensslin et al. 2015). Cultivation has expanded to more marginal land down the slopes, associated with the disappearance and extreme fragmentation of bushland and appearance and expansion of settlements (Soini 2005). Logging is insignificant in the upper forest zone, and above the forest belt, grazing and agriculture are non-existent. However, the Kilimanjaro National Park attracts an increasing number of visitors each year, generating increasing human impact on the sensitive alpine zone. Nevertheless, the development of ecotourism is a promising economic alternative for the poverty-stricken, rapidly growing population (Agrawala et al. 2003). While non-agricultural activities and paid employment are becoming increasingly important, considerable entry barriers to remunerable off-farm jobs persist for many households, restricting access to attractive non-farm opportunities (Soini 2005). On the other hand, experiences from Mt. Kenya and the Rwenzori Mountains in Uganda show that alpine tourism has so far failed to meet up to expectations in terms of economic benefits and the promotion of sustainable development, even though a stabilizing effect on the livelihood of rural households is discernible (Neuburger and Steinicke 2012). After the establishment of the Rwenzori Mountains National Park in 1992 (Fig. 1.65), land use restrictions directed the population pressure to the foothills, causing there high population density, unsustainable resource use, and social tensions with adjacent ethnic groups (Steinicke 2011). Poor households in the sub-counties bordering the national park still exhibit a great dependence on forest resources inside the park, which are illegally collected and have a significant impact on reducing income inequalities and making the poor less poor. In order to protect the park, encouraging a pro-poor conservation approach rather than increased law enforcement is required (Tumusiime et al. 2011). In southern Africa, prolonged grazing pressure, originating from prevailing extensive livestock farming mainly practised by commercial farmers, has accelerated soil erosion processes in the Drakensberg mountain region. Lesotho was considered one of the most severely eroded countries in Africa, commonly attributed to overstocking and overgrazing of cattle and sheep on communal lands (Acocks 1988). However, the landscapes of the Drakensberg region have been shaped by multiple factors including legislated disenfranchisement and territorial segregation since the 1800s (Salomon et al. 2012). The highland grasslands have been to some extent converted to cultivation and plantations, while especially the Lesotho Highland basalt grassland is heavily utilized for grazing and subject to severe erosion (Mucina et al. 2006; Brown and du Preez 2020). Recent conservation initiatives including the important Maloti-Drakensberg Transfrontier Park should be accompanied by promoting off-reserve conservation on privately or communally owned land.

Fig. 1.65 The establishment of protected areas in the East African mountain system in recent decades such as the Rwenzori National Park (in 1992) in Uganda has increased land use pressure in the surrounding foothills. (*Photo* © Udo Schickhoff, February 12, 2019)

1.4 Conclusions

An unprecedented dimension of change in the world's mountains is obvious from this review, triggered by global climate change and economic globalization. This novel dimension of change is increasingly well documented in relevant publications (see the comprehensive list of recent references), that allow to identify globally significant trends and processes of transformation, but also regional variations. The dramatic change in magnitude and rate of cryospheric and biotic responses and the rapid pace of implementing adaptation strategies in response to changing socio-economic frame conditions completes the overall picture known as the Great Acceleration which describes accelerating Earth system trends in the Anthropocene. Elevational zones in mountains of the world are experiencing strong levels of temperature increase in the frame of anthropogenic climate change, causing cascading effects on physical, biological, and human systems that, in turn, trigger feedbacks to the climate system. Pervasive cryosphere changes including glacier retreat, snow cover decline, and permafrost degradation increase natural hazard risks, and affect seasonal water supply in river systems, with potentially severe implications for agriculture, hydropower generation, and local water resources availability. Declining water supply from mountains will threaten livelihoods and food security of millions of lowland people, in particular in South and East Asia, and may lead to conflicts over water resources. Biotic responses to climate change such as phenological shifts, changing species distributions, invasion of non-native species, and changes in primary production will modify species composition of communities and thus structure and functioning of ecosystems, affecting the provision of ecosystem services for millions of people in downstream areas. Given the low capacity of alpine plant and animal species to adapt to novel climatic conditions, it must be assumed that loss of species, biodiversity decline, and impairment of ecosystem services will be inevitable. Human systems in the world's mountains are passing through a process of implementing adaptations to an increasing magnitude of impact from climate change and globalization processes. Conforming to the heterogeneity of poverty and marginalization levels within and between mountain regions in the Global South, in emerging markets and in industrialized countries, a wide spectrum of adaptations and responses depending on socio-economic conditions, political guidelines, and environmental changes is discernible. Transformations in mountain agriculture, extractive industries, tourism and other sectors are reflected in land use/land cover changes. In the majority of examined mountain systems in this review, current transformations provide the chance to counter the downward spiral of resource degradation, rural poverty, and livelihood insecurity. From an ecological point of view, the recent trend of reduced land use intensity in alpine zones and of the increase and enlargement of protected areas in mountain regions offers the chance for ecosystem recovery and more efficient biodiversity conservation. However, establishing land use systems in high mountain regions which safeguard livelihood and ecological sustainability remains a considerable task. It needs to be embedded in the overriding priority of mitigating adverse effects of drivers of environmental and socio-economic change in the world's mountains. In order to accelerate the implementation of the UN Sustainable Development Goals, the recognition of the global significance of mountain regions needs to be further consolidated and disseminated.

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