Chapter 2: Terrestrial and Freshwater Ecosystems and their Services

Coordinating Lead Authors: Camille Parmesan (USA/UK,France), Yongyut Trisurat (Thailand)

Lead Authors: Rita Adrian (Germany), Gusti Zakaria Anshari (Indonesia), Almut Arneth (Germany), Qingzhu Gao (China), Patrick Gonzalez (USA), Rebecca Harris (Australia), Mike Morecroft (UK), Jeff Price (UK), Nicola Stevens (South Africa), Gautam Hirak Talukdar (India)

Contributing Authors: Vanessa Bremerich (Germany), Sami Domisch (Germany), Simone Langhans (Switzerland), Julio C. Postigo (Peru/USA), Tomas Remenyi (Australia), Michael C. Singer (UK/France),

Review Editors: Carol Billini Franco (Dominican Republic), Yakiv Didukh (Ukraine), Andreas Fischlin (Switzerland)

Chapter Scientists: Julio C. Postigo (Peru/USA), Stavana Strutz (USA)

Date of Draft: 18 October 2019

Notes: TSU Compiled Version

Table of Contents

Executive Summary

Chapter 2: *Terrestrial and freshwater ecosystems and their services* provides an assessments of observed and projected impacts have been made across regions, species and whole biological systems (ecosystems), highlighting processes emerging on a global scale. Where sufficient evidence exists, differences in biologial responses among regions, taxonomic groups or types of ecosystems are presented, particularly when such differences provide meaningful insights into current or potential future autonomous or human-mediated adaptations. Human interventions that might build resilience of ecosystems and minimize negative impacts of climate change on biodiversity and ecosystem functioning are assessed. Such interventions include adaptation strategies and programmes, adaptation for biodiversity conservation, ecosystem-based adaptation, and climate resilient pathways. The assessments were done in the context of the Convention on Biologial Diversity (CBD) Aichi targets and Sustainable Development Goals (SDGs). This chapter highlights both success and failure of human interventions. Meanwhile, knowledge gaps and research priorities were included to encourage additional solution measures and fulfill the knowledge gaps.

For species with long-term records and in relatively undisturbed habitats, about half of species have shifted their ranges polewards and upwards and about 2/3 of species have advanced their spring timing, with these changes attributed to recent climate change *(very high confidence[1]).* These biological changes are consistent with expectations from regional or global warming processes. Coupled with an understanding of underlying mechanisms, coherence of patterns at both regional and global scales, and consistency with model projections of past and future trends, **this body of studies forms multiple lines of evidence making it *very likely*[2] that many observed range shifts and phenological changes can be attributed to regional and global climate changes *(very high confidence).*** {2.4.2; 2.4.5}

Responses in freshwater species are consistent with responses in terrestrial species, including poleward and upward ranges shifts, earlier timing of spring plankton development, earlier spawning in fish, and extension of the growing season. Observed changes in freshwater species are strongly related to anthropogenic climate change (ACC)-driven changes in the physical environment (e.g. increased water temperature and reduced ice cover). Rivers and lakes have warmed by 0.01 to 0.45°C decade^{-1}, starting in the 1970s. Ice coverage has reduced in length by >2 weeks. Projections indicate that 4.6% of historically ice-covered lakes in the northern hemisphere could experience intermittent loss of winter ice cover under +3.2°C warming. Prolonged thermal stratification, which has also been linked to ACC, has led to divergent responses in lakes, with already eutrophic lakes becoming more eutrophic (*very high confidence*) and nutrient-poor lakes becoming more nutrient limited *(medium confidence)*. In boreal coniferous forest, there has been an increase in terrestrial derived dissolved organic matter transported into rivers and lakes as a consequence of climate change (that has induced increases in run-off and greening of the northern hemisphere), as well as to changes in forestry practices. This has caused waters to become brown and more opaque, with complex positive and negative repercussions on water temperature profiles (lower vs upper water) and on primary production *(high confidence*). {2.3.1; 2.4.2; 2.4.4.1}

New studies since AR5 document more complex responses than in prior reports, and indicate that previous estimates of climate change impacts on wild species based on simple hypotheses have underestimated proportion of species responding to climate change. More complex analyses have documented cases of winter warming driving delayed spring timing of northern temperate species due to chilling requirements, and increased precipitation driving species' range shifts downward, eastward and westward in arid regions *(high confidence)*. New studies have shown that phenological changes have, in some cases, successfully compensated for local climate change and reduced degree of range shifts *(medium*

---

[1] FOOTNOTE: In this Report, the following summary terms are used to describe the available evidence: limited, medium, or robust; and for the degree of agreement: low, medium, or high. A level of confidence is expressed using five qualifiers: very low, low, medium, high, and very high, and typeset in italics, e.g., *medium confidence*. For a given evidence and agreement statement, different confidence levels can be assigned, but increasing levels of evidence and degrees of agreement are correlated with increasing confidence.

[2] FOOTNOTE: In this Report, the following terms have been used to indicate the assessed likelihood of an outcome or a result: virtually certain 99–100% probability, very likely 90–100%, likely 66–100%, about as likely as not 33– 66%, unlikely 0–33%, very unlikely 0–10%, exceptionally unlikely 0–1%. Additional terms (extremely likely: 95– 100%, more likely than not >50–100%, and extremely unlikely 0–5%) may also be used when appropriate. Assessed likelihood is typeset in italics, e.g., *very likely*.

confidence). Limited number of studies of this type make it difficult to estimate the generality of these effects globally. {2.4.2.3}

There are a growing number of studies documenting genetic evolution within populations in response to the new selection pressures that recent climate change has imposed. To date, observed genetic changes remain within the boundaries of known genetic variation for that species. Conclusions on adaptive potential derived from observed responses coupled with experimental results and underlying evolutionary and genetic theory are that it is *very unlikely* that evolution would prevent a species being driven extinct in the event its climate space disappears globally *(high confidence)*. {2.4.2.6}

Field research since the IPCC Fifth Assessment Report has detected biome shifts at numerous sites, poleward and upslope, that are consistent with increased temperatures and altered precipitation patterns driven by climate change, and support prior studies that attributed such shifts to anthropogenic climate change *(high confidence).* These new studies help fill prior geographic and habitat gaps, for example documenting upward shifts in the forest/alpine tundra ecotone in the Andes, Tibet and Nepal, and northward shifts in the deciduous/boreal forest ecotones in Canada. Globally, woody encroachment into open areas (grasslands, arid regions and tundra) is *likely* being driven by climate change and increased CO_2 in concert with changes in grazing and fire regime *(medium confidence)*. {2.4.3}

Globally, increases in temperature, aridity and drought have increased the length of fire seasons and doubled potentially burnable area *(medium confidence)*. Increases in area burned by fire have been attributed to anthropogenic climate change (ACC) in North America *(high confidence).* In parts of Africa, Asia, Australia, and South America, area burned has increased, consistent with anthropogenic climate change, but rigorous attribution studies have not been conducted. Human activities that cause or suppress fires can dominate the influence of ACC, increasing the difficulty of attribution of changes. Areas with the greatest increases in fire season length include the Amazon, western North America, Iran, and the Horn of Africa. Increased wildfire under continued climate change could increase risks to water supplies in combination with deforestation that increases sediment flows *(medium confidence)* {2.4.4.2; 2.5.5.2}

Continued climate change under high emissions scenarios could increase future wildfire frequency on one-third to two-thirds of global land by 2100 and decrease fire frequency on one-fifth of global land, with a net global fire frequency increase of ~30% per century *(medium confidence).* For ecosystems with historically low or no wildfire in which models project increased fire risk under climate change, including tropical rainforests, increased drought under continued climate change increases the risk of biome shifts; e.g., potentially leading to conversion of over half the area of Amazon rainforest to grassland. *(medium confidence)*. {2.4.4.2; 2.5.5.2}

Anthropogenic climate change has driven increased tree mortality directly through increased aridity or drought or indirectly through increased wildfire and insect pests *(limited evidence, medium confidence).* In the northern hemisphere, tree-feeding moths and bark beetles have shifted poleward and upslope and moths have become pests of new tree species. Warming winters and longer growing seasons in temperate and boreal forests have driven large increases in insect pest infestations that are causing greater tree damage than in the past *(very high confidence)*. {2.4.4.3; 2.5.5.3}

Terrestrial ecosystems protect globally critical stocks of carbon and provide an essential service of sequestration of carbon from the atmosphere but are at risk of carbon losses from deforestation and climate change (*high confidence*). Approximately four-fifths of carbon emissions from ecosystems comes from tropical deforestation, with most of the remainder from conversion of peatlands for human uses. In the Arctic, increased temperatures have melted permafrost at numerous sites, some areas are drying, and wildfire burns have affected some areas, resulting in soil carbon emission *(high confidence)*. There is growing evidence that some freshwaters are hotspots for carbon transformation, sources of CO_2 and CH_4 and, in some cases, carbon sinks. {2.4.4.4; 2.5.1}

Extreme weather events such as heatwaves, droughts, floods, storms, and tropical cyclones can have greater consequences on natural systems than changes in climate means. Extreme weather events have ***very likely*** become more frequent and more intense in many regions of the world as a result of anthropogenic climate change. In many cases, the magnitude of the transient changes during extreme events exceeds that of projected mean changes by 2100, even under high emissions scenarios (i.e. high increase in mean temperature and mean precipitation). Extreme events have resulted in significant impacts on marine, terrestrial and freshwater ecosystems around the world *(very high confidence)*. {2.3; Cross-Chapter Box EXTREMES in this Chapter}

Increased temperatures and changes to rainfall and runoff patterns; greater variability in temperature, rainfall, river flow and water levels; rising sea-levels and increased frequency of extreme events have led to greater areas of the world being exposed to climate hazards outside those to which species in those habitats are adapted *(high confidence).* It is *likely* that more frequent and intense extreme events, superimposed on longer-term climate trends, is pushing ecosystems to tipping points, beyond which abrupt and possibly irreversible changes are occurring. {2.3}

Percentages of species projected to suffer extinction vary from zero to 54% with a threshold for extinction of >80% of the species' climate space disappeared. With a threshold for extinction of >50% climatic range lost, under 3.2 °C warming, 49% of insects, 44% of plants, and 26% of vertebrates are projected to be at risk of extinction. At 2°C, this falls to 18% of insects, 16% of plants, and 8% of vertebrates and at 1.5°C, to 6% of insects, 8% of plants, and 4% of vertebrates. Differences in estimates of extinction risk stemmed from differing assumptions of thresholds for extinction risk and differing emissions scenarios, as well as from differing geographic regions and taxonomic groups, and differing modeling approaches. {2.5.3.3}

"Novel ecosystems" are *likely* to be increasingly common in the future, in which abiotic conditions with no current analog are expected to drive new combinations of species, and hence communities, with no current, and possibly no historical, equivalents. This is *likely* to have consequences for the structure, function and resilience of ecosystems, and the provision of ecosystem services. {2.3; 2.5.4}

There is evidence of autonomous adaptation by species which reduce their vulnerability to climate change *(medium evidence, medium agreement)*, although this is *very unlikely* to be sufficient to prevent global and local extinctions, particularly under high emissions scenarios. Genetic evolution has potential for helping local populations not already near their range boundaries to adapt to local climate change if that species contains sufficient genetic variation in relevant traits {~~2.3, 2.4.xx~~}. There is also recent evidence that plasticity, for example changes in phenology or behavioural changes that move an individual into cooler micro-climates, may allow species to persist *in situ* for longer than might be expected from local climatic changes. {2.4.2.6; 2.6.1}

There is increasing evidence that degradation and fragmentation of ecosystems exacerbates impacts of climate change *(medium evidence, high agreement)* and ecosystem restoration to build resilience is an important element of adaptation strategies. Extensive research in prior reports by the International Program for Biodiversity and Ecosystem Service (IPBES), and by the IPCC Special Report on Climate Change and Land (SRCCL) has compared the impacts of climate change on ecosystems with those of other drivers (e.g. land use change). Whilst climate change has not been the predominant influence to date, its relative impact is increasing in some systems. Whilst resilience can be enhanced by restoration, it cannot prevent all impacts of climate change, making it increasingly of concern for conservation planning and management and many adaptation plans reflect this. {2.2; 2.6.2}

Since AR5, a large number of adaptation plans and strategies have been developed to protect ecosystems and deliver ecosystem based adaptation for people *(medium evidence, high agreement).* There is much less evidence for deployment of adaptation measures in practice, although ecosystem based adaptation measures to reduce fluvial and coastal flood risk are starting to be deployed in a range of countries. There is little evidence of monitoring and evaluation of the effectiveness of many adaptation actions. This is an important evidence gap that needs to be addressed to ensure a baseline is available against which to judge effectiveness and develop and refine adaptation in future. {2.6.2; 2.6.3; 2.6.4; 2.6.5}

There is emerging evidence *(medium – high evidence)* that some land based mitigation approaches (e.g., Bioenergy with Carbon Capture and Storage (BECCS), plantation forestry) present a risk of maladaptation and a threat to natural and semi-natural ecosystems, especially if deployed at large scale. {Cross-Chapter Box MITIG in this Chapter}. Creating forests in non-forest biomes can be damaging to species, water resources and other ecosystem services. These risks of can be minimised by careful selection of locations, taking account of the natural vegetation of the region, the effects on biodiversity and the range of other ecosystem services desired by people, including food supply. Restoring natural and semi-natural ecosystems, such as forests, in the right locations can make a critical contribution to mitigation, adaptation, biodiversity conservation and Sustainable Development Goals.

Research gaps highlighted in prior reports have begun to be filled, but gaps still remain. The predominance of research on biodiversity and climate change in Europe and North America highlights a gap in the geographic coverage of research. Potentially substantial climate change risks in tropical forests, tropical mountains and freshwater systems suggest the need for greater coverage of these regions and ecosystems in future research. The emergence of the use of paleoecological information to inform contemporary biodiversity conservation under anthropogenic climate change suggest the need for further interdisciplinary research to fill this disciplinary knowledge gap. There is a need for more studies assessing the impacts of climate change on ecosystem services. Cultural services have been particularly under-represented. A key research question is understanding the mechanisms of change for multiple drivers as it is clear that there are strong linkages between the combined impacts of changes in climate, changes in land and water use, and resulting changes in biodiversity on the future supply of ecosystem services. Assessing the effectiveness of adaptation measures is essential, this requires effective monitoring of outcomes of management interventions as well as impacts; it is also important to develop stronger understanding of the links between human behaviour and adaptation outcomes in ecosystems. {2.7}

2.1 Introduction

Chapter 2: *Terrestrial and freshwater ecosystems and their services* provides an assessments of observed and projected impacts have been made across regions, species and whole biological systems (ecosystems), highlighting processes emerging on a global scale. Where sufficient evidence exists, differences in biologial responses among regions, taxonomic groups or types of ecosystems are presented, particularly when such differences provide meaningful insights into current or potential future autonomous or human-mediated adaptations. Human interventions that might build resilience of ecosystems and minimize negative impacts of climate change on biodiversity and ecosystem functioning are assessed. Such interventions include adaptation strategies and programmes, adaptation for biodiversity conservation, ecosystem-based adaptation, and climate resilient pathways. The assessments were done in the context of the Convention on Biologial Diversity (CBD) Aichi targets and Sustainable Development Goals (SDGs). This chapter highlights both success and failure of human interventions. Meanwhile, knowledge gaps and research priorities were included to encourage additional solution measures and fulfil the knowledge gaps.

The Working Group II Summary for Policy Makers of the 5th Assessment Report (WGII AR5 SPM) stated that "many terrestrial and freshwater species have shifted their geographic ranges, seasonal activities, migration patterns, abundances, and species interactions in response to ongoing climate change" (IPCC 2014). Based on long-term observed changes across the regions, they estimated that approximately 20–30% of the plant and animal species are at risk of extinction when global mean temperatures rise 2-3°C above preindustrial levels (Fischlin et al., 2007). In addition, WGII AR5 (IPCC, 2014) broadly suggested autonomous adaptation by ecosystems and wild species might occur, and proposed human-assisted adaptation to minimize negative climate change impacts.

Chapter 2: *Terrestrial and freshwater ecosystems and their services* has been assessed primarily based on recent scientific findings (after 2014), but includes prior literature when it was either missing in previous reports (AR4 and AR5), or when it serves to build a coherent body of evidence for developing confidence assessments. In addition, the recently published IPCC Special Report on 1.5°C warming (SR 1.5), the IPBES/Global Assessment and earlier regional assessments, and IPCC SRCCL were incorporated in this chapter. Recent climate metrics from WG1 climate Atlas of Global and Regional Climate Projections were collated to determine climate hazards on key vegetation communities, habitat ranges, biomes and ecosystem services.

The risk assessments for species, communities, key ecosystems and their services were based on the Risk Assessment Framework introduced in the IPCC AR5 (IPCC, 2014). Assessments of observed changes in biological systems emphasizes detection and attribution of climate change on evolutionary processes, freshwater ecosystems and wetlands, and wildfire and ecosystem services, which were inadequately assessed in previous reports. Where appropriate, assessment of interactions between climate change and other human activities is provided.

2.2 Points of Departure

2.2.1 Drivers of Impacts on Biodiversity, Ecosystem Functioning and Ecosystem Services

The main drivers of impacts on biodiversity, terrestrial and freshwater ecosystems functioning and ecosystem services have been assessed thoroughly in recent years, e.g., in the previous IPCC reports, reports of the FAO, IPBES (global and regional assessments), and the Global Environmental Outlook (Settele et al., 2014; FAO, 2018; IPBES, 2018a; IPBES, 2018b; IPBES, 2018c; IPBES, 2018d; IPBES, 2019; UNEP, 2019). Most recently, the IPCC Special Report on Climate Change and Land (SRCCL) has provided an assessment on land degradation and desertification, greenhouse gas emissions and food security in the context of global warming (IPCC, 2019).

Land-use and land cover change (LULCC), and the over-exploitation of resources from terrestrial and freshwater systems continues to be a major factor of natural ecosystem and biodiversity loss, especially in tropical forests, savannahs, rivers and lakes. [PLACEHOLDER FOR SECOND ORDER DRAFT: confidence statement and more concrete referencing to IPBES regional and global assessments and upcoming SRCCL and SROCC will be added]. In many regions of the world, fertiliser input, pollution of waterways, dam construction and the extraction of freshwater for irrigation put additional pressure on biodiversity and alter ecosystem function (IPBES, 2019). Likewise, for biodiversity, invasive alien species have been identified as a major threat, especially in freshwater systems, islands and coastal regions [PLACEHOLDER FOR SECOND ORDER DRAFT: confidence statement will be added] (IPBES, 2018a; IPBES, 2018d; IPBES, 2018b; IPBES, 2018c; IPBES, 2019). Increasing rates of urbanisation and built-up environment, together with the loss of fertile cropland area associated with urban expansion, has been identified as an additional area of concern both for biodiversity and the loss of ecosystem services (d'Amour et al., 2017; van Vliet et al., 2017). This issue received little attention in AR5, but has been recognised in the SRCCL [[PLACEHOLDER FOR SECOND ORDER DRAFT: citation to Chapter 5: Food Security in SRCCL will be added].

Climate change has been recognised as a considerable driver of change, especially in high latitude and high altitude terrestrial and freshwater systems in prior reports (Settele et al., 2014; IPCC, 2018) [[PLACEHOLDER FOR SECOND ORDER DRAFT: confidence statement to be added]. Warming temperatures have been attributed as causes of poleward and upward ranges shifts, advanced spring timing, changes in vegetation and soil microbial activity, prolongation of terrestrial and aquatic growing seasons, changes in carbon fluxes and changes specific to freshwater systems including increased water temperature, decreased lake-ice, increased thermal stratification, wetland drying. CO_2 is thought to contribute to the observed enhanced shrub encroachment in tropical savannahs, while current and potential future CO_2 impacts on freshwater systems also have started to gain attention (Donohue et al., 2013; Hasler et al., 2016; Stevens et al., 2016). Climate change and CO_2 are expected to become increasingly important as drivers of change over the coming decades (Ciais et al., 2013; Settele et al., 2014; IPBES, 2019; IPCC, 2019). For all drivers, and associated impacts, the treatment of freshwater systems has been relatively limited in previous assessments. In particular, the interlinkages between terrestrial and freshwater processes have not been fully explored (Settele et al., 2014; IPBES, 2019).

2.2.2 Interactions between Nature-Based Adaptation Measures and Mitigation

As demonstrated in the IPCC Special Report on 1.5°C warming (SR 1.5), each half degree of warming has large, negative impacts on biodiversity and the provision of many ecosystem services (Hoegh-Guldberg et al., 2018). In AR5, it had already been suggested that anthropogenic emission trajectories that correspond to a warming of circa 2°C (RCP 2.6) indicate a need for large-scale land based mitigation options, including CO_2 uptake in afforestation and reforestation measures (AR), and the growth of bioenergy (BE) crops (Settele et al., 2014; van Vuuren and Carter, 2014). The SR 1.5°C emphasises the essential role of land in limiting warming to as close to 1.5 °C as possible and well under 2°C (Rogelj et al., 2018).

Large-scale mitigation efforts that rely on land have been shown to challenge the achievement of multiple sustainable policy goals (Creutzig et al., 2015; Boysen et al., 2016; Fuss et al., 2018; Hof et al., 2018; Smith et al., 2019). These challenges arise from the large area requirements of Afforestation/Reforestation (AR) and Bioenergy crops (BECs), which can conflict with other land uses such as food and timber production or conservation. Land-management intensification, including increased fertiliser and irrigation, and increased freshwater exploitation also play a role (see e.g., (Hoegh-Guldberg et al., 2018; IPBES, 2019; IPCC, 2019; Seddon et al., 2019). Large scale climate change mitigation measures of this sort can be either positive or negative for terrestrial and freshwater ecosystems, depending what measures are adopted and where they are located. As such, the assessment of climate-change impacts on ecosystems is expected to increasingly include both direct and indirect aspects.

Beyond AR and BECs, the SRCCL has begun to analyse the prospects of applying a larger portfolio of different mitigation measures (Griscom et al., 2017; Smith et al., 2019). These measures have high potential for adaptation-mitigation co-benefits since sustainable management practices in ecosystems jointly with restoration efforts have been shown to be effective in adapting to impacts of climate change (IPCC, 2012). A number of ecosystem-based adaptation measures, such as restoration of forests and wetlands for flood and erosion control, are critical for maintaining and enhancing freshwater supply and quality. This emphasises again the value of treating the interlinkages between terrestrial and freshwater systems more explicitly than has been done in previous reports. However, since many potential mitigation measures remain to be implemented (Arneth et al., 2019; Smith et al., 2019) their interactions and potential co-benefits with adaptation have been as yet difficult to assess.

2.2.3 Extreme Events and their Importance for Impacts and Adaptation

Some extreme weather and associated episodic events such as floods, droughts, wind throws, insect outbreaks or fire have likely increased over recent decades, and are projected to occur with even more frequency in future as the climate warms further [PLACEHOLDER FOR SECOND ORDER DRAFT: confidence statement to be added] (IPCC, 2012). These disturbances affect ecosystem functioning and biodiversity but are still poorly captured in impact models, in particular, in cases where these interact such as the complex interactions among drought, fire and insect outbreaks (Allen et al., 2010; Anderegg et al., 2015), and hence are difficult to quantify in future projections (Handmer et al., 2012). As emphasised in the IPCC Special Reports on Extremes, these extreme events also pose large challenges for ecosystem-based adaptation. Ecosystem functionality that is used in such adaptation measures ("nature-based solutions") may be altered or destroyed by extreme episodic events (Handmer et al., 2012; Lal et al., 2012). [Cross-Chapter Box EXTREMES in this Chapter]

2.2.4 Ecological (Dis)equilibria

Settele (2014) in AR5 recapitulated that even at relatively undisturbed environmental conditions (e.g., in the absence of changing human influence) ecosystems are not in a static equilibrium with their environment. Episodic events such as droughts, pathogen outbreaks, floods or fires are an integral part of internal system dynamics. The combined degree and velocity of human-caused climate change may result in extreme events that have not been observed in the past (Reu et al., 2014; Ordonez et al., 2016). Previous reports have highlighted the possibility of resulting new ecosystem states stemming from shifts in thermal regimes, species composition, and energy and matter flows (Settele et al., 2014; Shin et al., 2019). Projecting such "tipping points" (see glossary) has been identified in previous reports as a challenge since neither monitoring programmes nor field studies, nor ecosystem and biodiversity modelling tools capture the underlying species-species and species-climate interactions sufficiently well to identify how biological interactions within and across trophic levels may amplify or dampen shifts in ecosystem states (Settele et al., 2014; Shin et al., 2019). Paleo and historical records show that flora and fauna have capability to adapt to natural climate change within bounds, while human societies may have not experienced the need to adapt to catastrophic climate change (see Cross-Chapter Box PALEO in Chapter 3)

Building on these previous analyses, and recent literature, chapter 2 in this AR6 provides new insights compared to previous assessments by (i) emphasising freshwater aspects, and the interlinkages between freshwater and terrestrial systems, (ii) assessing more clearly the link between biodiversity and ecosystem functioning, (iii) assessing impacts associated with climate change mitigation scenarios versus impacts of climate change, including interactions with adaptation, (iv) where possible, places findings in context of the United Nations Sustainable Development Goals (SDGs) 2030.

2.3 Hazards and Exposure

The rationale for including physical changes here is that they are fundamental drivers of all levels of biological organization, from individual species to communities to whole ecosystem characteristics. Many hazards specific to biological systems, particularly of freshwater systems, are not documented elsewhere in WGI or WGII of this AR.

The major climate hazards at the global level are generally well understood (WGI AR6 and Atlas). Increased temperatures and changes to rainfall and runoff patterns; greater variability in temperature, rainfall, river flow and water levels; rising sea-levels and increased frequency of extreme events means that greater areas of the world are being exposed to climate hazards outside those to which they are adapted (Figure 2.1; Figure 2.2; Cross-Chapter Box EXTREMES in this Chapter).

Many organisms have adapted to cope with long-term and short-term climate variability, particularly such variability as has been present during their evolutionary history, but as trends in climate change have increased, the magnitude and frequency of extreme events have also increased. An extreme weather event is an event that is rare at a particular place and time of year, often defined as being rarer than the 10th or 90th percentile of a probability density function estimated from observations (see glossary). It is *likely* that the combination of internal variability, superimposed on longer-term climate trends, is pushing ecosystems to tipping points, beyond which abrupt and possibly irreversible changes are occurring (Harris et al., 2018a; Jones et al., 2018; Hoffmann et al., 2019; Prober et al., 2019). Increases in the frequency and severity of heatwaves, droughts/aridity, floods, fire danger and extreme storms have been observed globally and these trends are projected to continue.

While the major climate hazards at the global level are generally well described with high confidence, there is less understanding about the importance of hazards on ecosystems when they are superimposed on each other. Simultaneous or sequential events (coincident or compounding events) can lead to an extreme event or impact, even if each event is not in themselves extreme. For example, the compounding effects of sea-level rise, extreme coastal high tide, storm surge, and river flow can substantially increase flooding hazard (Moftakhari et al., 2017); (Cross-Chapter Box SLR in Chapter 3). Risk assessments typically only consider a single climate hazard, potentially underestimating risk (Sadegh et al., 2018; Zscheischler et al., 2018).

Full understanding of the effect of changes in the timing of climate hazards and changing variability is still lacking for many terrestrial and freshwater systems. Understanding impacts of the rate of climate change has also been identified as less developed and more uncertain than changes in mean climate. It is generally assumed that the more rapid the rate of change, the greater the impact on species and ecosystems, but responses are taxonomically and geographically idiosyncratic. Although events from the extreme tails of probability distributions are not necessarily extreme in terms of impact, we use a percentile approach to describe extreme high and low temperature and precipitation because organisms are adapted to local levels of climate variability, so the magnitude of the deviation from the mean is likely to have the greatest biological impact (Harris et al., 2018a).

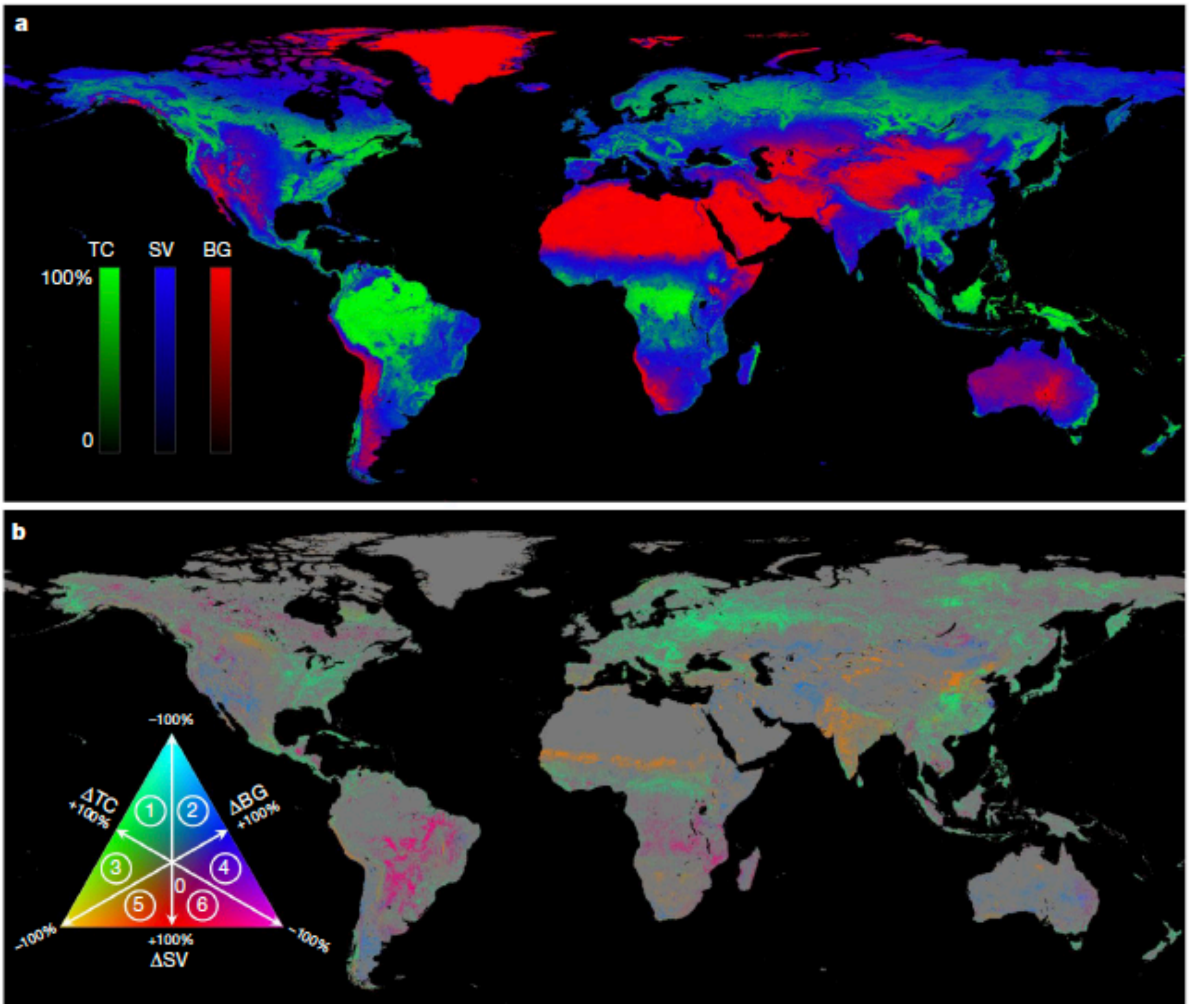
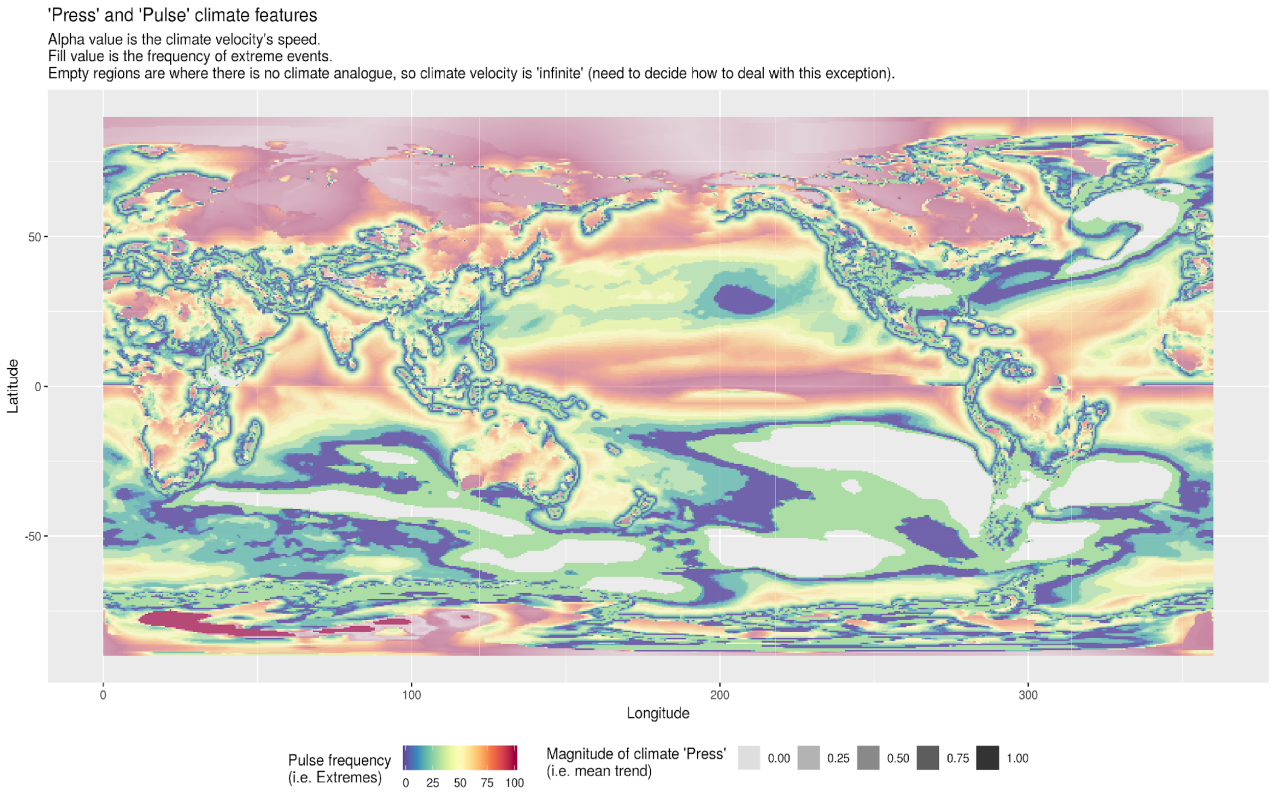


Figure 2.1: [[PLACEHOLDER FOR SECOND ORDER DRAFT: Map of non-climatic global change drivers (pollution, human population, land use changes, invasive species. Taken from (Song et al., 2018): A global map of land use change from 1982-2016. *(from Song 2018, Fig. 1):* "A satellite-based record of global TC, SV and BG cover from 1982 to 2016. a, Mean annual estimates. b, Long-term change estimates. Both mean and change estimates are expressed as per cent of pixel area at 0.05° Å~ 0.05° spatial resolution. Pixels showing a statistically significant trend ($n = 35$, two-sided Mann–Kendall test, $P < 0.05$) in either TC, SV or BG are depicted on the change map. Circled numbers in the colour legend denote dominant change directions: 1, TC gain with SV loss; 2, BG gain with SV loss; 3, TC gain with BG loss; 4, BG gain with TC loss; 5, SV gain with BG loss; and 6, SV gain with TC loss." Figure to be developed to include other drivers as well as land use change]



'Press' and 'Pulse' climate features

Alpha value is the climate velocity's speed.
Fill value is the frequency of extreme events.
Empty regions are where there is no climate analogue, so climate velocity is 'infinite' (need to decide how to deal with this exception).

Figure 2.2: Map of global hazards, showing the magnitude of the climate trend (climate velocity in ℃ km^{-1}) overlaid by the frequency of extreme events. Climate velocity is calculated as the rate of change in mean annual temperature and mean annual aridity (Precipitation – Evaporation) between the historical (1961-1990), current (2001-2020) and end of century (2081-2100) time periods. Extremes indices will include change in duration and intensity of drought; extreme rainfall; fire weather; heatwave duration and extreme low temperature. Figure will be a two-panel figure showing a) the changes we have already experienced (change between the historical (1961-1990) and current (2001-2020) periods, and b) the changes projected for the end of century period (2081-2100) in relation to the current (2001-2020) period. [PLACEHOLDER FOR SECOND ORDER DRAFT: Figure to be updated with metrics on extremes from WGI]

Table 2.1: [PLACEHOLDER FOR SECOND ORDER DRAFT: Table to provide a list of geographic areas with multiple hazards showing large change, along with their ecoregion (maybe group ecoregions here), and possible add projections for IUCN listed species (e.g., from Pacifici, 2015 and 2017)]]

2.3.1 Biologically Important Physical Changes in Freshwater Systems

2.3.1.1 Change in Thermal Habitat

The most coherent climate change-driven responses in freshwaters are related to the physical environment such as water temperature, lake stratification, water level and flowing regime, all affecting ecosystem functionality (Adrian et al., 2009; Adrian and Hessen, 2016). Global estimates of the rates of warming of lake surface waters are 0.21°C decade^{-1} from 1970-2010 (Kraemer et al., 2015), 0.34 and 0.45°C decade^{-1} from 1985-2009 (Schneider and Hook, 2010; O'Reilly et al., 2015). Increasing air temperature is mostly the primary driver of warming water temperature; the variability depends on combinations of seasonal air temperature trends, solar brightening, ice cover and local lake characteristics such as elevation, lake morphology, water clarity, or mixing regime (Kraemer et al., 2015; Winslow et al., 2015; Rose, 2016; Schmidt et al., 2019).

In rivers annual mean water temperature increased in 40 major streams and rivers throughout the US by 0.009-0.077 °Cyr^{-1} (Kaushal et al., 2010), typically correlated with increased air temperatures. However, the thermal regime of rivers is mediated by the overall complexity of the hydrological climate and land-use features of its watershed (flow regime, groundwater, earlier or reduced snow melt, cold water entrainment, dams, shading by vegetation) making the significance and magnitude of changes in river water temperature relative to changes in air temperature inherently site-specific (Bernhardt et al., 2018) (Lisi, 2015; Piccolroaz et al., 2016; Piccolroaz et al., 2018).

A direct response towards global warming has been a prolongation in thermal stratification of lakes. Stratification can trap nutrients below the phototrophic zone, rendering them unavailable for phytoplankton growth. In nutrient poor large lakes where internal nutrient loading via vertical mixing is often the primary nutrient source it leads to a reduction in algal biomass (Kraemer et al., 2017), while global warming reinforces the eutrophication of already eutrophic lakes. This trend is projected to continue with continued climate change (Shatwell et al., 2019). In northern temperate lakes, stratification is beginning earlier in spring and ending later in autumn than previously, resulting in an overall prolongation of stable thermal stratification and a subsequent prolongation of the growing season *(very high confidence)* (Kirillin, 2010; Adrian and Hessen, 2016). Prolonged thermal stratification leads to a decrease in the oxygen concentrations of deep water, causing stress to zooplankton and fish (Domis)(Adrian and Hessen, 2016); *very high confidence*). Chemical processes trigger the release of nutrients previously bound in the sediment. The combination of higher water temperatures, prolonged stratification and high nutrients encourage the growth of algae, thus increasing eutrophication - particularly the development of nuisance cyanobacteria blooms (Domis, 2013; Adrian and Hessen, 2016) (*very high confidence*).

Variability in warming patterns across depth, area, and season causes uncertainty in how thermal habitats in lakes will shift in response to climate change globally complicating projections of how organisms may shift their distributions spatially or seasonally to maintain isothermal conditions in the face of lake warming. Lake organisms will have to redistribute across depths and seasons in order to maintain thermal stasis in a warming climate. As lakes warm, thermal specialists may shift to new environments to which they may not be well-adapted. In some cases, decreases in bulk thermal habitat over specific temperature ranges could also be partially offset by translocating to a different lake.

Besides changes in air temperature, river water temperatures are governed by dimensions of flow direction and complex river network structures. They can be buffered by snow coverage, cold water entrainment or shading. Lower flows will exacerbate stream warming during summer, particularly during heat extremes. For example, Swiss lowland rivers were extremely sensitive to heatwaves while high-altitude snow-fed rivers and regulated rivers receiving cold water from higher altitude showed a damped thermal response (Piccolroaz et al., 2018). Because of these complexities, forecasting climate-driven thermal changes in stream habitats is difficult and inherently site-specific (Ruesch et al., 2012).

2.3.1.2 Changes in Water Level

Between 1984 and 2015 permanent surface water has disappeared from an area of almost 90,000 square kilometres, roughly equivalent to that of Lake Superior, and new permanent bodies of surface water covering 184,000 square kilometres have formed elsewhere (Pekel et al., 2016). All continental regions show a net increase in permanent water, except Oceania, which has a fractional (one percent) net loss. Much of the increase is from reservoir filling, although climate change is also implicated.

Loss is more geographically concentrated than gain. Over 70 per cent of global net permanent water loss occurred in the Middle East and Central Asia, linked to drought and human actions including river diversion or damming and unregulated withdrawal. The once extensive spring areas at Ras al Ain in Northern Syria have almost completely dried out, as has the spring of the Barada River near Damascus, along with almost the entire Damascus hydrological basin (Durwall, 2016). Losses in Australia and the USA linked to long-term droughts are also evident (Pekel et al., 2016). Water level trends in East Asia and Africa were especially variable from lake to lake. Fluctuations as high as 12m follow heavy rainfalls in lakes in the Danau Sentarum National Park, a Ramsar site in Borneo (Giesen and Anshari, 2016).

More than half of the global rivers undergo periodic drying, so increased frequency and intensity of droughts may cause transitions of many perennial rivers into intermittent rivers (Datry et al., 2016), and climate change projections of reduced rainfall in parts of western Asia (Chenoweth et al., 2011) suggest a threat to freshwater fish.

2.3.1.3 Changes in Discharge

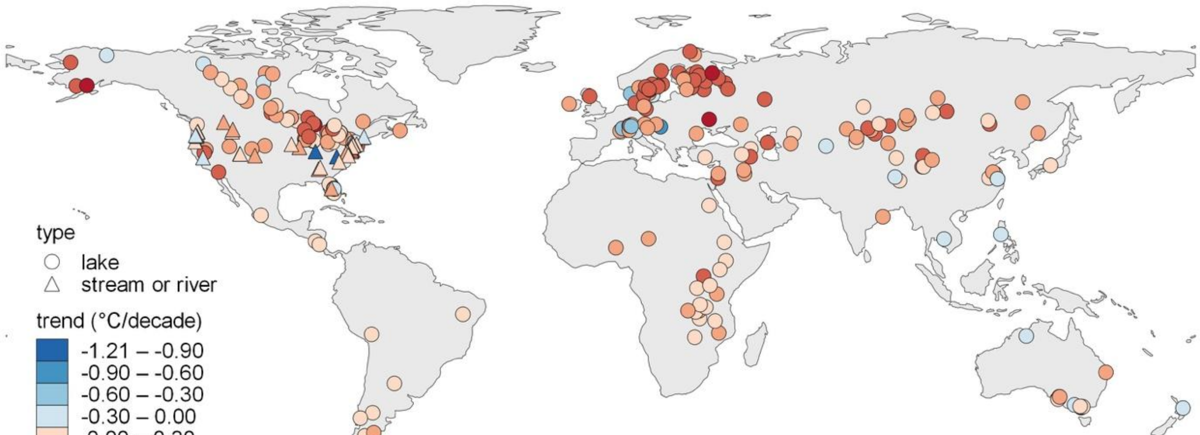
Closely related to changes in water level, reduced snowpack and earlier snowmelt, along with altered precipitation patterns attributed to climate change, have led to changes in the rate and timing of river discharge (Vorosmarty et al., 2010). Patterns in flow regimes can be directly linked to a variety of processes shaping freshwater biodiversity (Bunn and Arthington, 2002), hence any climate-change induced changes on flow regimes are likely to alter species composition (Thomson et al., 2012; Chessman, 2015; Rolls et al., 2016; Kakouei et al., 2018).

2.3.1.4 Loss of Ice

Climate warming has caused a widespread loss of lakes and rivers ice around the Northern hemisphere (Weyhenmeyer et al., 2011; Sharma et al., 2019) (Figure 2.3). Lake surfaces have been freezing later in winter and breaking-up earlier in spring – overall reducing ice duration by >2 weeks and increasing the number of ice-free years (Adrian et al., 2009; Kirillin et al., 2012; Sharma et al., 2019). The impact of climate warming on lake ice phenology will be disproportionately large in areas where winters are relatively mild and the duration of ice cover is already short or intermittent compared to areas where winters are consistently cold and ice cover periods are much longer (Weyhenmeyer et al., 2011; Adrian and Hessen, 2016). With rapidly changing winter conditions in temperate areas (Sharma et al., 2019), a global quantitative synthesis on under ice plankton composition in 101 lakes provided evidence that inter-seasonal connections are common for several plankton-associated and biogeochemical variables, linking processes in winter to both the previous and subsequent summers (Hampton et al., 2017).

Projections revealed that 4.6% of the ice covered lakes in the northern hemisphere could experience intermittent winter ice cover under current mitigation trajectories of +3.2°C (Sharma et al., 2019). With a global rise of 2°C in air temperature, it is projected that the number of lakes with intermittent ice will double, affecting up to 394 million people who live within one hour of the shores. In a worst-case scenario (air temperatures increase of 8°C), 230,400 lakes and 656 million people in 50 countries will be impacted (Sharma et al., 2019). This high degree of warming would likely push the ice-cover zone out of the United States and into northern Canada, as well as threaten the ice cover of lakes in cold Scandinavian nations (Reid et al., 2019). Unfrozen lakes lose more water to evaporation during the winter and warm faster during the spring, which can decrease levels of water and oxygen in the lake. Loss of ice will affect local winter cultural heritage (Sharma et al., 2019).

a)



b)

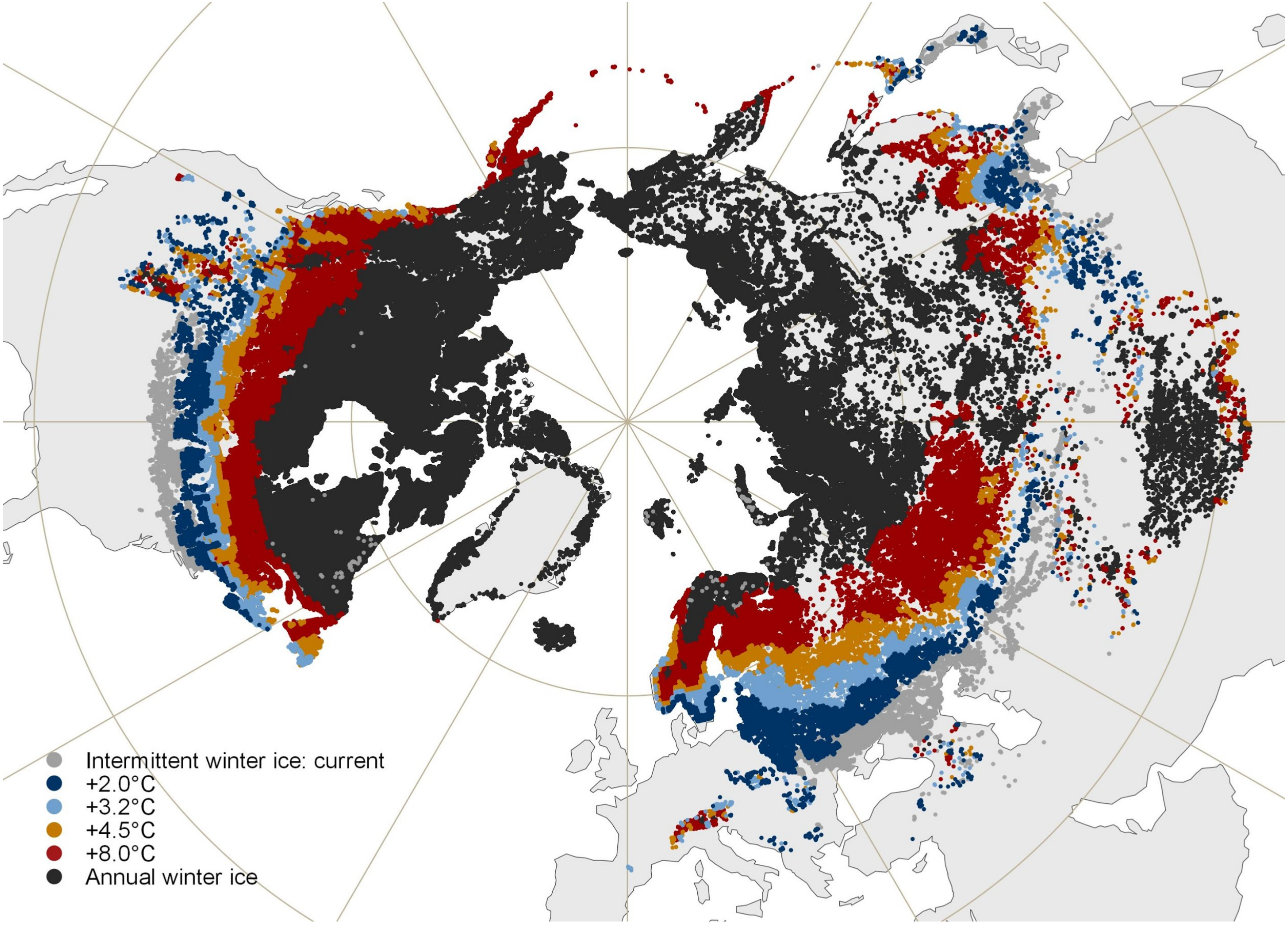


Figure 2.3: [PLACEHOLDER FOR SECOND ORDER DRAFT: Temperature and ice cover trends globally. a) Global trends in lake (O'Reilly et al., 2015) and river water temperature in the past decades (overall period between 1922-2009) and b) spatial distribution of current and future Northern Hemisphere lakes that may experience intermittent winter ice cover with climate warming (Sharma et al., 2019). Figure to be updated into a multi-panel figure adding trends in water temperature, loss of ice cover and changes in water level from WGI]

2.3.1.5 Extreme Weather Events and Freshwater Systems

Besides long term warming trends, we increasingly face weather events that produce extreme wind speeds and heavy rainfall, or heat waves which affect the thermal stratification of lakes, water-sediment interaction, run-off and underwater light conditions, with implications for oxygen conditions and nutrient dynamics. Episodic events of extreme wind speed or rain events have had strong but complex impacts on thermal structure, dissolved organic matter (DOM) loading and underwater photosynthetically active radiation (PAR) levels in lakes and rivers. Depending on lake type, the severity and timing of the extreme event, and the nature of entrainment from run-off and internal loads (e.g. coloured DOM), algal biomass has been either reduced or increased (Jennings et al., 2012; Havens et al., 2016; Kuha et al., 2016; Kasprzak et al., 2017). Not all extreme events will have a biological impact (Bailey and van de Pol, 2016). For instance, an extreme wind event may have little impact on phytoplankton in a lake which was fully mixed prior to the event. Conversely, storm effects on phytoplankton communities may compound when lakes are not yet recovered from a previous storm or if periods of drought alternate with periods of intense precipitation, potentially eroding ecosystem resilience (Leonard, 2014). Hence the timing of storm events, and antecedent conditions, may greatly influence the ecological impact of storms (Perga et al., 2018).

Entrainment of anoxic water from deep water as a result of wind induced changes in mixing, reduction in primary productivity (change in underwater PAR) and increased mineralisation of organic carbon delivered from the catchment have caused decreases in surface dissolved oxygen levels (Jennings et al., 2012). Summer fish kills have been related to summer temperature extremes and near-bottom oxygen depletion (Kangur et al., 2016). Oxygen depletion in the cold deep water body of lakes during heat extremes has forced fish to move upwards into the warm upper water layers (epilimnion) where thermal stress and metabolic costs increase. A once in 250-year flood event in 2009 caused the water column of a large oligotrophic lake to destabilise, followed by reduced primary production (de Eyto et al., 2016). Combined lower rates of gross primary production and higher rates of respiration have the capacity to shift lakes towards an increased degree of heterotrophy (Jennings et al., 2012). Climate change affects the light regime in rivers via longer ice-free periods, earlier leaf out and later leaf fall, increased drought severity, altered timing and severity of floods and earlier or reduced snow melt (Bernhardt et al., 2018). Floods deliver, bury and remove the organic matter stored within river beds (Bernhardt et al., 2018).

2.3.1.6 Projected Changes in Physical Characteristics of Lakes and Rivers

Projections of climate drivers important for freshwater ecosystems have improved since AR5, with assessments of large ensembles of GCM projections across South Asia (Zheng et al., 2018), assessments of hydrological extremes (Giuntoli et al., 2015; Pechlivanidis et al., 2017), and intercomparison projects now assessing future changes in runoff at unprecedented temporal (up to daily) and spatial resolution. The Inter-Sectoral Impact Model Intercomparison Project (ISI-MIP) (Warszawski et al., 2014) integrated nine hydrological models and five Global Climate Models to evaluate impacts and uncertainties in 12 large-scale river basins covering the global range of climate, topography and continental distribution - the Rhine and Tagus in Europe; Niger and Blue Nile in Africa; Ganges, Lena, Upper Yellow and Upper Yangtze in Asia; MacKenzie, Upper Mississippi and Upper Amazon in America and Darling in Australia.

Future trends are difficult to project because of uncertainties in the GCMs, downscaling approaches and hydrological models (Vetter, 2017). Uncertainties in precipitation projections continue to lead to a large spread in projected runoff changes in many regions (Krysanova and Hattermann, 2017; Zheng et al., 2018).

Gridded general circulation models (GCM) outputs (and its dependencies with hydrological models, ISIMIP (https://www.isimip.org/) are widely used, yet they yield uncertainties regarding our understanding as to how future climate change would impact freshwater species and communities. We still lack a comprehensive overview of freshwater habitats worldwide at a relevant spatial grain that is crucial to detect changes in species and community composition. Valuable data sets mapping the distribution have been developed very recently (e.g. rivers up to 60°N latitude (Lehner et al., 2008), lakes with a surface area of at least 10 ha (Messager et al., 2016), and wetlands at a 15 arc-sec spatial resolution (Tootchi et al., 2019). In riverine ecosystems, small headwater streams are estimated to contribute ca. 60-70% to the entire river length globally, harbouring species-rich communities. Yet, we currently lack both the high-resolution patterns of how small water bodies are distributed worldwide (but see e.g. the SWOT mission https://swot.jpl.nasa.gov/, satellite to be launched in 2021), and appropriate downscaled GCM outputs. For instance, (Isaak et al., 2016) showed that small headwater streams respond slower than expected to climate velocities, i.e. they are not warming as fast as expected, providing refugia for biodiversity.

2.3.2 Basic Processes Leading to Compound Events and Feedbacks

Understanding of the large-scale drivers and the local to-regional feedback processes that lead to extreme events is still limited and projections of extremes and coincident or compounding events remain uncertain (Prudhomme et al., 2014; Sillmann et al., 2017; Hao et al., 2018; Miralles et al., 2019). Extreme events are challenging to model because they are by definition rare and often occur at spatial and temporal scales much finer than the resolution of climate models (Sillmann et al., 2017; Zscheischler et al., 2018). Additionally, the processes that cause extreme events often interact, as is the case for drought and heat events, and are spatially and temporally dependent, for example, as is the case in soil moisture and temperature (Vogel et al., 2017). Understanding feedbacks between land and atmosphere also remains limited. For example, positive feedbacks between soil and vegetation, evaporation, radiation and precipitation are important in the preconditioning of extreme events such as heatwaves and droughts (Miralles et al., 2019).

Despite recent improvements in observational studies and climate modelling (Santanello et al., 2015; Stegehuis et al., 2015; PaiMazumder and Done, 2016; Basara and Christian, 2018), the potential to quantify or infer formal causal relationships between multiple drivers and/or hazards remains limited (Zscheischler and Seneviratne, 2017; Miralles et al., 2019). Additionally, hazards such as drought are often exacerbated by societal, industrial and agricultural water demands, requiring more sophisticated modelling of the physical and human systems. Observations of past compound events may not provide reliable guides to how future events may evolve, because human activity and recent climate change continue to interact to influence both system functioning and the climate state (Mehran et al., 2017; Wan et al., 2017).

2.4 Observed Impacts of Climate Change on Wild Biological Systems

2.4.1 Overview and Methodology

For biological systems we use the framework for detection and attribution outlined in AR5 in which attribution of observed biological changes is made not to global, but to local or regional climate changes, regardless of the underlying drivers of these changes (Parmesan et al., 2013; Cramer et al., 2014).

Data in prior reports were highly concentrated from the northern hemisphere. This bias is diminishing. Data from southern hemisphere regions are being added and there is now representation from every continent. However, the critique of "global" studies by Feeley (2017) argues that most of them are far from global and that considerable geographic and taxonomic bias remains.

Freshwater systems are well represented with respect to basic understanding of processes related to physical changes in lakes or streams, but there are relatively few long-term biological records and effects of climate change are often highly confounded with those of other human drivers, particularly eutrophication and invasive species (see Sections 2.1 and 2.3). For terrestrial systems, global meta-analyses in AR3 and AR4 concentrated on findings from relatively undisturbed areas, but more recent global and regional meta-analyses have tended to include all available relevant data (AR5 and later).

For studies conducted in relatively undisturbed areas, confidence in attributing observed changes to climate change is higher than for those done in areas with greater human disturbance, where the effects of local or regional climate change interact with other drivers of change. Overall confidence in attribution of a biological change to climate change can be increased in multiple ways (Parmesan et al., 2013). First, confidence rises when the time span of biological records is long (>50 years), such that long-term trends can not only be statistically distinguished from natural variability, but when decadal trends in climate match decadal trends in a biological response. Second, confidence is increased when the geographic area covered is large, which tends to diminish the effects of local confounding factors. Third, confidence is increased when there is a large body of experimental or empirical evidence of a significant mechanistic link between particular climate metrics and a biological response. Fourth, confidence is increased when particular fingerprints of climate change are documented that uniquely implicate climate change as the causal driver of the biological change (Parmesan and Yohe, 2003). These conditions constitute multiple lines of evidence which, when they converge, can provide *very high confidence* that climate change is the causal driver of an observed change in a particular biological species or system (Parmesan et al., 2013).

2.4.2 Observed Responses by Species and Communities (Freshwater and Terrestrial)

By the time of AR5, >4,000 species with long-term observational data had been studied in the context of climate change (Parmesan, 2006; Parmesan and Hanley, 2015). Since then, hundreds of new studies have been added (Scheffers et al., 2016). Global and regional meta-analyses of diverse systems, habitats and taxonomic groupings document that approximately half of all species with long-term records have shifted their ranges poleward and/or upward in elevation and ~2/3 have advanced their timing of spring events (phenology) (Table 2.2; (Parmesan and Hanley, 2015; Parmesan, 2019). Changes in abundance tend to match predictions from climate warming, with warm-adapted species significantly out-performing cold-adapted species in the same habitats (Bowler et al., 2017; Hughes et al., 2018). These meta-analyses each document a link between recent climate change and observed changes in natural biological systems. Attribution of changes in streams and lakes is difficult, as few systems are in undisturbed states. As with terrestrial ecosystems, changes in land-use and other human activities can interact with climate change, confounding interpretation of observed changes. Moreover, climate change-induced indirect temperature effects through changes in thermal, light and nutrient regimes along with an increase in organic matter transport via the catchment (browning) are often more pronounced than responses directly related to higher temperatures. Below, we give examples concentrating on freshwater systems. These have been under-represented in most reviews (Table 2.2), but long-term data sets, coupled with laboratory and field experiments, are now documenting the importance of the physical/hydrological environment in structuring lake and stream biogeochemistry and communities.

2.4.2.1 Range Shifts and local population extinctions

Globally, about half of species with sufficient long-term data and with minimal influence of confounding, non-climatic factors have shifted their ranges poleward and upward (Table 2.2; section 2.4.2). Ranges shifts stem from local extinctions along warm-range-boundaries, as well as from colonization of few regions at cold-range-boundaries. Local extinctions (disappearance of local populations within a species range) have been much better documented in the past than have whole species' extinctions, although attribution to climate change can difficult. Wiens (2016) published an analysis of local extinctions and found them widespread among plant and animal species. Specifically he found that 47% of 976 species examined had undergone local extinctions as determined by published studies on range shifts – either range contractions or unidirectional shifts (local extinctions at lower elevations or on the warm edge of the species range). The overall percentage was higher in tropical species, 55%, than temperate, 39% of the species that showed local extinctions. Higher in animals (50%) than plants (39%) and highest in freshwater habitats (74%), then marine (51%) and finally terrestrial (46%). The difference between animals and plants is opposite the projections of which taxa may be more susceptible to range shifts in a warming climate (Warren et al., 2018). This may be due to a number of reasons. For example, the number of animals (760) showing range shifts in the literature was much greater than the number of plants (216). Furthermore, many plants are long-lived, and/or have large seed banks in the soil, making contractions far more difficult to detect than expansions. Finally, many animals are more mobile (have greater dispersal capabilities) than plants making it more likely that there shifts will be more detectable (Warren et al., 2018).

Cold water fish, such as coregonids and smelt have been negatively affected at the southern borders of their distributions (Jeppesen et al., 2012). Systematic shifts towards higher elevation and upstream were found for 32 stream fish species in France following geographic variation in climate change (Comte et al., 2013). These shifts have resulted in range contractions of cold-adapted species living at high elevations. Bull trout (*Salvelinus confluentus*) in Idaho (USA), were estimated to have lost 11–20% (8–16% decade^{-1}) of the headwater stream lengths that were cold enough for spawning and early juvenile rearing, with the largest losses occurring in the coldest habitats (Isaak et al., 2010). Likewise, the distribution of the stonefly *Zapada glacier,* endemic to alpine streams of Glacier National Park in Montana (USA), has been reduced over several decades by upstream retreat to higher, cooler sites as water temperatures have increased and glacial masses decreased (Giersch et al., 2015).

In contrast warm-adapted species have invaded habitats that were formerly too cool for them. An invasive freshwater cyanobacterium, *Cylindrospermopsis raciborskii* originated from the tropics, has spread to temperate zones over the last few decades due to climate change-induced earlier increase of water temperature in spring (Wiedner et al., 2007). In general, Cyanobacteria, which exhibit optimal growth at high water temperature and profit from stable thermal stratification and high nutrient concentrations, are on the rise (Domisch et al., 2013; Adrian and Hessen, 2016) as climate change indirectly affects both thermal and nutrient regimes in their favour *(very high confidence)*.

Range-shift responses have led to predictable changes of community composition in freshwater systems (Barry et al., 1995; Devictor et al., 2012). For example, a gain in warm water habitat and loss in cold water habitat has affected the relative fish species composition in Lake Superior (Cline et al., 2013). A long-term study of 24 European lakes revealed a decline in the abundance of the cold–stenothermal Artic charr (*Salvelinus alpinus*) and an increase in eurythermal fish species (e.g. the thermo-tolerant carp *Cyprinus carpio,* common bream, pike perch, shad). Observed long-term trends in stream macroinvertebrates have also shown that changes in species composition and community structure can be attributed to climate change, e.g., in France over a 20 year period (1985-2004) triggered by hydroclimatic changes (Daufresne et al., 2007) and in Australia over a 13 year period (1994-2007) (Chessman, 2009), where species likewise responded to hydroclimatic changes with significant changes attributed to thermophily and rheophily of individual species.

2.4.2.2 Phenological Responses

In freshwater communities, phenological changes have been related to changes in the physical environment as water temperature has risen, ice cover has been reduced and thermal stratification prolonged in lakes. There has been earlier timing of spring phytoplankton and zooplankton development and ~~the~~ earlier spawning by fish, as well as extension of the growing season (Adrian et al., 2009; Domis, 2013; Adrian and Hessen, 2016) (*very high confidence*). Thackeray (2010) found a trophic level asynchrony in rates of phenological change, with timing of phytoplankton bloom showing the slowest rate of advance, followed by invertebrates and vertebrates. Whether those phenological shifts cause mistiming in interaction is under debate (Singer and Parmesan, 2010; Straile et al., 2012). The median return time of Atlantic salmon among rivers in Newfoundland and Labrador advanced by 12- 21 days over the past decades, associated with overall warmer climatic conditions. Warming-induced accelerated ontogenetic development may enable the production of additional generations within a year, as has been shown for copepods (Winder et al., 2009b; Adrian et al., 2012).

2.4.2.3 Complex Phenological Responses

Early meta-analyses tested the straightforward hypotheses that warming should shift timing earlier and ranges poleward. Once these trends had been established, exceptions to them became foci of study. For example, some plants in northern regions of the northern hemisphere were busily retarding their spring flowering instead of advancing it as expected with warming. These turned out to be species requiring vernalization (winter chilling) to speed spring development. For these plants, phenological changes result from combined effects of advancement caused by spring warming and retardation caused by winter warming. Incorporating this level of complexity into analyses revealed that a greater proportion of species were responding to climate change than estimated under the simple expectation that warming should always cause advancement (Cook et al., 2012).

Animal species can show vernalization equivalent to that in plants (Stålhandske et al., 2017). However, a semi-global meta-analysis across terrestrial animals failed to detect delaying effects of warming winters (Cohen et al., 2018). The same animal-based meta-analysis contrasted phenological changes in temperate-zone animals, which are principally explained by changes of temperature, with those at lower latitudes, which follow changes of precipitation (Cohen et al., 2018). Precipitation has also been implicated in exceptions to the rule that ranges should be shifting to higher elevations. In dry climates, increases of precipitation accompanying climate warming can facilitate downslope range shifts (Tingley et al., 2012).

[INSERT TABLE 2.2 HERE]

Table 2.2: Global Fingerprints of Climate Change Impacts across Wild Species. Updated from Parmesan and Hanley 2015 [PLACEHOLDER FOR SECOND ORDER DRAFT: update still in progress]. For each dataset, a response for an individual species or functional group was classified as (1) no response (no significant change in the measured trait over time), (2) if a significant change was found, the response was classified as either consistent or not consistent with expectations from local or regional climate trends. Percentages are approximate and estimated for the studies as a whole. Individual analyses within the studies may differ. The specific metrics of climate change analysed for associations with biological change vary somewhat across studies, but most use changes in local or regional temperatures (e.g., mean monthly T or mean annual T), with some using precipitation metrics (e.g. total annual rainfall). E.g. a consistent responses would be poleward range shifts in areas that are warming. Probability (P) of getting the observed ratio of consistent:not consistent responses by chance was $<10^{-13}$ for Parmesan & Yohe 2003, Root 2003, Root 2005 and Poloczanska 2013, and was <0.001 for Rosenzweig 2008 (source=publication) (Parmesan and Yohe, 2003; Root et al., 2003; Root et al., 2005; Rosenzweig et al., 2008; Poloczanska et al., 2013). Test were all binomial tests against p=0.5, performed by Parmesan

[START FAQ 2.1 HERE]

FAQ2.1: Are wild species able to move to new locations where the climate is becoming more suitable?

[PLACEHOLDER FOR SECOND ORDER DRAFT]

[END FAQ 2.1 HERE]

2.4.2.4 Changes in Body Size

In addition to changes in phenology and distribution, a decrease in body size has been suggested as a general response of species to climate change in freshwater species given the temperature related constraints of metabolism with increasing body size. Reduced body size in response to global warming has been documented for freshwater bacteria, plankton and fish, as well as a shift towards smaller species (Daufresne et al., 2009; Winder et al., 2009a). Observations of the Lake Erie yellow perch, *Perca flavescens,* showed that following short winters with higher spring water temperatures, females produced smaller eggs than females exposed to long winters and colder spawning waters (Farmer et al., 2015).

Evidence is weak for a consistent reduction in body size across taxonomic groups in terrestrial animals (Siepielski et al., 2019). Changes to fundamental ecological characteristics such as body size (and length of appendages) have been documented in terrestrial ecosystems in response to changing climate, but trends are not always linear or consistent across taxonomic groups or geographic regions (Gibson et al., 2011; Gotanda et al., 2015). Decreased body size is expected based on biogeographic rules such as Bergmann’s Rule, but both increases and decreases have been documented in mammals, birds and invertebrates and attributed to climate change (Teplitsky and Millien, 2014; Gotanda et al., 2015). Evidence is strongest for birds (Yom-Tov, 2001) and for some mammals (Yom-Tov and Yom-Tov, 2004) but contrasting patterns have been found in ectotherms (Chamaille-Jammes et al., 2006). Contrasting patterns may be due to short-term modifications in selection pressures (e.g., changes to predation and competition), variation in life histories or a result of interactions with climate variables other than temperature (e-g., changes to food availability with rainfall changes) and other disturbances (Yom-Tov and Yom-Tov, 2004) or body size shapes (linear vs. volumetric dimensions). A comparison between phytoplankton and tree size revealed that individual body masses in tree and phytoplankton communities followed power-law distributions. However, the average exponents of these individual size distributions differed. Phytoplankton communities showed an average individual size distribution exponent consistent with three-quarter-power scaling of metabolism with body mass while tree communities deviated from this pattern in a manner consistent with equivalence in energy use among diameter size classes. The divergent aspects of body size (volumetric vs. linear dimensions) shapes the ecological outcome of metabolic scaling in forest vs. pelagic ecosystems.

2.4.2.5 Chytrid Fungus and Climate Change

Infection by the chytrid fungus *Batrachochytrium dendrobatidis* (subsequently Bd), can cause a disease, chytridiomycosis. Bd is widely-distributed both in the Americas and in Australia. It successfully accompanied and caused disease in high-elevation Andean frogs as they expanded their ranges to reach 5,200-5,400m (Seimon et al., 2017). Bd has been associated with severe declines of tropical frogs, and early work argued that chytridiomycosis was the sole cause of these declines. However, the 1980's extinction of the Golden Toad, *Incilius periglenes*, in Costa Rica occurred in the apparent absence of Bd (Pounds et al., 2006), and the fungus has been recorded as endemic in frog populations that did not suffer disease, where it may be commensal rather than parasitic (Retallick et al., 2004; Daskin et al., 2011; Puschendorf et al., 2011; Zumbado-Ulate et al., 2014). These findings have stimulated laboratory experiments and intensive analyses of observed patterns of infection and disease in nature, leading to the developing consensus that the interaction between chytrids and frogs is climate-sensitive, and that both climate change and Bd were involved in global amphibian declines (Puschendorf et al., 2011; Cohen et al., 2019a). More specifically, (Cohen et al., 2019b) give evidence supporting their "thermal mismatch hypothesis," which expects that vulnerability to disease should be higher at warm temperatures in cool-adapted species and higher at cool temperatures in warm-adapted species.

2.4.2.6 Evolutionary Responses

Prior sections document species' tendencies to retain their climate envelopes by some combination of range shift and phenological change. However, this tracking of climate change may be incomplete, causing species or populations to experience hotter conditions than those to which they are adapted and to thereby incur "climate debts" (Devictor et al., 2012). The importance of population-level debt is illustrated by a study in which estimated debt values were correlated with population dynamic trends in a North American migratory songbird, the Yellow Warbler, *(Setophaga petechia)*. Debt values were obtained from genomic analyses independent of the population trends, and were distributed across the species' range in a mosaic, not simply concentrated at a range margin, rendering the results robust to being confounded broad-scale geographical trends (Bay et al., 2018).

In the absence of evolutionary constraints, climate debts might be cancelled by evolutionary increases in thermal tolerance and in ability to perform in high ambient temperatures. In species already showing local adaptation to climate, it is expected that populations currently living at relatively cool sites can evolve to adopt traits of those currently at warmer sites, as their experience of climate changes (Singer, 2017; Socolar et al., 2017). However, this does not mean that populations at warm range limits will be able to track climate change in situ by evolving to survive outside the species' traditional climate envelope (Singer, 2017); whether or not they can do so depends on the level of "niche conservatism" operating at the species level. This topic was reviewed in the context of climate change by (Lavergne et al.).

As expected, an increasing number of studies documents evolutionary responses to climate change in populations that are not at warm range limits (Franks and Hoffmann, 2012). Small organisms with short generation times should have higher capacity to genetically track climate change than species with long generation times, such as mammals (Boutin and Lane, 2014). Indeed, observed evolutionary impacts have been mainly documented in insects (freshwater and terrestrial). These changes include rapid evolution at expanding range margins, reviewed by (Chuang and Peterson, 2016), increased dispersal abilities at colonizing wavefronts (Thomas et al., 2001) and changes of host specialization at the same expanding range margins (Bridle et al., 2014).

Away from range margins, individual populations experiencing regional warming have been evolving in diverse traits related to climate adaptation. For example, pitcher-plant mosquitos in the northwest USA have evolved to wait for shorter daylengths before initiating diapause, allowing them to respond to lengthening summers by delaying their overwintering stage until later in the autumn and adding an extra generation each year (Bradshaw and Holzapfel, 2001). Among 26 populations of *Drosophila subobscura* studied on three continents, 22 experienced climate warming across two or more decades, and 21 of those 22 showed increasing frequencies of chromosome inversions characteristic of populations adapted to hot climates (Balanya et al.).

If a species' whose range limits have been determined by climate finds itself completely outside its traditional climate envelope, extinction is expected in the absence of "evolutionary rescue" (Bell and Gonzalez, 2009; Bell et al., 2019). To investigate the potential for evolution that could enable a species to survive in a novel climate entirely outside its traditional climate envelope, experiments have been carried out on poikilotherms to test thermal performances, thermal tolerances, and their evolvabilities (Castaneda et al., 2019; Xue et al., 2019). Tests of thermal performance have not been simple, since the subjects tend to show both long-term acclimation and transgenerational effects (Sgro et al., 2016). However, the results to date have been consistent. Despite the widespread existence of local adaptation to climate across species' ranges, substantial constraints exist to the evolution of greater stress tolerance (e.g. high temperatures and drought) at warm range limits (Hoffmann and Sgro, 2011; MacLean et al., 2019). For example, as temperature was experimentally increased, the amount of genetic variance in fitness of *Drosophila melanogaster* decreased, so in hot environments the flies had low evolvability (Kristensen et al., 2015). The hypothesis that tolerance of heat stress is evolutionarily constrained is further supported by experiments in which 22 Drosophila species drawn from tropical and temperate climes were subjected to extremes of heat and cold. They were found to differ as expected in cold tolerance, but not in heat tolerance or in the temperature of optimal performance (MacLean et al., 2019).

Plasticity, for example in acclimating to thermal regimes, can help organisms adapt to environmental change and the form and extent of plasticity can vary among populations that experience different climates (Kelly, 2019). Plasticity may, in response to novel extreme environments, generate phenotypic values that lie outside the prior range for the species, but plasticity itself has not been observed to evolve in the context of climate change (Kelly, 2019). In the light of these experimental results, it is not surprising that, to date, relevant genetic changes in nature (e.g. affecting heat tolerance) have not taken any species out of the boundaries of known previously-existing variation. Evolutionary rescue has not yet been observed in nature.

Episodes of hybridization between closely-related species have increased in recent decades as one species shifts its range boundaries, bringing it into greater contact with close relatives. For example, hybrids between polar bears and brown bears have been documented in northern Canada (Kelly et al., 2010). In North American rivers, hybridization between invasive rainbow trout and native cutthroat trout has increased in frequency as the invasive species expanded into warming waters (Muhlfeld et al., 2014). Whether climate-changed induced hybridizations can generate novel climate adaptations remains to be seen.

In sum, with present knowledge, evolution is *very unlikely* to be sufficient to prevent whole species' extinctions if a species' climate space disappears *(high confidence).*

2.4.3 Observed Changes in Vegetation/Communities/Biomes

2.4.3 Detection and Attribution for Biome Shifts

Attribution for biome shifts embodies the complexities of attribution for range shifts of individual species and, because of the extensive spatial scale of biomes, which can cross entire continents (Whittaker, 1975) (Olson et al., 2001; Woodward et al., 2004), non-climate factors assume greater importance in influencing spatial distributions of biomes (Ellis and Ramankutty, 2008). The most robust attribution studies focus on data from individual locations for which confounding factors are minimal, particularly recent land use change, and scale up by analyzing multiple locations across a long zone between biomes, and, as with individual species, multiple lines of evidence increase confidence (Hegerl et al., 2010; Parmesan et al., 2013). Multivariate statistical analyses may help assess relative weights among factors, including variables related to climate change, when multiple factors act together (Gonzalez et al., 2012). However, in ecological systems drivers are often acting in strong, significant interactions with each other, making quantitative assessment of strength of individual drivers difficult (Parmesan et al., 2013). In these cases, manipulative experiments are critical in assessing attribution to climate change drivers.

Certain biomes exhibit a relatively stronger relationship to climate, for example, arctic tundra, which generally shows a distinct ecotone with boreal conifer forest (Whittaker, 1975). In these areas, attribution of biome shifts to climate change can be relatively straightforward, if human land use change is minimal. However, other biomes, such as many grassland systems, are not at equilibrium with climate (Bond et al., 2005) and present a more complex challenge. In these systems their evolutionary history (Keeley et al., 2011; Strömberg, 2011; Charles-Dominique et al., 2016), distribution, structure and function have been shaped by interactions between climate and disturbances like fire and herbivory (Bakker et al., 2016) (Staver et al., 2011; Lehmann et al., 2014; Pausas, 2015; Malhi et al., 2016). Variability in disturbances are an inherent characteristic of these systems and a "control" situation is seldom available in nature. Furthermore, due to the integral role of disturbance in these biomes they have been widely affected by long-term and widespread shifts in grazing regimes, large scale losses of mega-herbivores and policies of fire suppression (Archibald et al., 2013; Malhi et al., 2016; Hempson et al., 2017). This inherent characteristic of these biomes makes it necessary to conduct climate change attribution on a case by case basis, and such assessments are complex as it is hard to separate out direct climate change impacts from either inherent variation within disturbance regimes or directional changes in background disturbances (detailed in section 2.3.4.3). Confidence in assessments will be increased when observed trends are supported by a mechanistic understanding of the response as identified by physiological studies, manipulative field experiments, greenhouse studies and lab experiments.

2.4.3.2 Global Patterns of Biome Shifts

Previous assessments found that field research on vegetation at the biome level detected latitudinal and elevational biome shifts at 19 sites in boreal, temperate, and tropical ecosystems and attributed the shifts to anthropogenic climate change (Gonzalez et al., 2010b), Figure 4.1 and Table 4.1 in (Settele et al., 2014)). Field research since the IPCC Fifth Assessment Report has detected additional biome shifts at numerous sites, upslope or latitudinal. While the shifts found in recent research efforts are consistent with the increased temperatures and altered precipitation patterns of climate change, they have not been attributed to anthropogenic climate change since the research efforts were not designed or did not conduct analyses necessary for attribution.

The biome shift detected over the longest period of time occurred on Chimborazo, a mountain in the Andes in Ecuador where Alexander von Humboldt surveyed 51 plant species in 1802 and researchers conducted a re-survey in 2012 (Morueta-Holme et al., 2015). During that period, temperature increased 1.7ºC and the upper limit of alpine tundra shifted upslope 500 m (Morueta-Holme et al., 2015). Other additional cases of upslope biome shifts that have been detected by field research and are consistent with climate change include upslope shifts of boreal conifer forest into alpine tundra in the Altai Mountains, Russia, 150 m from 1954 to 2006 (Gatti et al., 2019), in the Himalayas, Nepal, 50 m since ca. 1860, (Sigdel et al., 2018), in the Changbai Mountains, China, 33 m from 1985 to 2014 (Du et al., 2018), in the Great Basin mountains, USA, 19 m from 1950 to 2016 (Smithers et al., 2018), and across the mountains of Tibet, 300 m since ca. 1910 (Liang et al., 2016), shifts of temperate conifer forest into alpine tundra in British Columbia, Canada, from 1962 to 2005 (Jackson et al., 2016) and into subalpine grassland in California, USA since ca. 1920 (Lubetkin et al., 2017), an upslope shift of temperate deciduous into temperate conifer forest in Switzerland from 1983 to 2003 (Rigling et al., 2013), and an upslope shift of temperate shrubland and retraction of temperate conifer forest in Yellowstone, USA, ~150 m from 1988 to 2012 (Donato et al., 2016). Cases since the IPCC Fifth Assessment Report of poleward biome shifts that have been detected and are consistent with climate change include northward shifts of deciduous forest into boreal conifer forest in Québec, Canada, 5 km from 1970 to 2012 (Sittaro et al., 2017) and 20 km from 1970 to 2014 (Boisvert-Marsh et al., 2019). A network of vegetation plots at 117 tundra sites across the Arctic and mountain areas found a statistically significant increase of plant height from 1989 to 2015, a possible signal of more widespread biome shifts (Bjorkman et al., 2018).

In Europe, increased woodiness of open areas has been attributed to regeneration of vegetation after abandonment of agriculture, livestock grazing, timber cutting, and other local human land-use changes. Abandonment of traditional farming (haymaking) and livestock grazing has driven reforestation of alpine grassland, observed as an upward shift of treeline, at numerous sites across Europe (Ameztegui et al., 2016; Cudlin et al., 2017; Vitali et al., 2017). Likewise, increased use of fire to clear alpine pastures in Peru is facilitating upslope shifts of fire-tolerant shrubland (Bush et al., 2015).

In other systems (e.g. in the USA and Africa), research has detected biome shifts of grassland to woodland or forest but attribution is complex, as these systems are inherently driven by interactions of climate, atmospheric CO_2 levels, grazing, and/or fire. These biomes have been widely impacted by long-term and widespread shifts in grazing regimes, large scale losses of mega-herbivores and policies of fire suppression (Archibald et al., 2013; Malhi et al., 2016; Hempson et al., 2017). Human settlement has added an additional disturbance axis. For example, westward expansion of human settlement in the central U.S. may be contributing to westward shifts of eastern deciduous forest into grassland (Hanberry and Hansen, 2015).

In these systems, the role of climate change is assessed through inductive reasoning that relies on experimental manipulations and comparative approaches across landscapes to determine whether or not there may be a climate change signal, in concert with other drivers, that is shaping observed long-term changes (Potts et al., 2015). Attribution, then, relies on the synthesis of multiple results from experiments, field observations from natural and semi-natural sites to assess whether the observed trends are in line with projected outcomes of climate change. Experiments at field sites and in greenhouses in the USA, Australia and South Africa indicate that observed woody-encroachment into grasslands in those regions is consistent with effects of climate change and increased CO_2 in controlled experiments (Ellis and Ramankutty, 2008; Nackley et al., 2018; Manea and Leishman, 2019). Globally, woody encroachment into open areas (grasslands, arid regions and tundra) is *likely* being driven by climate change and increased CO_2 in concert with changes in grazing and fire regime (detailed in section 2.4.3.3).

2.4.3.3 Non-Forested Ecosystems

Figure 2.4: [PLACEHOLDERFOR SECOND ORDER DRAFT: Figure to show woody encroachment in open systems. Woody expansion into open ecosystems from 2000. Non-forested ecosystems have experienced a significant increase in tree and shrub cover at the expense of open areas. Light green circles will indicate increases in shrub cover, dark green

will indicate increases in tree cover. Only studies where change is attributed to climate change related processes or a combination of climate change and land-use change will be presented in this map.]

2.4.3.3.1 Arid Regions

Projections for deserts and xeric ecosystems predict that higher temperatures, lower rainfall and an increase in evaporation will cause a decline in soil moisture, causing a decline in plant growth and an expansion of arid vegetation, reduced ground cover and general aridification. Field studies in the Namib desert indicate an increase in woody plant cover in most localities, and a shift of mesic species into more arid environments (Rohde et al., 2019), with the trends being attributed to an increase in the amount of precipitation (including fog) (Gonzalez et al., 2010b) due to a westward expansion of convective rainfall in the arid-savanna –Namib transition and the eastern grass/shrublands. These findings accord with projected effects of global warming on the Benguela upwelling system (Haensler et al., 2011).

Arid grasslands and shrublands are experiencing changes in vegetation structure and function. Woody plant encroachment into arid shrublands in N. America (Caracciolo et al., 2016; Archer et al., 2017) and encroachment into arid shrublands, arid savannah and desert in Southern Africa is occurring (du Toit and O'Connor, 2014; Ward et al., 2014; Masubelele et al., 2015a; Masubelele et al., 2015b; Hoffman et al., 2019; Rohde et al., 2019) (*robust evidence, high agreement*) with recent research indicating that this trend is also occurring in Central Asia (Li et al., 2015) (*low evidence, low agreement*). In North American sagebrush steppe changes have been attributed to increases in temperature and earlier snowpack melt (Wuebbles et al., 2017; Mote et al., 2018; Snyder et al., 2019). Additional lines of evidence indicate that warming and shifts in rainfall seasonality, and elevated CO_2 interact with land-use (fire, and grazing/browsing regimes) to cause this trend with their relative importance and interaction strength differing markedly among locations (for review see Donohue et al., 2013; Caracciolo et al., 2016; Archer et al., 2017; Hoffmann et al., 2019; Rohde et al., 2019). Rising concentration of CO_2 is an important agent of change in arid areas as it acts to improve plant water use efficiency, which can benefit shrubs (*medium evidence, medium agreement*) (Polley et al., 1997; Morgan et al., 2004; Donohue et al., 2013).

Shifts in grass abundance have been documented in N. America, and a dominant grass species of the Chihuanan desert has expanded into arid grassland (Collins and Xia, 2014; Rudgers et al., 2018). The sagebrush steppes (cold deserts) in N. America are being invaded by non-native grasses (Chambers et al., 2014) with the invasion likely driven by an increase in favourable climates, particularly warmer temperatures, which benefit these grasses (Bradley et al., 2016; Hufft and Zelikova, 2016). In the South African semi-desert (Karoo), multiple lines of evidence indicate that the eastern semi-desert (Karoo) has experienced an increase in grassiness with arid grasslands expanding into semi-desert shrublands (du Toit et al., 2015; Masubelele et al., 2015a; Masubelele et al., 2015b). There is *robust evidence* that that these patterns have been caused by a significant increase in annual rainfall and a shift in rainfall seasonality (du Toit and O'Connor, 2014) but it is not certain if this change is from long-term climate cycles or anthropogenic climate change.

2.4.3.3.2 Grasslands

Grasslands occur across a broad range of climatic conditions but water is generally limiting for some part of the year, and most grasslands experience periodic droughts and a dormant season based on seasonal dry or cold conditions. Grazing, fire and climate are also important in determining the structure and function or grasslands (Blair et al., 2014).

The global extent of grasslands has experienced significant climate change related declines. Advances in treelines across the world continue to be documented (Song et al., 2018) with warming causing upslope increases in tree cover in mountain systems and an accompanying loss in montane grassland extent (Silva et al., 2016; Andela et al., 2017; Song et al., 2018; Aide et al., 2019). Since 1982 grassland in and around temperate continental forest has declined by 14% since 1982, boreal coniferous forest by 10% and subtropical humid forest by 9% (Song et al., 2018). Large amounts of grassland loss within temperate, mostly forested areas, can be been attributed to land abandonment, forest recovery, increasing number of plantations and large-scale reforestation and afforestation programs (Song et al., 2018).

Climatic drivers of woody expansion in temperature limited grasslands, particularly alpine grasslands is most frequently attributed to warming *(robust evidence, high confidence)* (Venter et al., 2018; Brandt et al., 2019), increase in water and nutrient availability from thawing permafrost (Zhou et al., 2015; Silva et al., 2016) *(medium evidence, high agreement)* and rising CO2 (Frank et al., 2015; Aide et al., 2019) *(medium evidence, medium agreement)*. Similarly, the widely documented shrub expansion in arid grasslands (du Toit and O'Connor, 2014; Archer et al., 2017) is likely being facilitated by warming through reducing frost effects , accelerating growth rates and extension of the growing season *(medium evidence, high agreement)* (du Toit and O'Connor, 2014; Saintilan and Rogers, 2015; Archer et al., 2017) and the role of CO2 in improving water use efficiency and hence water availability in arid systems (Donohue et al., 2013; Saintilan and Rogers, 2015; Archer et al., 2017). Additional drivers like altered fire regimes and modification of grazing regimes may further exacerbate these trends (Archer et al., 2017; Hoffman et al., 2019).

Remote sensing shows overall increasing trends in both the annual maximum NDVI and annual mean NDVI grasslands ecosystems between 1982-2011 (Gao et al., 2016b). Other remote sensing studies confirm an increasing trend in grassland NPP in China and Australia, but decreasing in trend Europe and North America from 1981 to 2010 (Gang et al., 2015). Multiple lines of evidence indicate that changes in grassland productivity are positively correlated with increases mean annual precipitation. A global review of experimental rainfall manipulation studies show that increases in precipitation amount increases grassland annual net primary productivity (ANPP), most notably in drier grasslands, whilst reductions in water availability reduced ANPP, both aboveground and belowground biomass (Wilcox et al., 2017). These observations are confirmed in field studies where reductions in grassland productivity occur during droughts and increases in summer aridity (Hoover et al., 2014; Brookshire and Weaver, 2015; Gang et al., 2015; Gao et al., 2016b; Wan et al., 2018).

Whilst grassland productivity is sensitive to changes in water availability, grasslands response to warming is variable. Studies on warming in N. American grassland productivity is not directly sensitive to warming, but instead the indirect effects of warming (greater evaporative demand and lower soil moisture) which exacerbate drought responses (Hoover et al., 2014). Remote sensing studies indicate that that increasing temperatures have a positive impact on grassland production and biomass most specifically in temperature limited regions (Piao et al., 2014; Gao et al., 2016b), although this evidence indicates that this relationship maybe weakening (Piao et al., 2014). Grasslands in hot areas are expected to show decreased production with increases in temperature (Gang et al., 2015) *(limited evidence, low agreement)*. Climate warming effects plant community productivity, composition, diversity and phenology. On the Qinghai-Tibetan Plateau, warming increased deep-rooted, drought resistant plants and reduced dominant species growth, resulting in rapid species losses on alpine grasslands. Generally, increasing temperature had a negative effect on species diversity in alpine grasslands on the Qinghai-Tibetan Plateau. Experimental warming of 2.5°C, and 20% increase in precipitation (Wan et al., 2018) enhanced the positive effect of warming on community diversity and biomass, and offset the negative effect of warming on gross ecosystem productivity and net ecosystem exchange (Wan et al., 2018; Wang et al., 2018)

Evidence indicates that grassland responses to warming and drought are being ameliorated by increasing CO_2 and associated improved water use efficiency (Roy et al., 2016). For example in a cool temperate grassland experimental warming led to a longer growing season and elevated CO_2 further extended growing by conserving water, which enabled most species to remain active longer (Reyes-Fox et al., 2014), although these effects are most likely to be strongest in drier regions (Hovenden et al., 2014) *(medium evidence, medium agreement)*.

2.4.3.3.3 Savanna

Remote sensing studies demonstrate widespread woody encroachment and forest expansion in water-limited savannas in Central and West Africa forest expansion and woody encroachment with increases in precipitation and atmospheric carbon dioxide likely determinants of change (Venter et al., 2018; Brandt et al., 2019; Zhang et al., 2019). Extreme high-rainfall anomalies also contributed to the greening of the Sahel (Brandt et al., 2019; Zhang et al., 2019). Regional studies, remote sensing and repeat aerial photo-analysis all demonstrate that a global trend of woody encroachment in open grassy ecosystems is occurring tropical savannas with an 8% increase per decade in South America, a 2.4% increase in African savannas and 1% increase per decade in Australia savannas (O'Connor et al., 2014; Espírito-Santo et al., 2016; Skowno et al., 2017; Stevens et al., 2017; McNicol et al., 2018; Venter et al., 2018; Rosan et al., 2019), and temperate (most notably N. American) savannas (reviewed in Archer et al., 2017) (*high agreement, robust evidence*). Additionally forest expansion into mesic savannas is occurring widely in Africa, South America and SE Asia (Marimon et al., 2014; Keenan et al., 2015; Baccini et al., 2017; Ondei et al., 2017; Stevens et al., 2017; Aleman et al., 2018; Rosan et al., 2019)

Widespread woody biomass increases are attributed to an interaction with global drivers like atmospheric CO_2 *(high agreement, medium evidence)* (Stevens et al., 2016; Stevens et al., 2017; Nackley et al., 2018), altered rainfall amount and intensity (Venter et al., 2018; Xu et al., 2018; Zhang et al., 2019) *(high agreement, high evidence)* and warming (Lehmann et al., 2014; Venter et al., 2018) *(medium agreement, low evidence)* interacting with local land use drivers of fire suppression *(high evidence, high agreement),* heavy grazing (*high agreement, high evidence*) (for reviews see Archibald, 2016; Archer et al., 2017; Venter et al., 2018), removal of native browsers *(high agreement, medium evidence)* and loss of mega-herbivores in Africa (Asner et al., 2016b; Daskin et al., 2016; Stevens et al., 2016; Bakker and Svenning, 2018; Davies et al., 2018) *(high agreement, high evidence).*

As savannas structure and function are shaped by interactions between fire, herbivory and climate (Lehmann et al., 2014), any changes in these drivers will impact the savanna biome.

2.4.3.3.4 Tundra

Arctic ecosystems are warming more than twice as rapidly as other regions of the planet with on average 2°C warming since 1950 (IPCC AR6 WGI) *(robust evidence, high agreement)*. The warming at high latitudes is leading to earlier snow and sea ice melt and longer growing seasons (WGI AR6) which are altering plant communities with feedbacks across tundra food webs (Post et al., 2009; Gauthier et al., 2013) *(medium evidence, high agreement)*. Satellite data and repeat photographs indicate wide-spread greening of tundra vegetation, though there is complexity around the interpretation of satellite data in high latitude ecosystems (Phoenix and Bjerke, 2016). Tundra ecosystem hydrology, soil moisture, permafrost, carbon storage and nutrient cycling could influence the changing climate conditions thus potentially altering the trajectories for tundra ecosystem with warming (Sistla et al., 2013; Crowther et al., 2016; Salmon et al., 2016; Wik et al., 2016). An estimate of methane emissions from 733 lakes above 50°N latitude of 16.5 Tg CH_4 Yr^{-1} constitutes roughly two-thirds of all natural methane sources in the region (Wik et al., 2016). Changes to Arctic marine ecosystems as a result of declining sea ice extent have repercussions in coastal regions and on land and vice versa in the Arctic (Kerby and Post, 2013; Post et al., 2013; Bhatt et al., 2014).

Three decades of experiments and monitoring indicate that climate warming is causing increases in shrub, grass and sedge species abundance, density, frequency and height, with decreases in mosses and/or lichens at some tundra sites (Myers-Smith et al., 2011; Bjorkman et al., 2018; Bjorkman et al., 2019) *(robust evidence, high agreement)*. Shrub growth is climate sensitive with greater annual growth in years with warmer growing seasons (Myers-Smith et al., 2015). Multiple lines of evidence indicate increases in plant species that prefer warmer conditions (Elmendorf et al., 2015; Bjorkman et al., 2018). Bare ground is decreasing as plant cover increases in long-term monitoring plots (Bjorkman et al., 2019; Myers-Smith et al., 2019a). Animal species such as moose, beavers and songbirds may already be responding to this vegetation change by expanding their ranges northward or upslope into shrub tundra (Boelman et al., 2015; Tape et al., 2016a; Tape et al., 2016b; Tape et al., 2018). In addition to direct warming responses, thawed permafrost, altered hydrology and soil moisture conditions and enhanced nutrient cycling are indirect climate change mechanisms likely leading to pronounced vegetation change (Schuur et al., 2009; Natali et al., 2012) *(medium evidence, medium agreement)* as soil moisture status influences the temperature sensitivity of plant growth and canopy heights (Myers-Smith et al., 2015; Ackerman et al., 2017; Bjorkman et al., 2018). Vegetation change may be influenced by permafrost thaw and increasing active layer depths in tundra soils (Natali et al., 2012; Myers-Smith et al., 2015; Keuper et al., 2017; Myers-Smith et al., 2019b). Below-ground plant growth dynamics may be decoupled from above-ground dynamics in tundra ecosystems with below-ground root growth continuing until soils refreeze in autumn (Iversen et al., 2015; Blume-Werry et al., 2016; Radville et al., 2016).

2.4.3.3.5 Mediterranean-Type Ecosystems

All five Mediterranean-Type Ecosystems (MTEs) of the world have experienced extreme droughts in the past decade, with South Africa and California reporting the worst on record (Diffenbaugh et al., 2015; Williams et al., 2015; Otto et al., 2018; Sousa et al., 2018). These droughts have been linked to climate change and are expected to become more frequent and severe (AghaKouchak et al., 2014; Otto et al., 2018). Ecosystems across the MTEs have shown a range of direct responses to various forms of water deficit, but have also been affected by increasing fire activity linked to drought, and interactions between drought or extreme weather and fire, affecting post-fire ecosystem recovery. In general, responses include one or more of: shifts in functional composition and state shifts, decline in vegetation health, decline or loss of dominant or characteristic species, shifts in composition towards more drought- or heat-adapted species and declining diversity. Many studies documenting type conversions or shifts in composition report increasing prevalence or dominance by invasive alien species, consistent with the expectation that invasive species are likely to benefit from climate change and colonize affected sites more readily than hot- and/or drought-adapted indigenous species.

Remote sensing approaches have shown a drought associated decline in canopy health in forests within MTE in the Southern Afrotemperate Forests of South Africa and a decline in canopy water content in forests in California (Asner et al., 2016a). Several studies reported climate associated responses of dominant or charismatic species; in the Fynbos of South Africa a high mortality in the Clanwilliam Cedar over the period 1931-2013 occurred in individuals at lower, hotter elevations. Portuguese shrublands have experienced a loss of deciduous and evergreen oaks to pyrophytic xeric trees. In California, there has been high canopy foliage die-back in the Giant Sequoia over the 2012-2015 drought (Stephenson et al., 2019) and increasing dominance of oaks relative to pines due to increases in climatic water deficit and large-scale mortality due to the interaction between drought in insect outbreaks (McIntyre et al., 2015; Fettig et al., 2019).

In the Great Basin, USA, a small part of which has a climate with Mediterranean characteristics, two conifer species shifted upslope 19 m from 1950 to 2016 (Smithers et al., 2018). Reduced winter precipitation in a Californian grassland has caused long-lasting and potentially unidirectional reductions diversity through the loss of native annual forbs. Diversity declines also occurred in the Fynbos of South Africa, (Slingsby et al., 2017) due to increasing prevalence of extreme hot and dry weather during the post-fire regeneration phase. In particular they found that graminoid species (Poaceae, Cyperaceae and Restionaceae) and species that resprout after fire were worst affected (Slingsby et al., 2017).

In California, USA, numerous factors, including climate, wildfire, land-use change, invasive species, and nitrogen deposition have contributed to conversion of some areas of forest, woodlands, or shrublands to grasslands (Abelson et al., 2015; Jacobsen and Pratt, 2018; Park et al., 2018; Park and Jenerette, 2019; Syphard et al., 2019). Climate change has caused observed increases in fuel aridity and area burned by wildfire across the western U.S. from 1985 to 2015 (Abatzoglou and Williams, 2016). Local and global climatic variability led to a 4-year decrease the average fire return time in Fynbos, South Africa when comparing fires recorded between 1951-1975 and 1976-2000 (Wilson et al., 2010). The effects of climate change on heat, fuel, and ignition limitations of wildfire shows spatial and temporal variation (Section 2.3.6.1).

2.4.3.4 Forests

2.4.3.4.1 Tropical forest

Tropical forests cover has declined (Hansen et al., 2013; Kohl et al., 2015; Liu et al., 2015; Baccini et al., 2017), with forest declines outweighing gains by 3.6 times (Hansen et al., 2013), driven by primarily by deforestation (Lewis et al., 2015). Expansion of tropical forest cover and extent into savannas and grasslands have occurred in Africa (Baccini et al., 2017; Aleman et al., 2018), South America and Australia (Marimon et al., 2014; Ondei et al., 2017; Stevens et al., 2017; Rosan et al., 2019). Upslope range shifts of tropical deciduous forest into alpine grassland in the Andes mountains in Colombia, Ecuador, Peru, and Argentina has been recorded (Fadrique et al., 2018; Aide et al., 2019) *(high evidence, high agreement)*. These shifts are likely driven by interactions between changing land-use like fire suppression and climate changes like increased rainfall, warming and elevated CO_2 either through CO_2 fertilisation or increases in water-use efficiency (Cernusak et al., 2013; Yang et al., 2016) (Van Der Sleen et al., 2015) *(robust evidence, medium agreement)*.

An extensive decline in tropical forest area is primarily driven by deforestation and land conversion, but climate change is contributing to forest degradation by reducing the resilience and health of forests (Malhi et al., 2014). Although data since AR5 from long-term monitoring plots and remote sensing continues to confirm aboveground biomass increases in intact tropical forest in the Amazon (Gatti et al., 2014; Brienen et al., 2015; Baccini et al., 2017), Africa and SE Asia (Qie et al., 2017), most likely elevated CO_2 (Ballantyne et al., 2012; Brienen et al., 2015; Sitch et al., 2015; Yang et al., 2016; Mitchard, 2018)*(high evidence, medium agreement),* these gains have been declining over the past two-decades and with the in the Amazon (Brienen et al., 2015), SE Asia (Qie et al., 2017) and potentially in Central Africa. Declines in productivity are most strongly associated with warming, reduced growth rates during droughts (Bennett et al., 2015; Bonal et al., 2016; Corlett, 2016), drought related mortality (Brando et al., 2014; Zhou et al., 2014; Brienen et al., 2015; Corlett, 2016; McDowell et al., 2018) and fire (Liu et al., 2017), with mechanisms being identified through long-term monitoring and field experiments (Bonal et al., 2016) *(robust evidence, high agreement).* The interaction of increases in frequency and severity of droughts and shorter tree residence times due to increases in growth rates caused by elevated co2 maybe an additional interactive factor increasing tree mortality (Malhi et al., 2014; Brienen et al., 2015). However, not all tropical forests have exhibited increasing mortality (McDowell et al., 2018; Meakem et al., 2018). The vulnerability to drought varies between tree species and tree size with large long lived trees are at risk for highest rates of mortality (Bennett et al., 2015). Vulnerability to mortality also varies between forest types with aseasonal forest appearing to be most vulnerable to drought impacts (Corlett, 2016). Additionally there is emerging evidence that there is a shift in species composition in Neotropical forests and West Africa with *low-medium evidence* indicating that changes are associated with intensification of dry spells and drought (Corlett, 2016; Van der Sande et al., 2016; Aguirre-Gutiérrez et al., 2019; Esquivel-Muelbert et al., 2019).

Drought, warming and increases in forest fragmentation have interacted to increase the prevalence of fires in tropical forest (*medium evidence, high agreement*). Warming interacts with forest plants by increasing water stress in trees (Corlett, 2016). Warming and forest fragmentation dramatically increases dry-season desiccation of forest canopies and deforestation leads to hotter and drier regional climates (Malhi et al., 2014; Lewis et al., 2015). Deforestation and fragmentation promote invasion of grasses into forest edges which promotes the spread of fire (Baccini et al., 2017). This risk is intensified with warming and drought (Brando et al., 2014; Balch et al., 2015; Lewis et al., 2015). Droughts and fires have additive effects on tree mortality. Evidence mostly from the Amazon indicates that droughts and fires have additive effects on increasing mortality, reducing canopy cover and aboveground biomass (Brando et al., 2014; Balch et al., 2015; Lewis et al., 2015).

2.4.3.4.2 Temperate forests

[PLACEHOLDER FOR SECOND ORDER DRAFT]

2.4.3.4.3 Boreal forests

[PLACEHOLDER FOR SECOND ORDER DRAFT]

2.4.4 Observed Changes in Ecosystem Processes and Services

2.4.4.1 Browning of Rivers and Lakes

In boreal, coniferous areas there has been an increase in terrestrial derived dissolved organic matter (DOM) transport into rivers and lakes that causes a change in the optical properties of water (generally increased opacity) and a shift toward a brown colour. This is a consequence of climate change induced hydrological intensification, climate change-driven greening of the northern hemisphere, and changes in forestry practices (Finstad et al., 2016; Creed et al., 2018; Hayden et al., 2019) *(high confidence).* Number of days above 0°C or a moderate increase in temperature of 2°C has the potential to increase DOM export into freshwater (Weyhenmeyer and Karlsson, 2009). Browning creates a positive feedback by absorbing photosynthetically active radiation resulting in an acceleration of upper water (epilimnetic) warming (Solomon et al., 2015). Browning of lakes leads to shallower and more stable thermoclines and thus to an overall cooling of deep water temperatures (driven by an increase in the relative volume of deeper waters - the hypolimnion) (Solomon et al., 2015; Williamson et al., 2015). Brownification can provoke a transition of the seasonal mixing regime from a mixed lake (polymictic) to one that is seasonally stratified (Kirillin and Shatwell, 2016). Browning can drive a shift from auto- to heterotrophic-based production with a subsequent decline in energy transfer efficiency and a reduction of biomass at higher trophic levels (Ellison et al., 2017). Browning may also accelerate primary production through input of nutrients associated with DOM in nutrient poor lakes (Thrane et al., 2014; Seekell et al., 2015; Ellison et al., 2017).

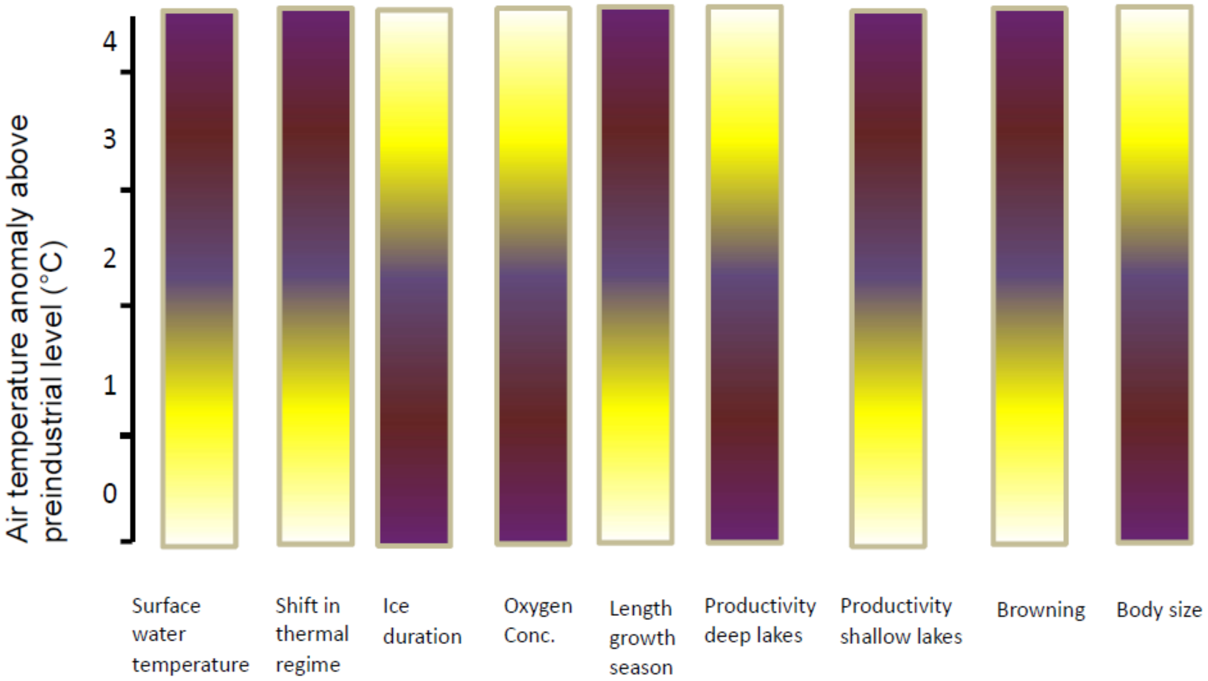


Figure 2.5: Large scale climate induced changes in freshwater ecosystems in relation to air temperature anomalies above pre-industrial level. Colour gradients from light to dark depict increasing ranges of positive respectively negative relationship between temperature anomalies and the response variables.

2.4.4.2 Wildfire

Wildfire is a natural and essential component of many forest, shrubland, and grassland ecosystems. Anthropogenic climate change can increase wildfire by exacerbating its three principal driving factors – heat, fuel, and ignition (Stephens et al., 2014; Jolly et al., 2015). At the same time, non-climate factors exacerbate wildfire, with farmers and livestock herders in tropical regions intentionally setting fire to clear lands and secondary forest for agricultures and governments in temperate zone countries implementing policies to suppress fires, even those naturally ignited (Andela et al., 2017; Lasslop and Kloster; Aragao et al., 2018). Fire suppression produces unnatural densities of understory trees and accumulations of coarse woody debris, the major fire fuels (Ruffault and Mouillot, 2015; Hessburg et al., 2016; Kelley et al., 2019).

Since the IPCC Fifth Assessment Report, four published research efforts have detected wildfire increases, analyzed the contribution of climate factors, and attributed the wildfire increases mainly to anthropogenic climate change (Abatzoglou and Williams, 2016; Holden et al., 2018; Kirchmeier-Young et al., 2019; Mansuy et al., 2019). These cases of detection and attribution all concern wildfire in North America. Across the western United States, increases in vegetation aridity due to hotter temperatures from anthropogenic climate change doubled burned area from 1984 to 2015, compared to what would have burned without climate change (Abatzoglou and Williams, 2016). In addition, the hotter temperatures of anthropogenic climate change combined with statistically significant decreases of summer rainfall from 1979 to 2016 to increase burned area across the western United States (Holden et al., 2018). In British Columbia, Canada, the hotter maximum temperatures of anthropogenic climate change increased burned area in the fire season of 2017 to seven to eleven times the area that would have burned without climate change (Kirchmeier-Young et al., 2019). In protected areas of Canada and the United States, climate factors explained the majority of burned area from 1984 to 2014, with climate factors (temperature, precipitation, relative humidity, evapotranspiration) outweighing local human factors (population density, roads, and built-area) (Mansuy et al., 2019).

Globally, 4.2 million km^2 of land per year burned on average from 2002 to 2016 (Giglio et al., 2018) with the highest fire frequencies in the Brazilian Amazon, African deciduous tropical forest and savanna in wide latitudinal zones north and south of the Congo rainforest, northern Australia, and Kazakhstan (Earl and Simmonds, 2018; Andela et al., 2019). For global terrestrial area as a whole, average fire frequency increased slightly from 1900 to 2000, but the change was not statistically significant, based on spatial analyses (Gonzalez et al., 2010b) of field estimates of fire area (Mouillot and Field, 2005). Global average fire frequency was 4 ± 6 fires per century, corresponding to a fire return interval of 27 ± 17 years. Fire frequency increased on two-fifths of global land, slightly less than the area of decrease. Fire increased across the tropics of Africa, Asia, and South America, due to increased burning to clear agricultural fields, while fire decreased across Australia, North America, and Russia due to extensive suppression (Gonzalez et al., 2010b).

Global burned area decreased at a statistically significant rate of −1.4 ± 0.49% per year from 1998 to 2015 (Andela et al., 2017), based on spatial analyses of the Global Fire Emissions Database (Giglio et al., 2013), which combines remote sensing data from the Moderate Resolution Imaging Spectroradiometer (MODIS), the Along-Track Scanning Radiometer (ATSR), and the Visible and Infrared Scanner (VIRS). In contrast, spatial analyses of Global Fire Emissions Database burned area from 1996 to 2015 using linear quantile regression found a lower rate of decrease, 0.7% per year, that was not statistically significant (Forkel et al., 2019). Because the remote sensing time series is short, the high interannual variability of fire makes trends sensitive to the years chosen for the start and end of the analysis (Forkel et al., 2019). Much of the decreasing trend derives from two years: 1998 with high burned area and 2013 with low burned area. Furthermore, the fire reduction effect of reduced vegetation cover following expansion of agriculture and livestock herding can counteract the fire increasing effect of increased heat of climate change (Lasslop and Kloster, 2017; Arora and Melton, 2018; Forkel et al., 2019). The reduction of burning needed after the initial clearing for agricultural expansion drives much of the decline in fire in the tropics (Andela et al., 2017; Earl and Simmonds, 2018; Forkel et al., 2019). In the Serengeti-Mara savanna, Kenya and Tanzania, burned area decreased 40% from 2001 to 2014, but the change was not statistically significant (Probert et al., 2019). Nevertheless, an increase in domestic livestock could have caused the numerical reduction of burned area, with grazing reducing the grass cover that fuels wildfires in the savanna (Probert et al., 2019).

From 1998 to 2015, burned area increased significantly in African deciduous tropical forest and savanna south of the Congo rainforest, northern Australia, and the Cerrado of Brazil, but decreased significantly in African deciduous tropical forest and savanna north of the Congo rainforest, central Australia, and Kazakhstan (Andela et al., 2017). In the Amazon, wildfire has increased since 1973, with deforestation as the main cause (van Marle et al., 2017; da Silva et al., 2018). From 2003 to 2015, deforestation in the Amazon showed a statistically significant decline, but burned area showed no statistically significant change (Aragao et al., 2018). Deforestation in the Amazon fragments the rainforest and increases the flammability of vegetation (Alencar et al., 2015). In Africa, recent decreases in wildfire occurred in areas of precipitation increase (Zubkova et al., 2019). In the area around Sierra de Gredos, Spain, small wildfires increased from 1979 to 2008 but were related more to local human land use change than to climate change (Viedma et al., 2018). Across Southern Hemisphere Mediterranean ecosystems in Australia, Chile, and South Africa, wildfire increased from 1958 to 2014, correlated most to the Southern Annular Mode, which is partly driven by anthropogenic climate change (Mariani et al., 2018). In Australia, the warm phase of the El Niño-Southern Oscillation (ENSO), which is not primarily driven by climate change, increases fire risk (Dowdy, 2018). Repeated anthropogenic peatland fires annually occur in Southeast Asia, particularly in Sumatra, Kalimantan and Malay Peninsula, and these underground fires become severe, causing toxic haze pollution to human and wildlifes and large carbon emission, during the occurrence of ENSO and Indian Ocean Dipole (IOD) (Hoscilo et al., 2011; Page and Hooijer, 2016; Erb et al., 2018b; Wijedasa et al., 2018; Uda et al., 2019).

In North America, long time series of field observations have facilitated analyses of time series longer than the global remote sensing analyses. Anthropogenic climate change has increased wildfire significantly in western North America (Abatzoglou and Williams, 2016; Holden et al., 2018; Kirchmeier-Young et al., 2019; Mansuy et al., 2019), where ecosystems are relatively drier and more sensitive to increases in aridity under climate change. In California, burned area increased ~500% annually and ~800% in summer from 1972 to 2018 (Williams et al., 2019) and the upper elevation of fire occurrence shifted upslope ~300 m from 1908 to 2012 (Schwartz et al., 2015). In Arizona and New Mexico, the area of high severity fire increased from 1984 to 2015 (Singleton et al., 2019). Across the western U.S., the area burned by fires > 4 km2 tripled from 1984 to 2011 (Dennison et al., 2014) and increased ~1300% from 1972 to 2012 (Westerling, 2016). In Alaska, burned area doubled from 1943 to 2012 (Calef et al., 2015) while, in the interior of Alaska, a region that is sparsely inhabited where climate factors outweigh local human factors in causing wildfires, lightning-ignited burned area increased ~500% from 1975 to 2015, a trend (Veraverbeke et al., 2017). Across Canada, burned area doubled from 1959 to 2015, with natural ignitions increasing, but human ignitions not changing (Hanes et al., 2019).

The global increases in temperature of anthropogenic climate change and increases in aridity and drought due to climate change have increased the length of the fire weather season on one-quarter of global vegetated area, increased average fire season length by one-fifth, and doubled global burnable area affected by a fire season >1 standard deviation above the mean, from 1979 to 2013 (Jolly et al., 2015). Areas with the greatest increases in fire season length include the Amazon, western North America, Iran, and the Horn of Africa. Climate variability explained approximately one-third of inter-annual variability in burned area from 1997 to 2016 (Abatzoglou et al., 2018). In forest areas, the relationship of burned area is strongest to fuel aridity, a function of temperature; in non-forest areas the relationship of burned area is strongest to high precipitation in the previous year, which can produce high grass fuel loads (Abatzoglou et al., 2018). Globally, vegetation and moisture dominate other factors in controlling burned area, ignitions are particularly important in savannas and boreal forest, and human fire suppression dominates other factors in agricultural areas (Kelley et al., 2019).

The use of fire in agriculture and livestock raising has generated a second fire season on approximately one-quarter of global land with fire activity, based on spatial analyses of bi-modal fire areas and neighbouring uni-modal areas that shows more intensive land use and sub-optimal fire climate in the bi-modal areas (Benali et al., 2017). These areas include parts of Australia, Brazil, Canada, India, Kazakhstan, Russia, and the US. The intense human influence on fire occurrence reveals itself in lower fire frequencies on Sundays and Fridays in parts of the world where those are traditional religious days of rest (Earl et al., 2015).

The increased temperature and aridity of climate change have driven post-fire changes in plant regeneration and species composition in South Africa (Slingsby et al., 2017) and tree regeneration in the western U.S. (Davis et al., 2019). In the unique Fynbos vegetation of the Cape Floristic Region, South Africa, heat and drought in the year following a fire and legacy effects of eradicated exotic species reduced native plant species regeneration, reducing richness 12% from 1966 to 2010 and shifting the average temperature tolerance of species upward by 0.5ºC (Slingsby et al., 2017). During that time, annual average temperature had increased significantly 1ºC and the maximum number of consecutive dry days increased significantly from four to six days per year. In areas across the western U.S. that burned in wildfires, the increasing heat and aridity of climate change from 1979 to 2015 pushed low-elevation ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) forests across critical thresholds of heat and aridity that reduced post-fire tree regeneration by half (Davis et al., 2019). In the Rocky Mountains, mean summer temperature was the most important factor determining post-fire regeneration of ponderosa pine and Douglas-fir trees (Kemp et al., 2019), post-fire drought caused sharp declines in post-fire tree seedling establishment in subalpine forest, and the number of sites with no post-fire tree regeneration doubled in one sample from 1988 to 2011 (Stevens-Rumann et al., 2018). On south-western U.S. sky islands, high-severity fires converted some forest patches to shrublands, a biome shift, by killing pines and leaving oaks to re-sprout (Barton and Poulos). In California, frequent fires converted temperate conifer to temperate broadleaf forest, a biome shift, but this was a restoration of vegetation after a century of unnatural fire suppression (Nemens et al., 2018).

In addition to the carbon emissions from fires (see section 2.4.4.4), increased wildfire causes other self-reinforcing feedbacks to climate change. In Siberia boreal forest, reduced evapotranspiration following fire increased summer temperature 0.07-0.3ºC from 2005 to 2014, but increased albedo led a slight winter cooling, lower in magnitude than the summer warming (Liu et al., 2018). In the western U.S., reduced albedo following fire caused snowmelt to advance five days from 1999 to 2018, an effect that persisted more than a decade following fire (Gleason et al., 2019). From 1984 to 2012, earlier snowmelt showed a significant correlation to burned area (O'Leary et al., 2016). Earlier snowmelt in turn contributes to vegetation aridity (Westerling, 2016). It is also important to note that evapotranspiration rate of feather moss peatlands after severe wildfires increased 400%, which leads to high vulnerability to future warm climate (Kettridge et al., 2019).

2.4.4.3 Tree Mortality

Anthropogenic climate change can directly cause tree mortality through increased aridity or drought or indirectly through wildfire and insect pests. Catastrophic failure of the plant hydraulic system, in which a lack of water causes the xylem to lose hydraulic conductance, is the principal mechanism of drought-induced tree death (Anderegg et al., 2016; Adams et al., 2017; Anderegg et al., 2018; Choat et al., 2018). Previous assessments have shown widespread cases of tree mortality detected in boreal, temperate, and tropical ecosystems, mainly from drought that is consistent with climate change (Allen et al., 2010; Settele et al., 2014). Only a few cases, however, have been formally attributed to anthropogenic climate change, in western North America (van Mantgem et al., 2009), the African Sahel (Gonzalez et al., 2012), and North Africa (le Polain de Waroux and Lambin).

Since the Fifth Assessment Report (Settele et al., 2014), research has documented numerous additional cases of drought-induced tree mortality around the world. Many of these have been included in five efforts to update a previous meta-analysis of drought-induced tree mortality (Allen et al., 2010) with additional cases (Allen et al., 2015; Bennett et al., 2015; Martinez-Vilalta and Lloret, 2016; Greenwood et al., 2017; Hartmann et al., 2018). These reviews indicate more rapid mortality (Allen et al., 2015), rising background mortality rates (Allen et al., 2015), mortality increasing with tree size (Bennett et al., 2015), one-fourth of mortality cases leading to a vegetation shift (Martinez-Vilalta and Lloret, 2016), and multiple non-climate factors contributing to tree mortality, including timber cutting, grazing, and local air pollution (Martinez-Vilalta and Lloret, 2016). Furthermore, the logarithmic mortality response of trees to drought is consistent across biomes and functional groups, although tree species with denser wood and lower specific leaf area showed lower mortality (Greenwood et al., 2017).

In Amazon rainforest, where annual average temperature has increased ~1.2ºC from 1950 to 2018, (Marengo et al., 2018), biomass mortality in a set of 310 field plots increased ~40% from 1983 to 2011, leading to a decline of the standing carbon stock of the Amazon during that period (Brienen et al., 2015). In another sample of Amazon rainforest plots, however, tree mortality did not show a statistically significant change from 1965 to 2016, but rose abruptly in severe drought years, mainly years of negative anomalies in the El Niño-Southern Oscillation (Aleixo et al., 2019). Nearly half the area of the Amazon has experienced extremely dry conditions during El Niño, which can cause extensive wildfire (Anderson et al., 2018). Wildfire can increase tree mortality rates by >600% above rates in non-burned areas, with the higher mortality persisting a decade after a fire (Silva et al., 2018). Experimental burns in Amazon rainforest found abrupt doubling and quadrupling of tree mortality in drought years in plots burned annually and every three years, compared to background mortality (Brando et al., 2014).

In Africa, recent research in the African Sahel has continued to detect substantial tree mortality (Kusserow; Brandt et al., 2018; Trichon et al., 2018) and declines in tree biodiversity (Hanke et al., 2016; Kusserow, 2017; Ibrahim et al., 2018). Across southern Africa, nine of the 13 oldest known baobab trees (*Adansonia digitata*) have died since 2005 (Patrut et al., 2018). In South Africa, savanna trees experienced an order of magnitude increase in mortality following two years of half the average rainfall (Case et al., 2019).

In the United States, research since the IPCC Fifth Assessment Report shows that tree mortality has continued to increase in areas of increasing temperature. In California, the most severe drought in a century of weather station measurements occurred from 2012 to 2016 due to record heat and extremely low precipitation and analyses attributed one-tenth to one-quarter of the 2012-2014 period of the drought to anthropogenic climate change (Williams et al., 2015). The resulting subsurface moisture exhaustion and soil drying became so severe that anthropogenic climate change increased tree mortality in Sierra Nevada temperate conifer forests by one-quarter from 2012 to 2015 (Goulden and Bales, 2019). In one network of long-term monitoring plots, nearly one-quarter of trees died from 2014 to 2017, with mortality rates of ponderosa pine (*Pinus ponderosa)*and sugar pine (*Pinus lambertiana)* increasing up to 700% of pre-drought rates (Stephenson et al., 2019). Across the Sierra Nevada, ponderosa pine and large old trees of most species experienced the greatest mortality during the California drought (Fettig et al., 2019; Stephenson et al., 2019). Mortality was most severe at the lowest, hottest elevations, indicating species sensitivity to heat, and in plots with high tree density, indicating tree competition for water (Paz-Kagan et al., 2017; Young et al., 2017b; Restaino et al., 2019). In other parts of the western U.S. experiencing drought, lodgepole pine (*Pinus contorta*) mortality increasing 700% from 2000 to 2013 (Anderegg et al., 2015) and piñon pine (*Pinus edulis*) experiencing over 50% mortality from 2002 to 2014 (Redmond et al., 2018). In boreal North America, bark beetles have caused extensive mortality of white spruce (*Picea glauca*) (Hansen et al., 2016). In addition, the boreal species yellow-cedar (*Callitropsis nootkatensis*) has experienced up to 70% mortality in much of its range due to a reduction of snow cover from the increased winter temperatures of climate change, which has reduced protective insulation and led to freezing and death of fine roots (Hennon et al., 2012; Buma et al., 2017).

In California and across the western U.S., drought first weakens trees, then infestations of bark beetles are what most often kills them (Anderegg et al., 2015; Kolb et al., 2016; Lloret and Kitzberger, 2018; Redmond et al., 2018; Stephens et al., 2018; Fettig et al., 2019; Restaino et al., 2019; Stephenson et al., 2019). The increased heat of climate change has allowed bark beetles to move farther upslope and north than before and to survive through the winter, when they would previously die (Raffa et al., 2008; Bentz et al., 2010; Jewett et al., 2011; Macfarlane et al., 2013; Raffa et al., 2013; Hart et al., 2017).

Across the western U.S., bark beetles caused mortality of 7%, of forest area from 1984 to 2012, fire caused mortality of 6%, and timber harvesting caused mortality of 12% (Hicke et al., 2016; Berner et al., 2017), although these studies did not quantify tree mortality directly caused by drought. As described in this section above, field evidence shows that the increased heat of climate change has driven bark beetle infestations. As described in the section on fire above, analyses have attributed recent increases in wildfire in western North America mainly to anthropogenic climate change (Abatzoglou and Williams, 2016; Holden et al., 2018; Kirchmeier-Young et al., 2019; Mansuy et al., 2019). So, the second and third major causes of tree mortality in the western U.S. are tied to climate change.

Field research has documented tree mortality in the past decade from insect infestations related to hotter temperatures on trees in other countries, including cork oak (*Quercus suber*) in Tunisia (Bellahirech et al., 2019) and Norway spruce (*Picea abies*) across nine countries in Europe (Marini et al., 2017; Mezei et al., 2017). Increasing temperatures have caused a range expansion of bark beetles from glacial refugia in the Alps northward up to Scandinavia (Okland et al., 2019).

2.4.4.4 Terrestrial carbon stocks and changes

Terrestrial ecosystems contain stocks of 450 Gt (380-540 Gt) carbon in vegetation, 1700 ± 250 Gt carbon in soils, and 1400 Gt ± 200 Gt carbon in permafrost (Hugelius et al., 2014; Batjes, 2016; Jackson et al., 2017; Strauss et al., 2017; Erb et al., 2018a) (IPCC AR6, Working Group 1, Chapter 5). These stocks substantially exceed the ~900 Gt in unextracted fossil fuels (IPCC AR6, Working Group 1, Chapter 5). Land use change in terrestrial ecosystems, including agriculture, forestry, and other land use, emitted carbon to the atmosphere at a rate of 1.5 ± 0.7 Gt y^{-1} from 2008 to 2017, ~14% of combined anthropogenic emissions from land use change, fossil fuel combustion, and cement production (Le Quéré et al., 2018). Terrestrial ecosystems naturally removed carbon from the atmosphere at a rate of -3.2 ± 0.7 Gt y^{-1}. Therefore, they comprised a net sink at a rate of -1.7 ± 1.8 Gt y^{-1} from 2008 to 2017 for the world (IPCC AR6, Working Group 1, Chapter 5). Because most deforestation occurs in tropical forests, they comprise a net source of carbon to the atmosphere at a rate of 0.4 ± 0.09 Gt y^{-1}, with removals of -0.4 ± 0.03 Gt y^{-1} out-balanced by emissions of 0.9 ± 0.08 Gt y^{-1} (Baccini et al., 2017). Three-quarters of the net emissions from tropical forests come from the Americas. In contrast, boreal and temperate forests comprised net sinks of ~1 Gt y^{-1} and 0.7 Gt y^{-1}, respectively, from 1993 to 2012 as tree growth outbalanced losses (Liu et al., 2015).

Thus, terrestrial ecosystems protect globally critical stocks of carbon and provide an essential service of sequestration of carbon from the atmosphere but are at risk of carbon losses from deforestation and climate change. IPCC AR6, Working Group 1, Chapter 5 examines the biogeochemistry of the ecosystem parts of the carbon cycle. This section provides information on ecological aspects of carbon stocks and flows.

The ecosystem that attains the highest aboveground carbon density in the world is forest dominated by coast redwood (*Sequoia sempervirens*), in California, USA, with 2600 t ha^{-1} ± 100 t ha^{-1} (Van Pelt et al., 2016). Coast redwood is the species with the tallest tree in the world with a height of 115 m (Sillett et al., 2015). The tree species with the most massive individual trees in the world is giant sequoia (*Sequoiadendron giganteum*), in California, USA, with a tree of 580 t dry mass (Sillett et al., 2015). The ecosystem with the second highest documented carbon density in the world is forest dominated by mountain ash (*Eucalyptus regnans*) in Victoria, Australia, with ~1900 t ha^{-1} (Keith et al., 2009). Because coast redwood and mountain ash forest do not cover extensive areas (coast redwood ~17 000 km^2 (Fernandez et al., 2015); mountain ash ~2500 km^2 (Volkova et al., 2018)), their total carbon stocks are not large. Tropical forests, which cover ~25 million km^2 (Saatchi et al., 2011), contain the largest ecosystem carbon stock in the world, with 180-250 Gt in above- and belowground biomass (Saatchi et al., 2011; Baccini et al., 2012; Avitabile et al., 2016). Half of the tropical forest carbon stock grows in the Americas, ~30% in Africa, and ~20% in Asia (Baccini et al., 2012). Within the tropics, tropical evergreen broadleaf forests (rainforests) in the Amazon, the Congo, and Indonesia attain the highest carbon densities, reaching a maximum of 230 t ha^{-1} in the Amazon (Mitchard et al., 2014) and the Congo (Xu et al., 2017). The Amazon contains a carbon stock of 45-60 Gt (Baccini et al., 2012; Mitchard et al., 2014; Englund et al., 2017), with differences in estimates arising from different satellite data sources and methods for calibrating remote sensing data with field measurements of biomass.

The ecosystem with the highest soil carbon density in the world is the Congo Cuvette Centrale swamp forest peatlands, with an average of ~2200 t ha^{-1} (Dargie et al., 2017). Other ecosystems with high soil carbon densities are Arctic tundra, with an average of ~900 t ha^{-1} (Tarnocai et al., 2009), and the mangrove forest peatlands of Kalimantan, Indonesia, with an average of 850 ± 320 t ha^{-1} (Murdiyarso et al., 2015). Arctic tundra, which covers 18 million km^2, contains the largest soil carbon stock in the world, 1400 Gt ± 200 Gt carbon in permafrost to below 3 m depth (Hugelius et al., 2014 2015). Global peatlands contain 530 to 600 Gt carbon (Hodgkins et al., 2018).Tropical peatlands, which cover 440 000 km^2 (Page et al., 2011), contain a substantial soil carbon stock of 100 ± 30 Gt (Dargie et al., 2017). Half of this stock is in the peatlands of Indonesia (Page et al., 2011), one-third is in the Congo, and the rest is in the South and Central America, and in the Pacific (Dargie et al., 2017).

National parks and other protected areas, which cover 20 million km^2, 15% of global terrestrial area (UNEP-WCMC et al., 2018), contain ~90 Gt carbon in vegetation and ~150 Gt carbon in soil, ~20% and 9% of global stocks, and remove carbon from the atmosphere at a rate of 0.5 Gt y^{-1}, ~16% of global removals (Melillo et al., 2016). So, protected areas protect a disproportionate share of global vegetation carbon but account for a proportionate share of sequestration. Protected areas at the strictest level of conservation contain carbon at higher densities and the subset of these in tropical ecosystems contain an aboveground stock of 36 ± 16 Gt C (Collins and Mitchard, 2017).

High biodiversity and ecosystem carbon occur together, with rainforests in the Amazon, the Congo, and Indonesia containing the largest carbon stocks (Saatchi et al., 2011; Baccini et al., 2012; Avitabile et al., 2016) and the highest vascular plant species richness (Kreft and Jetz, 2007) in the world. In tropical old-growth forests, aboveground carbon and tree species richness are correlated for plots of 0.1 ha within a continent (Poorter et al., 2015). Aboveground carbon and tree species richness are not correlated for plots of 1 ha or for plots across continents (Poorter et al., 2015; Sullivan et al., 2017), although aboveground carbon and genus richness are correlated globally (Cavanaugh et al., 2014). Higher species richness tends to increase the variation in plant traits in a community, leading to niche complementarity, higher and more efficient use of sunlight, water, nutrients and other resources, causing higher productivity (Poorter et al., 2015). It may also have a selection effect, an increase in the probability of including a particular species that is highly productive (Poorter et al., 2015). Across the Amazon, approximately 1% of tree species contain 50% of the aboveground carbon (Fauset et al., 2015). The effects of higher species richness may diminish with larger plot size because of species redundancy (Poorter et al., 2015) and not hold globally because of overriding edaphic or climatic factors.

Aboveground carbon in tropical forest shows correlations to vertebrate species richness but probability values of the correlations are not reported (Deere et al., 2018; Di Marco et al., 2018). In logged and burned tropical forest in Brazil, species richness of plants, birds, and beetles increased with carbon density only up to ~100 t ha^{-1} (Ferreira et al., 2018). In temperate zone ecosystems in the lower 48 U.S. states, the relationship of tree species richness to aboveground carbon density was linear and positive for arid and semi-arid climates but increased at low densities and decreased at high densities in dry sub-humid and humid zones (Fei et al., 2018).

Approximately four-fifths of the 1.5 ± 0.7 Gt y^{-1} of carbon emissions from ecosystems (2008-2017) comes from tropical deforestation with most of the remainder from conversion of peatlands (Houghton and Nassikas, 2017; Le Quéré et al., 2018). In tropical forests, ~40% of the aboveground carbon loss from 2000 to 2012 came from managed forests, including plantations and agroforestry areas (Tyukavina et al., 2015). In the Amazon, ~70% of cleared forest from 1990 to 2005 was converted to livestock pasture (De Sy et al., 2015). While carbon emissions from national parks and protected areas, 38 ± 17 Mt y^{-1}, comprise only 1-4% of global emissions, from 15% of global land, a small subset of 10 protected areas produces one-third of all protected area carbon emissions (Collins and Mitchard, 2017).

A dynamic global vegetation model estimated that old-growth forests removed carbon from the atmosphere at a rate of 0.8 Gt y^{-1} (0.7-1 Gt y^{-1}) from 2001 to 2010 and that re-growth of secondary forests removed 1.3 Gt y^{-1} (1-2 Gt y^{-1}) (Pugh et al., 2019), lower than a previous estimate based on national forest cover reports (Houghton et al., 2012). Modeling suggests that the reduction of global burned area from increasing cropland area and fire suppression have increased removals of carbon from the atmosphere 0.1 Gt y^{-1} from 1960 to 2009, one-fifth of total removals from global vegetation (Arora and Melton, 2018).

Total greenhouse gas emissions from fires (2007-2016) was 2.2 Gt y^{-1} (1.8-3.0 Gt y^{-1}) (van der Werf et al., 2017). Of these fire emissions, 0.6 Gt y^{-1} came from forest fires and peat burning (van der Werf et al., 2017), contributing to the global total of 1.5 ± 0.7 Gt y^{-1} carbon emissions (Houghton and Nassikas, 2017; Le Quéré et al., 2018), 0.1-0.4 Gt y^{-1} converted to charcoal and other forms of pyrogenic carbon (Santín et al., 2016; Jones et al., 2019), and the remainder was considered balanced by short-term regrowth of grasses (van der Werf et al., 2017). Burning experiments at 48 forest and savanna sites around the world found that burned plots contained 36 ± 13% less soil carbon than unburned plots 64 years later, with the greatest losses in savanna and broadleaf forest (Pellegrini et al., 2018).

The Amazon was a net carbon emitter of ~0.2 Gt y^{-1} from 2003 to 2005 (Exbrayat and Williams, 2015) and 0.3 ± 0.2 Gt y^{-1} from 2005 to 2008 (Yang et al., 2018), following a severe drought in 2005, and accounted for most of the net carbon emissions of 0.3 ± 0.1 Gt y^{-1} from Neotropical forest from 2003 to 2014 (Baccini et al., 2017). In the Amazon, two-thirds of the carbon emissions from 2003 to 2014 came from increased tree mortality, fire damage, decreased growth, and other forms of tropical forest degradation in the fragments of forest that were not cut (Baccini et al., 2017). Degraded forest can contain carbon at half the density of intact forest, a condition that can persist after 15 years of regrowth (Rappaport et al., 2018). Furthermore, the abundance of tall trees, full canopy, and other forest structure attributes critical for conserving biodiversity recover even more slowly than carbon stocks (Rappaport et al., 2018). Carbon density after drought-induced wildfire can be 60-80% of unburned forest 15 years after burning (Longo et al., 2016) and just 75% after 30 years (Silva et al., 2018). Deforestation and degradation reduce the potential for natural regeneration of forest. Remote sensing of Amazon rainforest regeneration and modelling of potential maximum biomass suggests that forests lost one-third of their regeneration potential from 1993 to 2012 (Exbrayat et al., 2017). In contrast, analysis of field plots at 45 sites across Neotropical forests found that secondary forest regenerated to 90% of its old-growth carbon density in 66 years (Poorter et al., 2015).

In Peninsular Malaysia, Sumatra and Borneo, peatlands emitted ~2.5 Gt carbon from 1990 to 2015 in slow oxidation due to draining for agriculture, converting the peatlands in that time from a sink to a source (Miettinen et al., 2017). Deforestation of mangrove forests emitted 0.07-0.2 Gt y^{-1} of carbon from 1980 to 2005, 10-30% of emissions from deforestation in the country (Donato et al., 2011; Murdiyarso et al., 2015). Across global grasslands, climate controlled NDVI increases across one-third of the area from 1982 to 2011 (Gao et al., 2016b; Gao et al., 2016c).

In Australia, fires consumed nearly one-tenth of net primary productivity (NPP) of natural vegetation from 2000 to 2017 with most in the tropical savannas of northern Australia, where fires consumed nearly half of NPP (Murphy et al., 2019).

In Canada, the USA, and Mexico, wildfire emitted 0.1 ± 0.02 Gt y^{-1} of carbon from 1990 to 2012, but modelling indicates that regrowth balanced this out and produced a net ecosystem carbon balance of 26 ± 5 Mt y^{-1} of carbon (Chen et al., 2017). In California, USA, two-thirds of the carbon emissions from natural ecosystems from 2001 to 2010 came from the 6% of the area that burned (Gonzalez et al., 2015). Carbon emissions of 90 ± 20 Mt y^{-1} exceeded the -20 ± 5 Mt y^{-1} of carbon removals by growth, making California ecosystems a net emitter (Gonzalez et al., 2015).

In the Arctic, increased temperatures have melted permafrost, significantly reducing soil carbon at one tundra site in Alaska 5% y^{-1} from 2009 to 2013 (Plaza et al., 2019). While soil carbon changes have not been quantified across the entire Arctic, a network of boreholes has found an average permafrost temperature increase of 0.39° ± 0.15°C from 2007 to 2016 (Biskaborn et al., 2019) and individual high Arctic sites have recorded increases from 2000 to 2018 of 0.4-0.7ºC in Alaska, USA, 0.7-0.9ºC in Canada, and 0.1-0.8ºC in Russia (Romanovsky et al., 2019), signalling melting and possible carbon losses. Wildfires burning in tundra across Canada and the USA are melting permafrost, up to 0.5 m in depth in Alaska 1930 to 2010 (Brown et al., 2015), exposing the peatland carbon (Brown et al., 2015 2018). Wildfires in the Northwest Territories, Canada, in stands younger than the historic 60-year fire return interval, have begun to consume legacy soil carbon, deposits laid down up to 1600 years ago (Walker et al., 2019). For the State of Alaska as a whole, carbon losses from fires and permafrost melting from 1950 to 2009 were lower than increased vegetative growth, producing a weak carbon sink of 0.4 Mg y^{-1}, although methane emissions caused a net increase in radiative forcing (McGuire et al., 2018).

In Africa, carbon removals from tree growth in miombo tropical dry forest outbalanced emissions from deforestation, producing a weak sink in Zambia from 2000 to 2014 (Pelletier et al., 2018). In the African Sahel, vegetation growth at one semi-arid savanna site in Senegal balances emissions from subsistence pastoralism with livestock at a low density, leading to a neutral carbon balance (Assouma et al., 2019). Drought across sub-Saharan Africa drylands caused tree mortality and vegetation loss with carbon emissions of 50 Mt y^{-1} from 2010 to 2016 (Brandt et al., 2018).

Because of extensive deforestation, forests globally experienced a net loss in total area from 2010 to 2015 (Keenan et al., 2015). Reforestation and afforestation, including establishment of rubber plantations totaled 1.1 million km^2 in the period, with most of this from reforestation in China (Keenan et al., 2015). Six national programs of ecological restoration of forests, shrublands, and grasslands in China covered 1.5 million km^2 and sequestered 0.8 ± 0.08 Gt C in biomass and soil from 2000 to 2010 (Lu et al., 2018).

2.4.4.5 Primary productivity

Anthropogenic climate change can increase vegetation primary productivity through carbon dioxide fertilization and increases in the length of the growing season, as assessed by previous IPCC reports. IPCC WGI AR6 Chapter 5 assesses the place of net primary productivity (NPP) in global biogeochemical cycles. This section assesses ecological aspects of NPP.

Vegetation and soils have removed more CO$_2$ from the atmosphere by photosynthesis than the emissions from deforestation and other land use change since the period 1970-1979, according to carbon accounting, and since the 1990-1999 according to atmospheric inversion models (Le Quéré et al., 2018). This net carbon sink has since been increasing, according to dynamic global vegetation models (Huntzinger et al., 2017; Liu et al., 2019b) and atmospheric inversion models (Peylin et al., 2013; Ciais et al., 2019), and both types of models indicate that increasing atmospheric CO$_2$ is the primary cause of the increase (Schimel et al., 2015b; Fernandez-Martinez et al., 2019), although no formal attribution analysis has been conducted.

Analysis of eddy covariance flux data indicated that global gross primary productivity (GPP) was 123 ± 8 Gt y^{-1} from 1998 to 2005, that tropical forests produced one-third, and that precipitation primarily controlled NPP (Beer et al., 2010). Analysis of near infrared remote sensing indicated GPP of 147 ± 16 from 2003 to 2015 (Badgley et al., 2019). The difference, between the two estimates does not indicate a temporal trend, but a difference in methods.

The Normalized Difference Vegetation Index (NDVI) is a remotely-sensed indicator of green leaf surface area that is correlated with net primary productivity in vegetation with one canopy layer (Tucker, 1979). Analysis of NDVI from 1982 to 2009 indicates that global average leaf area index increased in the period, a greening that is consistent with CO$_2$ fertilization (Zhu et al., 2016). The global NDVI time series only increased in the first part of the remote sensing time series, however, from 1982 to 1998, and stay flat from 1999 to 2015 (Yuan et al., 2019). During the second period, an increase in the atmospheric vapour pressure deficit signalled an increase in aridity that constrained NPP. Globally, water availability (Jung et al., 2017), terrestrial water storage (Humphrey et al., 2018), and drought (Yu et al., 2017b) drive interannual changes in GPP. Free-air CO_2 enrichment and chamber experiments indicate that nitrogen and phosphorus constrain CO_2 fertilization (Terrer et al., 2019). In the Amazon, phosphorus availability constrains plant response to CO_2 fertilization (Fleischer et al., 2019).

The increase in NDVI from 1982 to 2012 suggests a 16% decline in global vegetated area where growth is limited by temperature (Keenan and Riley, 2018). NDVI increased in boreal tundra in Russia (Miles et al., 2019), across China from 1982 to 2009 (Piao et al., 2015), and at the boreal forest-tundra treeline in Canada (Bolton et al., 2018).

NDVI did not increase at other sites around the world. In southwest Australia, NDVI in Mediterranean vegetation declined from 2000 to 2011, particularly at the shrub-tree ecotone, indicating a potential biome shift driven by wildfire (Brouwers and Coops, 2016). In France, beech (*Fagus sylvatica*) forest showed decreased enhanced vegetation index (EVI) for three years following severe drought (Vicca et al., 2016). In addition, field-measured growth in temperate mixed forest in the eastern U.S. did not show the climate and CO_2 sensitivities of the NPP models (Rollinson et al., 2017). In the Cerrado savanna of Brazil, agricultural abandonment or expansion could explain approximately half the area of EVI increase from 2001 to 2015 (Rosan et al., 2019).

Primary productivity has also changed in lakes. As metabolic costs increase at higher water temperatures, primary production should decrease with climate warming. However, global warming reinforces the eutrophication of already eutrophic lakes through the prolongation of thermal stratification, particularly the development of nuisant cyanobacteria blooms (Domis, 2013) (Adrian and Hessen, 2016). Moreover, nutrient depleting in the upper layers (epilimnion) can be compensated for by a shift towards buoyant cyanobacteria species capable of nitrogen fixation (Huber et al., 2012). Conversely, in nutrient-poor large lakes, where internal nutrient loading via vertical mixing is often the primary nutrient source, warming-induced prolonged thermal stratification can lead to a reduction in primary production (Kraemer et al., 2017). Within-lake variability in algal biomass in 18 globally distributed lakes increased with increasing variability in wind speed (Rusak, 2018). This overall increase in variability in algal mass production related to climate change impacts decreases the predictability of the services lakes provide.

2.4.5 Conclusions on Observed Impacts

The consistency of patterns of biological change with expectations from regional or global warming processes, coupled with an understanding of underlying processes, the coherence of these patterns at both regional and global scales, all form multiple lines of evidence (Parmesan et al., 2013) that it is *very likely* that observed range shifts and phenological changes in individual species can be attributed to regional and global climate changes *(very high confidence)* (Table 2.2) (Parmesan et al., 2013).

New studies since AR5, with more sophisticated analyses designed to capture complex responses, indicate that past estimates of the proportion of species impacted by recent climate change have been underestimates due to their unspoken assumptions that local or regional warming should lead solely to poleward/upward range shifts and advancements of spring timing (Duffy et al., 2019). More complex analyses have documented cases of winter warming driving delayed spring timing of northern temperate species due to chilling requirements, and increased precipitation driving species' range shifts downward, eastward and westward in arid regions *(high confidence)*. Further new studies have shown that phenological changes have, in some cases, successfully compensated for local climate change and reduced degree of range shifts *(medium confidence)*. Limited number of studies of this type make it difficult to estimate the generality of these effects globally.

Responses in freshwater species are consistent with responses in terrestrial species, including poleward and upward ranges shifts, earlier timing of spring plankton development, earlier spawning in fish, and extension of the growing season. Observed changes in freshwater species are strongly related to anthropogenic climate change (ACC)-driven changes in the physical environment (e.g. increased water temperature and reduced ice cover). Prolonged thermal stratification, which has also been linked to ACC, has led to divergent responses in lakes, with already eutrophic lakes becoming more eutrophic *(very high confidence)* and nutrient-poor lakes becoming more nutrient limited *(medium confidence)*. In boreal coniferous forest, there has been an increase in terrestrial derived dissolved organic matter transported into rivers and lakes as a consequence of climate change (that has induced increases in run-off and greening of the northern hemisphere), as well as to changes in forestry practices. This has caused waters to become brown and more opaque, with complex positive and negative repercussions on water temperature profiles (lower vs upper water) and on primary *production (high confidence)*.

Field research since the IPCC Fifth Assessment Report has detected biome shifts at numerous sites, poleward and upslope, that are consistent with increased temperatures and altered precipitation patterns driven by climate change, and support prior studies that attributed such shifts to anthropogenic climate change *(high confidence)*. These new studies help fill prior geographic and habitat gaps, for example documenting upward shifts in the forest/alpine tundra ecotone in the Andes, Tibet and Nepal, and northward shifts in the deciduous/boreal forest ecotones in Canada. Globally, woody encroachment into open areas (grasslands, arid regions and tundra) is *likely* being driven by climate change and increased CO_2 in concert with changes in grazing and fire regime *(medium confidence)*.

[INSERT TABLE 2.3 HERE]

Table 2.3: Confidence in detecting and attributing observed changes in terrestrial and freshwater species and systems to climate change. [[PLACEHOLDER FOR SECOND ORDER DRAFT: table to be expanded with continued literature search]. Summary table is fully detailed in Table 2.A.1. Lines of evidence for attribution of observed changes to climate change and increased CO_2 are used to support stated confidence in attribution of key statement on observed biological changes to climate change and increased atmospheric CO_2. Icons represent lines of evidence.

2.5 Risk Assessments for Species, Communities, Key Ecosystems and their Services

The Risk Assessment Framework was recently introduced in the IPCC AR5 (2014) but not defined in IPCC AR4 (2007). Risk means the probability of harmful consequences resulting from climate change. Risk results from the interaction of vulnerability, exposure, and hazard (see Chapter 1). Risk is often represented as the probability of occurrence of hazardous events or trends multiplied by the impacts if these events or trends occur (IPCC, 2014).

Using the risk assessment framework above, this chapter defines vulnerability as a pre-existing condition, incorporating the extent to which biodiversity is susceptible to, or unable to cope with, the adverse effects of climate change related to spatially explicit effects. Adaptive capacity, then, includes consideration of multiple traits at the species level, including physiological constraints, capacity for behavioural adaptation, potential for genetic change, dispersal abilities, capacity for distributional change, and resource constraints.

2.5.1 Observed and Projected Feedbacks Between Climate and Ecosystems

The possibility of feedbacks and interactions between climate drivers and biological systems or ecological processes was identified as a significant emerging issue in AR5. It is virtually certain that land cover changes affect regional and global climate through changes to albedo, evapotranspiration and roughness (*very high confidence*) (Perugini et al., 2017). There is growing evidence that biosphere-related climate processes are being affected by climate change in combination with disturbance and land use change (*high confidence*). It is virtually certain that land surface change caused by disturbances such as forest fire, hurricanes, phenological changes, insect outbreaks and deforestation affect carbon, water, and energy exchanges, thereby influencing weather and climate (*very high confidence*) (Bright et al., 2013; Brovkin et al., 2013; Naudts et al., 2016; Prăvălie, 2018) (Table 2.4, Figure 2.6).

Feedbacks among climate warming, permafrost thaw, nutrient dynamics and carbon cycling are complex, but have the potential to accelerate vegetation change and influence the global climate system (Myers-Smith et al., 2011; Loranty and Goetz, 2012; Pearson et al., 2013; Crowther et al., 2016; Grosse et al., 2016; Turetsky et al., 2019) (medium evidence, high agreement). Tundra canopy heights are increasing (Bjorkman et al., 2018), which could result in albedo, soil temperature and permafrost thaw feedbacks (Sturm et al., 2005; Myers-Smith et al., 2011; Loranty and Goetz, 2012; Pearson et al., 2013; Frost et al., 2018; Addis and Bret-

Harte, 2019). Increased plant litter deposits into soils from increasing tundra plant biomass could influence tundra carbon cycling and storage (Cornelissen et al., 2007; Buckeridge et al., 2010; Myers-Smith et al., 2019a). With up to 80% of tundra biomass found below-ground (Iversen et al., 2017), below-ground vegetation and microbial responses may determine tundra soil carbon dynamics and the resulting soil carbon feedbacks (Sistla et al., 2013; Crowther et al., 2016; Xue et al., 2016; Van Gestel et al., 2018). Recent increases in high latitude and tundra fires and the charcoal record in lake sediment showing high fire frequency during warmer periods over the past 10,000 years suggest that fires may become more common in future tundra ecosystems (Mack et al., 2011; Hu et al., 2015). Ecological feedbacks that further accelerate the terrestrial carbon cycle in the Arctic could make substantial contributions to carbon emissions and global climate warming (Mack et al., 2011; Hugelius et al., 2013; Pearson et al., 2013; Crowther et al., 2016; Xue et al., 2016; Turetsky et al., 2019).

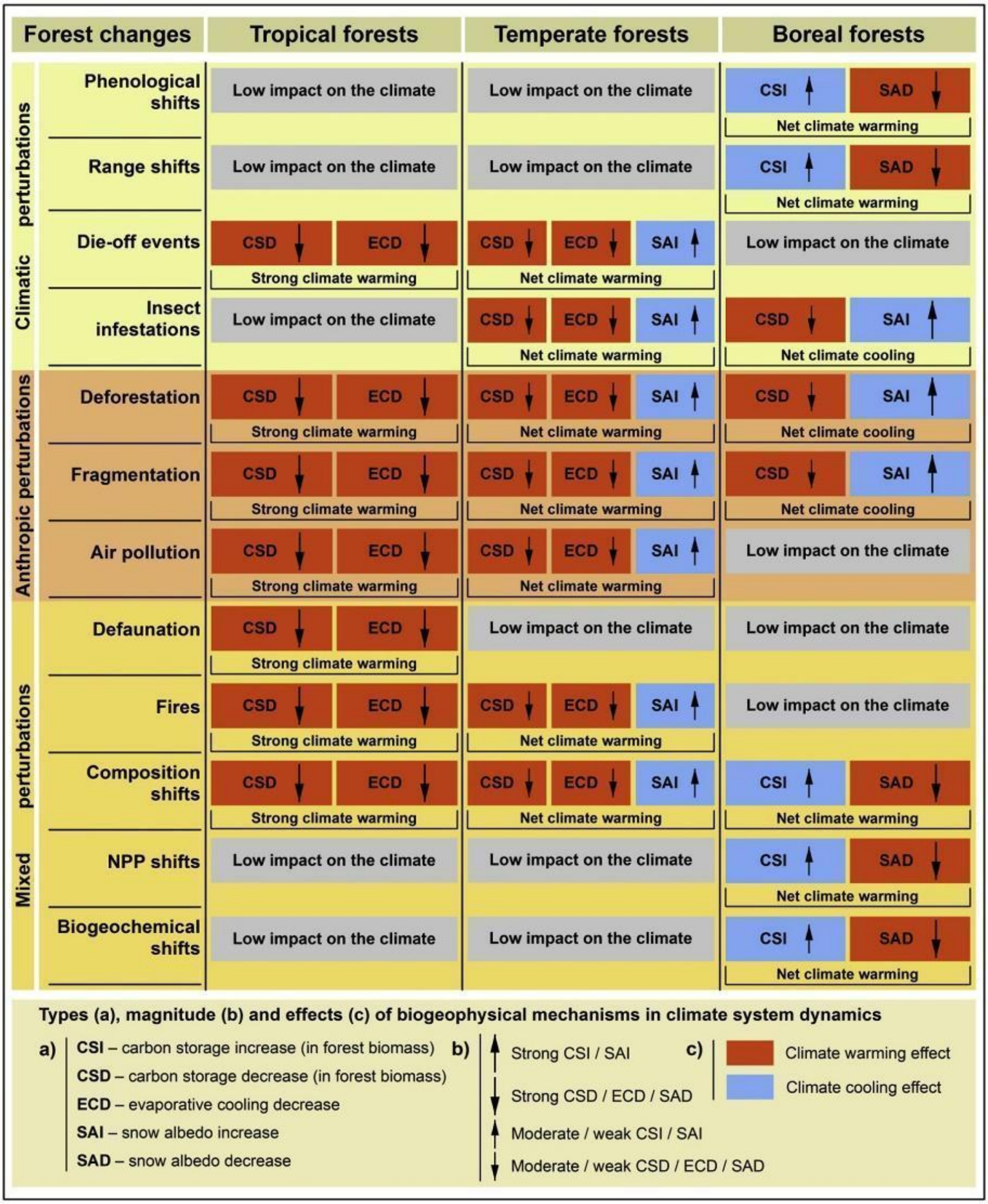
There is growing evidence that freshwaters are hotspots for C transformation, sources of CO_2 and CH_4 and, in some cases, carbon sinks (Bastviken et al., 2004; Battin et al., 2009; Tranvik et al., 2009; Aben et al., 2017; Li et al., 2018a; Bartosiewicz et al., 2019; Jansen, 2019; Marcé et al., 2019). This is particularly so for highly polluted lakes, dried river beds, reservoirs and farmland ponds, which are now being recognised as very large sources of methane due to significant organic loadings from sewage and agriculture and enhanced microbial activity in rewetted river beds (Badiou et al., 2019; Beaulieu et al., 2019; Marcé et al., 2019; Ortega, 2019). As aquatic productivity increases with ongoing climate change and population pressure, increases in CH_4 emissions from freshwater systems are expected (Tranvik et al., 2009; DelSontro et al., 2018; Bartosiewicz et al., 2019; Beaulieu et al., 2019).

Feedbacks from terrestrial and aquatic ecosystems to atmospheric CO_2 concentrations and the interactions between climate drivers and human disturbance contribute substantial uncertainty to projections of future climate (Brovkin et al., 2013; Schimel et al., 2015b; Berzaghi et al., 2018; Marcé et al., 2019; Scharfenberger et al., 2019). These feedbacks can be positive or negative feedbacks (i.e. climate cooling effects) (Lemordant and Gentine, 2019), vary spatially and seasonally (Duveiller et al., 2018), and act over large geographic areas and long time periods, making them difficult to observe and quantify directly (Schimel et al., 2015a). The relative magnitudes of the feedbacks remain uncertain (Loranty et al., 2014; Alkama and Cescatti, 2016), while the regional effects of disturbance are increasingly being studied (e.g. REFS), the strength of biophysical feedbacks at the global scale remain uncertain (Keenan and Riley, 2018; Liu et al., 2019b). As a result, projections of the atmosphere–land CO_2 flux into the future are highly uncertain, with no agreement even on the sign of the flux by the end of the century (Friedlingstein et al., 2014; Sitch et al., 2015; Bonan and Doney, 2018).

For example, changes in vegetation phenology and productivity associated with global warming and CO_2 fertilization effects (Mao et al., 2016), in combination with land management (Wang et al., 2018), have important implications for the terrestrial water cycle (Zeng et al., 2018a; Zeng et al., 2018b), ecosystem functioning and global carbon cycling. Observed global vegetation changes (“global greening”) has been associated with increased global evapotranspiration, a key component of the global water cycle and energy balance which influences global rainfall, temperature, and atmospheric motion (Zeng et al., 2017). Similarly, large-scale tree mortality associated with high temperatures, water stress and/or insect outbreaks (e.g. Mountain pine beetle, *Dendroctonus ponderosae*; Asian longhorn beetle (*Anoplophora glabripennis*)) alter biogeochemical and biogeophysical processes in forests (Landry et al., 2016), by decreasing leaf area index and gross primary productivity, leading to reduced evapotranspiration and increased land surface temperature (Bright et al., 2013). Insect outbreaks reduce terrestrial carbon uptake and storage (Arora et al., 2016) and have consequences for surface albedo, although some impacts may be counteracted by long-term forest regrowth (Ghimire et al., 2015).

[INSERT TABLE 2.4 HERE]

Table 2.4: Terrestrial and freshwater ecosystem feedbacks which affect the Earth's climate system dynamics; following (Prăvălie, 2018).



	Forest changes	Tropical forests	Temperate forests	Boreal forests
Climatic perturbations	Phenological shifts	Low impact on the climate	Low impact on the climate	CSI ↑ SAD ↓ Net climate warming
	Range shifts	Low impact on the climate	Low impact on the climate	CSI ↑ SAD ↓ Net climate warming
	Die-off events	CSD ↓ ECD ↓ Strong climate warming	CSD ↓ ECD ↓ SAI ↑ Net climate warming	Low impact on the climate
	Insect infestations	Low impact on the climate	CSD ↓ ECD ↓ SAI ↑ Net climate warming	CSD ↓ SAI ↑ Net climate cooling
Anthropic perturbations	Deforestation	CSD ↓ ECD ↓ Strong climate warming	CSD ↓ ECD ↓ SAI ↑ Net climate warming	CSD ↓ SAI ↑ Net climate cooling
	Fragmentation	CSD ↓ ECD ↓ Strong climate warming	CSD ↓ ECD ↓ SAI ↑ Net climate warming	CSD ↓ SAI ↑ Net climate cooling
	Air pollution	CSD ↓ ECD ↓ Strong climate warming	CSD ↓ ECD ↓ SAI ↑ Net climate warming	Low impact on the climate
Mixed perturbations	Defaunation	CSD ↓ ECD ↓ Strong climate warming	Low impact on the climate	Low impact on the climate
	Fires	CSD ↓ ECD ↓ Strong climate warming	CSD ↓ ECD ↓ SAI ↑ Net climate warming	Low impact on the climate
	Composition shifts	CSD ↓ ECD ↓ Strong climate warming	CSD ↓ ECD ↓ SAI ↑ Net climate warming	CSI ↑ SAD ↓ Net climate warming
	NPP shifts	Low impact on the climate	Low impact on the climate	CSI ↑ SAD ↓ Net climate warming
	Biogeochemical shifts	Low impact on the climate	Low impact on the climate	CSI ↑ SAD ↓ Net climate warming

Types (a), magnitude (b) and effects (c) of biogeophysical mechanisms in climate system dynamics

a)
CSI – carbon storage increase (in forest biomass)
CSD – carbon storage decrease (in forest biomass)
ECD – evaporative cooling decrease
SAI – snow albedo increase
SAD – snow albedo decrease

b)
↑ Strong CSI / SAI
↓ Strong CSD / ECD / SAD
↑ Moderate / weak CSI / SAI
↓ Moderate / weak CSD / ECD / SAD

c)
🟥 Climate warming effect
🟦 Climate cooling effect

Figure 2.6: Perturbations and examples of implications (biogeophysical mechanisms) on climate system dynamics (warming/cooling) for the three global forest biomes - (adapted from Fig. 5, (Prăvălie, 2018)). [PLACEHOLDER FOR SECOND ORDER DRAFT: open ecosystems and freshwater ecosystems to be added]

2.5.2 Protected Areas

National parks and other protected areas conserve the most intact ecosystems in the world, with the global system of protected areas now covering 15% of global terrestrial area (UNEP-WCMC et al., 2018). These areas are threatened by deforestation, agricultural expansion, and urbanization to the extent that one-third of global protected land is under intense human pressure (Jones et al., 2018). Gaps in the global system of protected areas leave mountain elevational gradients, which are key to facilitating upslope shifts of species and biomes under climate change, insufficiently protected, with protection absent in 40% of mountain ranges (Elsen et al., 2018). Mountain ranges in Africa and Asia are least protected (Elsen et al., 2018).

Climate change exposes protected areas to potentially higher levels of hazard. In the United States, spatial analyses of 1895-2010 climate trends show that anthropogenic climate change has disproportionately exposed the U.S. National Park System to hotter and drier conditions, with temperatures increasing at twice the national rate and precipitation decreasing on 12% of the area, compared to 4% for the U.S. as a whole (Gonzalez et al., 2018). For the protected areas of Mexico, U.S., and Canada, over half the current protected area does not have a climate analogue under RCP8.5 that is also protected and only 6-8% of the total protected area is located in refugia (Batllori et al., 2017). Across North and South America, the protected areas in the equatorial zone are exposed to the highest climate velocities under emissions scenario A2 (Carroll et al., 2015).

Nevertheless, protected areas can occupy a key position in a landscape under climate change. Across North America, protected areas cover just one-tenth of the land, but conserve one-quarter of climate refugia (Michalak et al., 2018). The protected areas of Québec, Canada, would be the recipients of substantial numbers of plants and animals whose range would shift north under climate change, with half of the protected areas potentially experiencing >80% species turnover under emissions scenario A2 (Berteaux et al., 2018). In the protected areas of Finland, spatial analysis of the southern edge of the range of bird species shifting north from the 1970s to the 2000s and the northern edge of species shifting from the south showed that bird abundance was higher inside protected areas than outside, indicating that the areas may be effectively facilitating range shifts under climate change (Lehikoinen et al., 2019).

2.5.3 Projected Impacts at Species and Community Levels

2.5.3.1 Assessment of models and sources of uncertainties

A wide range of methods have been developed for projecting the impacts of climate change on biodiversity into the future. They can be classified into three main approaches: 1) statistical models (e.g. species distribution models, (Elith and Leathwick, 2009)); 2) mechanistic / process-based models; and 3) trait-based (Pacifici et al., 2015).

The species distribution models (SDMs) or niche-based models predict the potential geographic areas of suitable climate for the species in historical, current, and as its future climate preferences (Trisurat, 2018; Vieira et al., 2018). There are limitations in all models and it is critical that modellers understand the assumptions, proper parameterization of the model, and the limitations of whichever model technique is used. This includes a sound understanding of the differences in the climate models, emission scenarios or representative concentration pathways, and what baseline was used. As there are limitations in the biodiversity modelling techniques, there are also limitations in what one can (and should not) do with climate model data, especially in terms of understanding bias correction and downscaling limitations.

There are several systems that automate the development of species distribution models and projections into the future (e.g., (Hallgren et al., 2016)) as well as various R packages and libraries. There is also an increasing amount of guidance on how to select statistical models (Beaumont et al., 2016), other model types (Foden et al., 2019) and on how to properly use climate model data (Suggitt et al., 2017). Buisson 2010 (Buisson et al., 2010) found that model algorithms explain most of the model deviance, followed by GCMs across 2020, 2050 and 2080. Standards to apply in species distribution modelling have also been put forward (Araujo et al., 2019). For freshwater systems, SDMs require a specific design to be used that accounts for the longitudinal connectivity and up/downstream processes along the water network. However, only recently have freshwater-specific network-based models been more widely used (Peterson et al., 2013), superseding grid-based (and spatially-implicit) versions that originated from the terrestrial realm.

It is important to consider that none of these modelling techniques are predictions of the future, they are projections of possible futures. Perhaps the best way to think about the outputs of all impact models is that they are hypotheses of what a future world might look like if the climate in the climate models come to pass. As most published results are for an ensemble of projections then they will be an approximation for a mid-point of a range of models. Computational limits, basic ecological understandings of interactions and funding limit how far and how fast these modelling techniques can advance.

Many of the limitations of the models are well known, including the absence of interactions from many models, as well as the sometimes limited inclusion of traits in species distribution models. These can play an important role in shaping species' sensitivity and adaptive capacity to climate change (Dawson et al., 2011). Suggestions have been made on how to start bringing more biotic interactions in species distribution models have recently been published (Early and Keith, 2019). However, this assumes that detailed knowledge of how species interact is known. In addition, these models perform poorly for endemic and restricted species, which are normally listed as threatened species (Platts et al., 2014).

Species traits alone have been used to estimate potential climate change impacts (Foden et al., 2013; Cizauskas et al., 2017). Many species distribution models bring in some traits in explicitly like dispersal (Warren et al., 2018). What is less well understood is that statistical species distribution modelling techniques incorporate some traits implicitly. All SDMs work off of occurrences of a species at a series of points. What is being modelled is the relationship between the points and the climate estimated at those points. While there is an assumption that this is some measure of the physiology/physiological limits of the species (and, in many cases, may very well be) it is also a proxy for everything else at that point (e.g., habitat, food resources). Failure to understand this can lead to model errors – for example, when the vegetation and species are both modelled at a single point in the same model. Traits, such as where a species is normally located (canopy, mid-storey, understorey) are all captured in implicitly in the model as the species occurrence was likely from that location. Thus the difference between the understorey and the canopy in terms of climate would implicitly be differentially in the model (including shading effects). Thus, when increases shading is brought up as a potential adaptation for a species, or when statements are made about canopy species being more exposed to the climate, they are missing the point that these 'partials' are in the overall model, they just cannot be parsed out in normal techniques.

Statistical models are independently tested as part of the model development process. However, in terms of considering any model output as a hypotheses, tests should also be made against observations of changes in range (for example). Climate changes in many parts of the world are such that it should be possible to test model projections against observations. One problem with this is that many biodiversity model projections have been for higher levels of climate change. It is only recently that models have been developed looking at smaller amounts of climate change, like 1.5°C (Hoegh-Guldberg et al., 2018; Warren et al., 2018). However, there have still only been a few studies looking at how well models are performing in terms of observations, and many of these have been on islands which increase the difficulty in assessing model performance (Fordham et al., 2018).

Mechanistic approaches, also known as process-based models predict species' responses to changing environmental conditions by explicitly incorporating known biological processes, thresholds and interactions (Morin and Thuiller, 2009; Maino et al., 2016). Input data for the mechanistic niche models are typically obtained from laboratory and field observations or from mathematic equations. Mechanistic models are able to accommodate a broad range of climate change impact mechanisms and include species-specific characteristics such as dispersal distances, longevity, fecundity, genetic evolution, phenotypic plasticity. However, physiological, demographic and distribution knowledge is restricted to only a few well-studied species.

Most models are at large scales (20km–50km) with some local smaller scales. However, within these larger areas there will be areas that become warmer sooner, and others that will remain cooler, longer (e.g., higher elevations, near water, shaded hillsides). Studies have shown that, locally, micro-refugia can offset some levels of warming (Suggitt et al., 2015; Suggitt et al., 2018).

2.5.3.2 Risk Assessment as Applied to Biodiversity

For risk assessment at the species and community levels, non-modeling approaches based on known underlying biological traits or processes, as well as expert opinion are used to temper model outputs with ground-based validation, adding realism and reliability to risk assessments. Trait-based assessment approaches use species' biological characteristics as predictors of extinction risk due to climate change biological characteristics to predict species' sensitivity and adaptive capacity to climate change. Climate exposure can be estimated using GIS-based modelling, statistical programs or expert judgment (Chin et al., 2010). The trait-based approaches are widely applied to all taxonomic groups and across the regions and are useful for broad-scale conservation because they do not require modelling expertise (Pacifici et al., 2015; Willis et al., 2015). Limitations include the difficulty in knowing where to set thresholds for all assessed species and not knowing which traits are the most important. Furthermore, most methods don't allow direct comparison of vulnerability and risk among taxonomic groups and remain largely unvalidated.

Recent studies combined either two or three approaches for climate change risk assessment of biodiversity in order to capture the advantages of each and avoid the limitations. (Warren et al., 2013) used combinations of SDMs and trait-based approaches to estimate the proportions of species losing their climatically suitable range under the various future climate and dispersal rate scenarios. Similarly, Garcia (2014) combined the spatial projections of climate change exposure with traits to assess vulnerability of sub-Saharan African amphibians. Laurance (2012) combined 31 functional groups of species and 21 potential drivers of environmental change to assess both the ecological integrity and threats for tropical protected areas on a global scale. Keith (2014) used the combination of three approaches (SDMs-trait-mechanistic) to determine how long before extinction a species would become eligible for listing as threatened based on the IUCN Red List criteria. The results show that the combined approaches were more sensitive to climate change than using a single criterion.

2.5.3.3 Risk of Species' Extinctions

This assessment of current findings is of studies across a range of taxa and using as many modelling techniques as possible to allow for a better synthesis of results. [[PLACEHOLDER FOR SECOND ORDER DRAFT:: findings will be updated with continued literature search].

2.5.3.3.1 Overview

Extinction risk estimates whether or not a particular species may be at risk of extinction over the coming decades if climatic trends continue. There is often a delay between environmental conditions deteriorating and a species going locally extinct, particularly for long-lived species such as many large mammals, some fish, and trees. Individuals can continue to survive long after conditions for successful reproduction are gone, a state termed "extinction debt". This is sometimes termed "climate debt" when climate change is implicated in degradation of the affected species' habitat.

Since the AR5 there have been 2,959 papers that had the words "extinction" and "climate change" in their title, abstract or keywords (Web of Science search, 17 April 2019 for the years starting in 2014). The number of papers on the topic have increased from 486 published in 2014 to 634 published in 2018.

2.5.3.3.2 Projections for Freshwater Biodiversity

Freshwater comprises only 0.01% of the water on earth and freshwater wetlands encompasses about 5.4-6.8% of the global surface area (Lehner and Doll, 2004), but hold 9.5% of the Earth's described animal species (Freshwater Animal Biodiversity Assessment; (Balian et al., 2008). More than 29% of the 25,007 freshwater species assessed on the IUCN Red List (www.iucnredlist.org) are globally threatened with extinction (Baillie and Butcher, 2012). Amphibians and freshwater fish are among the groups most threatened to go extinct, with climate change cited as a primary factor through changes in water temperatures, stream flow, loss of cold water habitat, increased variability of precipitation, and increased disease risk from warming temperatures (Reid et al., 2019). These add to stresses from changes in land use (affecting run-off and material transfers from land to water) and overexploitation (IPBES, 2019). Global scenarios of freshwater fish indicate that by 2075, global climate change and its associated impacts on hydrology may lead to loss of local fish biodiversity of up to 75% (Xenopoulos et al., 2005).

Biogeography of fish is strongly linked to local climatic conditions, given their ectothermic (none heat-producing) nature. Climate change potentially threatens ~50% of global freshwater fish species (Durwall, 2016) and 33% of the European freshwater fish species (Janssen et al., 2016). Fish in Europe are especially susceptible to climate change in Mediterranean regions with warmer climate and which are exposed to recurrent flood and drought events and critically surpassed temperature thresholds (Santiago et al., 2016; Jaric et al., 2019). Out of 20 highest ranking climate-susceptible species, 12 are endemic to Greece, one endemic to Greece and southern Albania, and seven to the Iberian Peninsula (Jaric et al., 2019). A global assessment of the vulnerability of freshwater fish to climate warming has been shown to be largest for fish species in the northern hemisphere (Comte and Olden, 2017). The study also points to the overlap of climatic and non-climatic drivers of change threatening fish species particularly located in freshwater basins in southern Europe, southeast North America and central Asia (Comte and Olden, 2017).

Cold-water habitats and associated obligate species are particularly vulnerable to potential impacts of climate change. Loss in cold water habitat has been documented and projected for cold water fish such as salmon and salmonids (Santiago et al., 2016; Merriam et al., 2017; Fullerton et al., 2018). However, increased discharge was projected to offset the effects of increased air temperature with no consistent loss of suitable brook trout habitat by the end of the 21st century within the Appalachian watershed. Loss of cold water habitat at the river network-scale was related to periods of low flow regimes. As such below tributary refugia are essential for metapopulations (groups of individual populations connected on a decadal time scale) to persist (Merriam et al., 2017). Thermal habitat in mountain streams, where water temperature is buffered against increases in air temperature, are important thermal refuges for cold water species. Mountain landscapes will thus play an important role in the preservation of cold water habitat (Isaak et al., 2016).

Ensemble distribution models across North America (Shah et al., 2014) indicated that stream macroinvertebrate taxa (at the genus level) would track their preferred climatic conditions; models show a distinct northward shift, depending on the emission scenario. Likewise, (Pyne and Poff, 2017) assessed the site-specific extirpation likelihood for 88 aquatic insect taxa, and projections showed that climate-change induced hydrological alteration would result in a 30–40% loss of taxa in warmer, drier ecoregions and 10–20% loss in cooler, wetter ecoregions. Continental ensemble model projections across Australia indicate that across 85% of all Odonates, a high degree of range contractions of 56–69% (RCP 6.0, medium emissions scenario and RCP 8.5, high emissions scenario, respectively) by 2085 would be possible that can be attributed to anthropogenic climate change (Bush et al., 2014).

Likewise, ensemble projections across two spatial scales and two emission scenarios with projected changes temperature and precipitation until 2080 in Europe (regional (Domisch et al., 2011) and continental (Domisch et al., 2013) showed that range shifts of stream macroinvertebrates may occur along the river network, and into North-eastern direction, possibly leading to species turnovers and altered community structures. Here, community turnovers would be expected as cold-adapted species are projected to lose, while warm-adapted species are predicted to gain climatically suitable habitat (Domisch et al., 2011; Domisch et al., 2013). While a number of warm-adapted species may experience range expansions, both along the network and across Europe, the majority of species were predicted to lose climatically suitable areas by on average 38–44%, depending on the emission scenario (A2a and B2a). Especially endemic species (here: occurrence restricted to only one ecoregion) showed a distinct pattern reading possible range contractions, irrespective on their temperature preference (Domisch et al., 2013).

Based on catchment-scale species data and climate projections (Markovic et al., 2014) projected a threat by the 2050s for 1648 European freshwater plants, fishes, molluscs, odonates, amphibians, crayfish and turtles. Climate change will most likely cause a decrease in habitat suitability across the current range area by the 2050s for the vast majority of studied freshwater species, combined with a north-eastward shift in species distributions. Molluscs, noted for their limited dispersal capability, are predicted to be the most at risk group. The negative effects of climate change are most prominent for rare species in all taxonomic groups which face the ‘double jeopardy’ of being both rare and vulnerable to climate change (Jaric et al., 2019). Moreover, aquatic insects (Ephemeroptera, Plecoptera, and Trichoptera) of southern European ecoregions are most endangered in terms of potential vulnerability to climate change based on a study covering 23 Euorpean ecoregions (Conti et al., 2014). Currently protected area in Europe are most likely not sufficient to provide habitat for the majority of rare molluscs and almost half of the rare fish species (Markovic et al., 2014).

In addition, integrated protection schemes are necessary (Abell et al., 2007) that account for the longitudinal connectivity of freshwater habitats. For instance, in a global assessment, (Abell et al., 2017) found that ca. 70% of river reaches (by length) have no protected areas in their upstream catchments, and only 11.1% (by length) achieve full integrated protection. Given this point of departure, shifting climatic / hydroclimatic conditions, and species tracking their climatic envelope (Parmesan and Yohe, 2003), are likely to further decrease protection for threatened species.

Especially for threatened species, such as the relict Himalayan dragonfly (*Epiophlebia laidlawi*) which is endemic in Asia to the Hindu Kush Himalayan region, ensemble models showed that possible range contractions would be contingent on climate change (Shah et al., 2012). (Kuemmerlen et al., 2015) showed with model projections of 72 stream macroinvertebrate taxa in China that it is critical to jointly account for climate change and possible land use changes in the models, given that both climate change and land use change would have detrimental effects on species richness and range size when viewed in isolation, but with variable effects when combined.

2.5.3.3.3 Global Projections of Extinction Risk

Global: Over time the different assessment reports have used different criteria and confidence levels, based on different bodies of literature, to estimate future extinction risk. In most of these cases the risk has been assessed from the literature and based on overall range contractions. In AR4, the extinction risk was carefully quantified based on the literature of the time and was projected to be "There is *medium confidence* that approximately 20–30% of species assessed so far are *likely* to be at increased risk of extinction if increases in global average warming exceed 1.5–2.5°C (relative to 1980 to 1999). As global average temperature increase exceeds about 3.5°C, model projections suggest significant extinctions (40–70% of species assessed) around the globe. While no specific amount of range contraction is tied specifically to these estimates they approximately correspond to 50% reductions in range size (IPCC, 2007). In AR5, the percentages were removed and the text stated "a large fraction of terrestrial and freshwater species face increased extinction risk under projected climate change during and beyond the 21st century, especially as climate change interacts with other pressures…(*high confidence*)" (Field et al., 2014). The confidence was felt to be higher as no exact number was given and other stressors were taken into account.

A series of global analyses came out towards the end of the Fifth Assessment Report, using a variety of different modelling techniques (both statistical models and trait-based approaches). These were summarized by Urban (2015) who examined 131 studies modelling the potential impacts of climate change on extinction risks using a range of modelling techniques (Figure 2.7). Urban found high variability among individual studies, with percentages of species predicted to suffer extinction varying from zero to 54% according to the geographic locations and taxonomic identities of the species included, plus the assumptions made in each study about future climates, climatic debts, the thresholds of percent of range lost used for extinction risk, and effectiveness of dispersal. Differences in estimates of extinction risk stemmed from differing assumptions of thresholds for extinction risk and differing emissions scenarios, as well as from differing geographic regions and taxonomic groups, and differing modelling approaches. Overall, approximately 5% of the species were estimated to become extinct at 2°C warming, 8.5% at 3°C and 16% at 4.3°C (all relative to pre-industrial) Urban (2015). These percentages seem much lower than in previous IPCC reports.

One reason for differing estimates of species' at risk of extinction may be that differing levels of climatic debt were assumed in the studies. Climatic debt occurs when a species has lost so much habitat that it still exists, but is doomed to extinction. The most frequent assumptions found by Urban (2015) were that extinction was assumed inevitable after loss of 80%, 95%, or 100% of habitat (if 100% must be lost before the species is extinct, there is no current debt). Changing the threshold from 100% to 80% increased estimated extinction risk from 5% to 15%. At lower levels (e.g., 50%) the risk is even higher. In one recent study examining more than 100,000 terrestrial species the percentages of species projected at risk of extinction through losing more than 50% of their range was 49% for insects, 44% for plants, and 26% for vertebrates at ~3°C global rise in temperature (Warren et al., 2018). Those estimates dropped considerably at lower levels of warming, down to 18%, 16%, and 8% at 2°C; and 6%, 8% and 4% at 1.5°C (Warren et al., 2018).

Finally, while dispersal may benefit individual species, in terms of extinction risk, it poses additional risks to communities and ecosystems as interactions between species are changed or eliminated. Even a low level for global extinction risk is clearly an indicator of much higher local extinction risks as species' ranges shift (mostly contract) to follow their suitable climates.

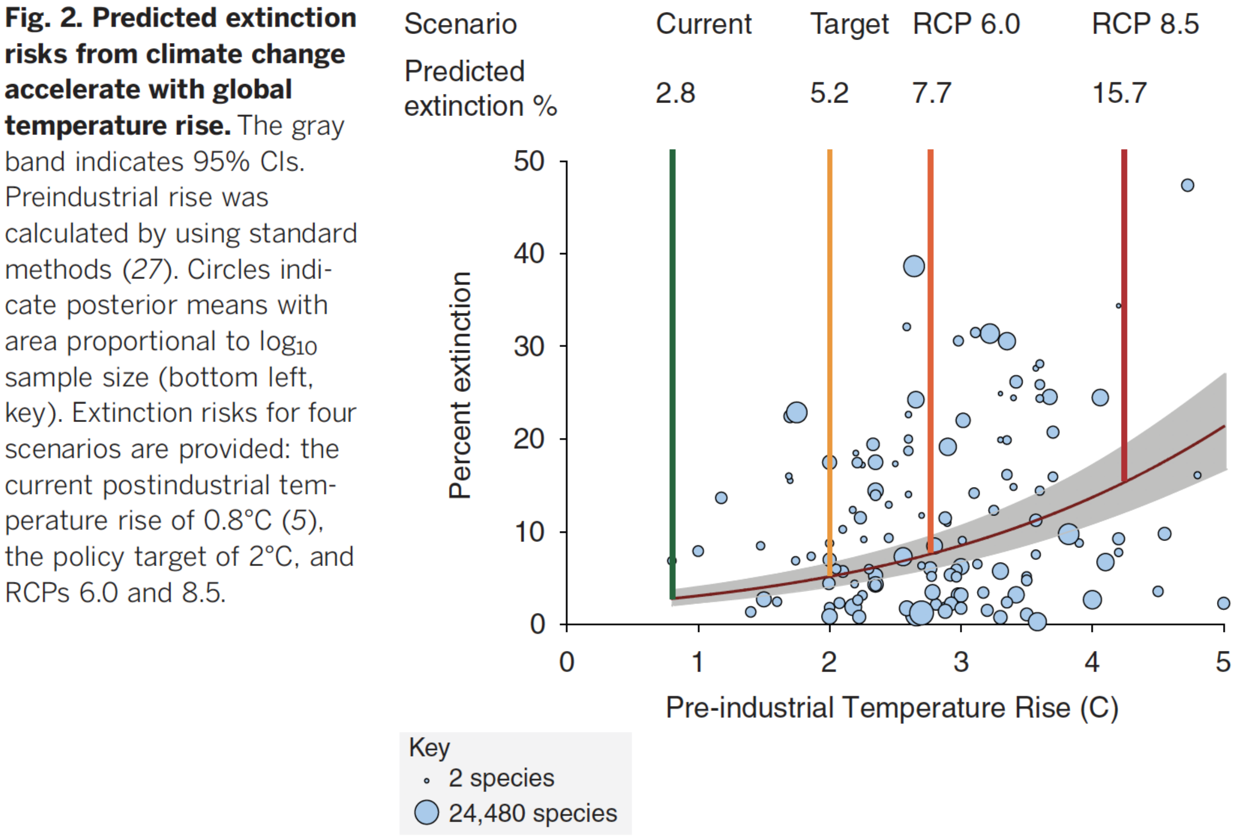


Fig. 2. Predicted extinction risks from climate change accelerate with global temperature rise. The gray band indicates 95% CIs. Preindustrial rise was calculated by using standard methods (*27*). Circles indicate posterior means with area proportional to $\log_{10}$ sample size (bottom left, key). Extinction risks for four scenarios are provided: the current postindustrial temperature rise of 0.8°C (*5*), the policy target of 2°C, and RCPs 6.0 and 8.5.

Figure 2.7: [PLACEHOLDER FOR SECOND ORDER DRAFT: to be updated with new literature. Taken from Figure 2 of Urban (2015).]

[START FAQ 2.2 HERE]

FAQ2.2: Is Climate Change driving species extinct?

[PLACEHOLDER FOR SECOND ORDER DRAFT]

[END FAQ 2.2 HERE]

2.5.4 Projected Impacts at Level of Whole Ecosytems

2.5.4.1 Global overview, assessment of models, and sources of uncertainties

There are several models used to project biome or ecosystem level changes under various climate change scenarios. Model outputs are generally in agreement in projecting broad patterns at the global scales, and vary greatly in how well model projections are in accord with recent observed changes, particularly at the local and regional scales. [[PLACEHOLDER FOR SECOND ORDER DRAFT: section to be developed. Scope: Whole-system Level Model Projections of Impacts (includes vegetation models, biome models) – discussion of uncertainty stemming from different biological models, different climate models. Comparison across models - level of model agreement (displayed in both a table and a map).]

2.5.4.2 Modelled Changes Within and Between Land-Cover Classes Globally

Climate change, change in atmospheric CO_2 levels, nitrogen (N) deposition, air pollution and land-use change will have large impacts on vegetation cover, ecosystem processes, and habitat and species composition over the next decades, with the relative importance of these drivers very likely differing between biomes/regions. Most studied are the effects of future climate change (in combination with atmospheric CO_2), and to a lesser degree land-use change (Pereira et al., 2010; Warren, 2011; Pecl et al., 2017), while impacts on pollution effects have received little attention in large-scale projections. Global vegetation and Earth system models agree on climate-change driven shifts of biome boundaries of hundreds of km over the next century, but with large discrepancies (even for similar scenarios) between models and between scenarios regarding the speed and overall degree of change (Gonzalez et al., 2010a; Pereira et al., 2010; Pecl et al., 2017). Climate (combined with CO_2) effects result in increased global forest cover by 2050 and 2100, with the magnitude of the change increasing with the forcing (Davies-Barnard et al., 2015). For already relatively small temperature increases (<2°C above pre-industrial) alterations of 2–47% of the areal extent of terrestrial ecosystems has been projected, increasing drastically with higher-warming scenarios ((Warren, 2011), and refs therein). Probabilistic methods confirm the risk of drastic changes in vegetation cover (e.g., forest to non-forest or vice versa) at the end of the 21st century already being substantial for <=2°C warming scenarios, especially in tundra, and in tropical forest and savanna regions, with more subtle changes (within a given biome types) likely to occur in all regions (Ostberg et al., 2013).

Figure 2.8: [PLACEHOLDER FOR SECOND ORDER DRAFT: figure will portray a map that shows a range of DGVMs and scenarios for the change in modelled PFT composition. Data are available from: Warlind, 2014 (LPJ-GUESS) (Wårlind et al., 2014); Kim , 2018 (LPJ, LPJ-GUESS) (Kim et al., 2018); and will be possibly also available from some of the ISIMIP runs/Biome sector.]

"Novel ecosystems" -abiotic conditions and communities made up of combinations with no current or historical equivalent- will be increasingly common in the future [PLACEHOLDER FOR SECOND ORDER DRAFT: uncertainty statement will be added] (Radeloff et al., 2015; Ordonez et al., 2016) which will require new approaches to conservation that are designed to adapt to rapid changes in species composition and ensuing conservation challenges. Climate change will impact biodiversity hotspots. For instance, for two contrasting future scenarios, at the end of the 21st century loss of present-day climate analogues have been estimated to negatively influence 25% of endemic species on average per hotspot, with largest effects in low latitudes and island locations (Bellard et al., 2014). Projected future changes in species ranges, species extinctions and community diversity may be under- or overestimated by models that do not explicitly account for species interactions such that loss of one species would trigger loss of others (Bellard et al., 2012; Schleuning et al., 2016). The use of different scenarios, different diversity metrics and different time horizons make generalisations challenging. What is more, diversity loss in one habitat/region cannot be compensated for by gains in another one, therefore diversity estimates have to be done on a regional scale.

Species diversity has a large influence on ecosystem function, and *vice versa*, while ecosystem functioning is relevant as precursor for ecosystem services (Hooper et al., 2012; Mokany et al., 2016). For instance, it has been argued that loss of global forest diversity would result in decline of productivity (Liang et al., 2016). Terrestrial ecosystems contribute notably to climate regulation by removing presently around 30% of anthropogenic CO_2 emissions (Le Quéré et al., 2018). However, the latest report of IPCC WG1 places low confidence on how carbon stocks and fluxes will evolve over the coming decades (Ciais et al., 2013). The rate and even the sign of changes in simulated trajectories of future vegetation cover and ecosystem biophysical and biogeochemical exchange fluxes with the atmosphere or waterways are highly inconsistent between ecosystem process models, even for a given scenario (Ciais et al., 2013; Piao et al., 2013; Friend et al., 2014; Nishina et al., 2014). Large uncertainties in global projections arise from interactions of the ecosystem water balance, and nitrogen (and other nutrient) availability with the carbon cycle (Ciais et al., 2013; Zaehle, 2013; Wieder et al., 2015a; Pugh et al., 2016), as well as from simulated ecosystem carbon residence times (Friend et al., 2014; Ahlstrom et al., 2015; Koven et al., 2015a). Likewise, for a given scenario, differences in projected climate from different Earth System Models can lead to increases or decreases of ecosystem functioning such as global terrestrial carbon uptake (Ahlström et al., 2012).

2.5.4.3 Conclusions Drawn from Ecosystem/Biome Level Models Of Future States

[PLACEHOLDER FOR SECOND ORDER DRAFT]

2.5.4.4 Focus on Specific Sub-Regions

2.5.4.4.1 Tropical forest

Future amount and seasonality of precipitation, increased temperatures, and prolonged droughts and droughts moderated fires are likely key factors for the distribution of tropical humid and dry forests (Bonai et al., 2016; Corlett, 2016; Anderson et al., 2018; da Silva et al., 2018; Fontes et al., 2018; O'Connell et al., 2018; Aguirre-Gutiérrez et al., 2019; Bartlett et al., 2019; Stan and Sanchez-Azofeifa, 2019). Even though at least part of the tropical forest biome is projected to shift towards warmer and drier climate, most multi-model studies assuming rapid economic growth/business-as-usual scenarios (A2, A1B, RCP8.5) show an increase in future woody biomass and areas of woody cover towards the end of the 21st century. Although substantial climate-change driven changes are projected in all studies, a forest “dieback”, as postulated in particular for the Amazon region, does not occur in the majority of simulations (Malhi et al., 2009; Poulter et al., 2010; Rammig et al., 2010; Higgins and Scheiter, 2012; Huntingford et al., 2013; Davies-Barnard et al., 2015; Wu et al., 2016). Still, uncertainties are large and some model experiments have found reduced forest area in particular in regions that are today at the dry end of tropical forest, for instance a reduction of between ca. 35-55% by 2050 was found for the Amazon for RCP2.6-8.5 (Nobre et al., 2016).

Model projections of future biodiversity in tropical forests are rare/absent. Arguably, species are most vulnerable to climate change effects in higher altitudes or at the dry end of tropical forest occurrence (Krupnick, 2013; Nobre et al., 2016; Trisurat, 2018). Tropical lowlands might lose plant species as temperatures rise above species’ heat tolerance but could also lead to novel communities of heat tolerant species (Colwell et al., 2008; Trisurat et al., 2009; Trisurat et al., 2011; Krupnick, 2013). Statistical models that correlate data on species abundance with information on human pressures (such as land-use change, population density) found for tropical and sub-tropical forests that birds, invertebrates, mammals and reptiles show a decline in their probability of presence with declining forest cover, which is particularly pronounced in forest specialists or narrow-ranged birds (Newbold et al., 2014). Bird (2012) estimated that potentially >10% of Amazon avian species to be threatened by future deforestation by a business-as-usual scenario (and nearly 8% when mitigation measures are considered). Continued harvesting for tropical timber, land cover change, and extreme fires are seen as a threat to species such as dipterocarps (Koh et al., 2011; Krupnick; Gaveau et al., 2014; Miettinen et al., 2017; Lilleskov et al., 2019) while species persisting in human-modified ecosystems might increase in abundance (Newbold et al., 2014)whereas persistence of species in fragmented forested landscapes depend on existence of corridors that allow to access food sources and breeding habitats (Gardner et al., 2009). Climate change is considered to enlarge the area of suitability of the booming tree crops (e.g. oil palm, acacia, Eucalyptus, and rubber) (Cramb et al., 2015) (Koninck et al., 2011; Nath, 2016; Hurni et al., 2017; Varkkey et al., 2018) in south-east Asia and shift environmental suitability for mammals to either higher altitudes or the existing forest remnants. As a consequence the suitable area for mammals is reduced by between a median of 47.7% (RCP 2.6) and 67.7% (RCP8.5) by 2070, with large variability depending on the different species (Brodie, 2016). The area under rubber plantations had increased from 1536 km2 (8%) of Yunnan province in China in 2002 to 4242 km2 (22%) in 2010 and potentially increases pressure on remaining biodiversity both within and outside of protected areas (Zomer et al., 2014) (see also Cross-Chapter Paper Tropical Forests).

2.5.4.4.2 Tundra ecosystems and permafrost/northern peatlands

Arctic tundra is one of the few terrestrial ecosystems where climate change impacts are already clearly visible and climate change is the dominant driver of changes in biodiversity and ecosystem (Settele et al., 2014; Uboni et al., 2016). Research on the plausible futures of Arctic tundra has focused on climate change as the sole, or dominant direct driver of change. This focus is coherent with the very large projected changes in climate combined with the relatively modest projected changes in other direct drivers such as land use change, overexploitation or pollution. Climate models project that warming for the Arctic tundra is likely to continue at about double the global rate. Wintertime warming, for example, is projected to exceed 8°C for much of Arctic tundra by the end of the century for scenarios of high greenhouse gas emissions and 2°C for scenarios of low greenhouse gas emissions (as compared to the period 1986-2005, (*in SOD: cite IPCC WGI – latest updates*)). The Arctic is also projected to have among the largest increases in precipitation globally, although there is high uncertainty in these projections. In contrast to climate change, land use change is projected to be very low in Arctic tundra systems (van Asselen and Verburg, 2013).

Models of the response of tundra species to future climate change have focused on several well-studied animal species such as polar bears, seals, fox, caribou and birds (Settele et al., 2014). Polar bears have received considerable attention because populations have been rapidly declining in several regions with climate change as one of the likely drivers (Settele et al., 2014). Recent projections of polar bear population dynamics have added considerably to the understanding of the underlying mechanisms (Atwood 2016, Lunn 2016, Dey 2017), and these projections are in agreement with previous work indicating projected negative impacts of future climate change on polar bears in most regions. Atwood (2016) found that climate change had projected negative effects on polar bear populations via its effects on sea ice (see also Lunn 2016), and that these were much more important determinants of bear population dynamics than being hunted or other interactions with people. Dey (Dey et al., 2017) found that a shift from seal to bird nest predation by polar bears as a mechanism of adaptation to climate change is projected to be insufficient to halt the decline of bear populations. Several other tundra dependant species show contrasting projected responses to climate change. Increases in shrub dominance due changes due to climate warming, are projected to have uncertain or widely varying effects on tundra plant species, ground squirrels and birds (Mod and Luoto, 2016; Thompson et al., 2016; Wauchope et al., 2017). For example, many more bird species are projected to profit from than decline due to modest increases in shrubs, while large shifts in shrub dominance are projected to have negative effects on most bird species (Thompson et al., 2016).

Models of plant functional types and vegetation response to climate project that the observed increases in shrub dominance and in boreal forest encroachment driven by recent warming (Settele et al., 2014) are to continue over the next century and accelerate under the higher greenhouse gas emissions scenarios leading to a shrinking of the area of tundra globally (Mod and Luoto, 2016; Gang et al., 2017). These changes in vegetation, when combined with warming and increased precipitation effects on soil thawing and carbon cycling, are projected to modify greenhouse gas emissions and have biophysical feedbacks to regional and global climate. Recent studies have provided important additional insights into projections since the latest IPCC report. Multi-model comparisons using dynamic vegetation models show that Arctic tundra ecosystems are generally projected to continue to sequester carbon throughout most of the 21st century. But there is much higher uncertainty than previously recognized with some models indicating a shift to very large carbon sources by the end of the century, and that the bulk of this uncertainty arises from differences between the vegetation models rather than from differences in climate models or greenhouse gas emissions scenarios (Nishina et al., 2015; Ito et al., 2016). Additional mechanisms, when taken account in models, suggest that climate change may strongly interact with other factors, such as fire, to even further increase uncertainty in projections of tundra ecosystem function (Jiang et al., 2017). A model recently developed specifically for Arctic tundra ecosystems suggests that the paradigm of increasing shrub domination due to climate change should also be nuanced, because this response appears to depend heavily on concurrent changes in precipitation as well as the dynamics of local thawing of the permafrost (van der Kolk et al., 2016).

2.5.4.4.3 Savannahs

Worldwide, woody cover has been observed to increase in savannas (Buitenwerf et al., 2012; Donohue et al., 2013), as a result of enhanced levels of CO_2 combined with changed grazing impacts. Especially in warm and dry environments, elevated CO_2 fosters plants with C_3 photosynthesis (often woody plants) in contrast to C4 species (Moncrieff et al., 2014; Midgley and Bond, 2015; Knorr et al., 2016a). Future fire spread is expected to be reduced when ecosystems become increasingly woody-dominated (Knorr et al., 2016b) which then provides a further positive feedback on shrub and tree growth.

Those dynamic vegetation models that account realistically for mechanisms of plant growth, competition and demography have been shown to reproduce savanna distribution and fire-vegetation interactions satisfactorily (Baudena et al., 2015; Moncrieff et al., 2016), suggesting that statistical distribution models will not provide a complete picture of future ecosystem functioning and diversity in tropical grassy biomes. Given that on different continents the relative importance of climate, fire and other factors in shaping savanna vegetation and distribution varies, models that incorporate specific intercontinental differences are superior compared with models that seek to model savanna distribution with a single, global parameterisation (Moncrieff et al., 2016). For future climate change and CO_2 concentrations (RCP4.5), savanna expanse declined by around 50% by 2070 in Africa and South America; changes were small in Australia (Moncrieff et al., 2016). A broad range of future climate and CO_2 changes were found to enhance vegetation C storage in Australian savannas. The effect was smallest when precipitation when rainfall seasonality was largest (and growing season length shortest) but the enhanced woody cover was not sufficient to suppress grass growth and fire (Scheiter et al., 2015) Increases in woody vegetation in what is now grass-dominated would possibly come with a carbon benefit, but is expected to decrease biodiversity.

About 50% of Brazilian Cerrado has been transformed to agriculture and pastures (Lehman and Parr, 2016), and African savannas have been proposed to follow a similar tropical agricultural revolution pathway in order to enhance agronomical prosperity (Ryan et al., 2016). Given projected large population increases in many regions of Africa for all SSPs (Knorr et al., 2016b) pressure towards land conversion together with intensification on existing grazing and croplands will increase, with large uncertainty arising from assumed socio-economic development. Assumptions about large agricultural technological and knowledge advances tend to result in crop/pasture area being stable or even declining in future despite of population growth or dietary changes. For Africa, such a scenario led to even an increase in average mean species abundance by 2050 (Alkemade et al., 2013). Managing land for climate change mitigation will enhance already existing pressures on land resources in future. In an Integrated Assessment Model analysis, implementation of a global forest conservation policy under a RCP 2.6 framework that mimics a REDD scenario led to expansion of agricultural land into what is currently grasslands and savannas by 2050 and 2100, with associated loss of carbon (and inferred loss of biodiversity (Popp et al., 2014; Searchinger et al., 2015)*)*. Upon implementing a tax on all terrestrial carbon emissions this "spill-over" was stopped and very little land-use change was found overall in 20150 and 2100 compared to the present – an effect that was possible by at the same time allowing for assumption of large technology increases regarding food production to compensate for the low crop and pasture area changes. Land-cover change projections under the RCP 2.6 and 4.5 scenarios, combined with a "random forest model" resulted in a decline in grassy biome habitat loss in parts of sub-Saharan Africa by 2070, but in different regions for the RCP2.6 and 4.5 case (Aleman et al., 2016; Aleman et al., 2017), reflecting differences in the underlying socioeconomic assumptions.

Land-use change effects in tropical grassy ecosystems must be viewed with respect to their interactions with fire. Longer fire return interval enhances woody cover in Australian savannas (under SREAS A1B climate and CO_2 (Scheiter et al., 2015)). Population growth and has also been shown to decrease annual burned area through factors such as fire suppression, controlled burning and/or fire suppression (Archibald et al., 2013). Population scenarios, including different degrees of urbanisation, were found to have equally large effects on projected burnt area than different climate and CO_2 scenarios (Knorr et al., 2016a). When human effects were included in simulations, burnt area declined in most cases, dominated by the African continent due to its large contribution to global burnt area. Still, human fire risk does not decline if people move into increasingly fire-prone regions.

[INSERT TABLE 2.5 HERE]

Table 2.5: Projected vulnerabilities and risks of ecosystems to biome shifts from spatial analyses of vegetation biogeography.

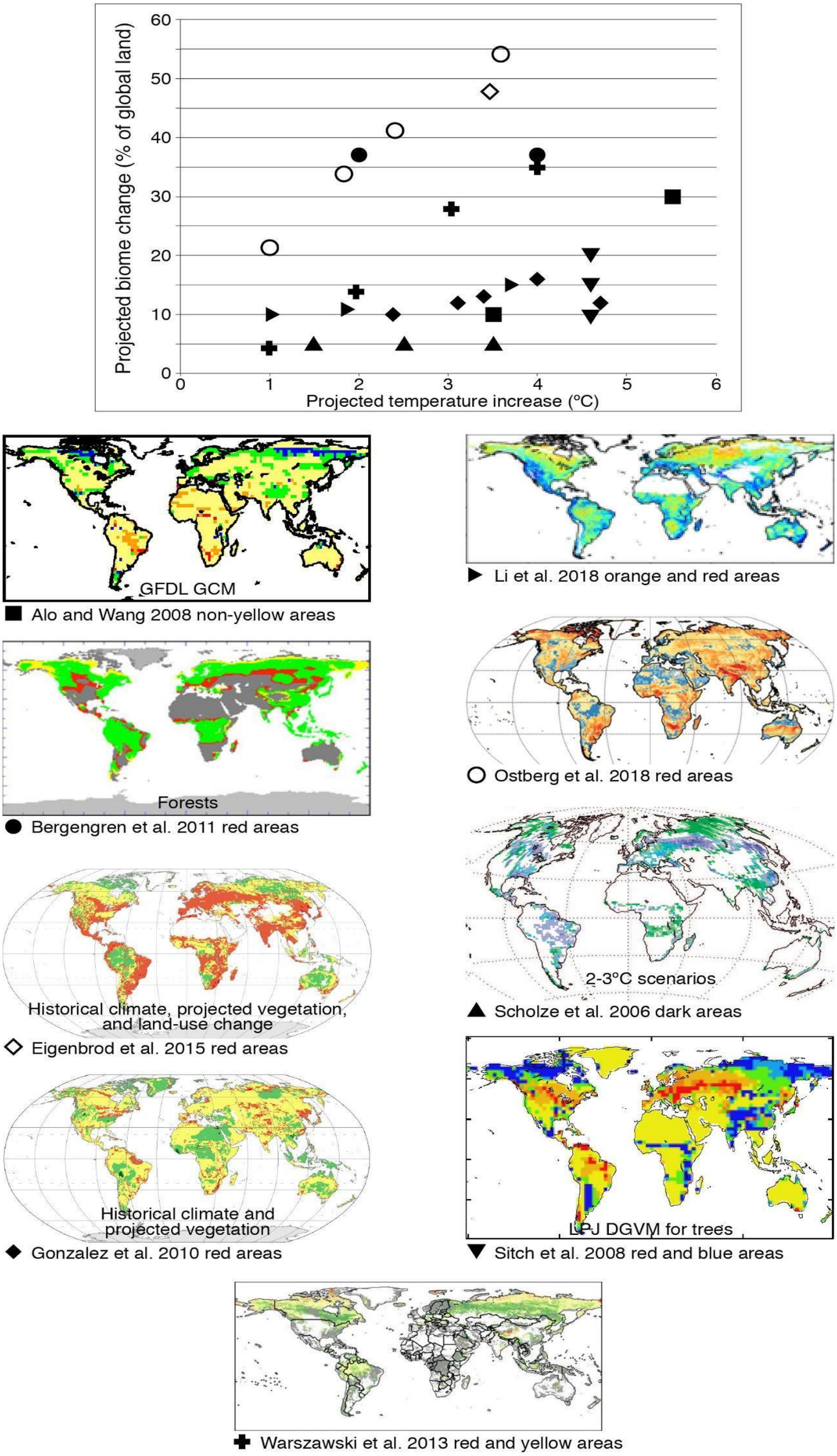


Figure 2.9: Projected vulnerabilities and risks of ecosystems to biome shifts from spatial analyses of vegetation biogeography. Open symbols denote combined risk from climate change and land-use change.

[START BOX 2.1 HERE]

Box 2.1: Assessing Past Projections of Change

To understand how climate change is going to impact us into the future we use models to project what the future impacts of climate change will be. However it is vital to continually assess the accuracy of our past projections against new evidence so as to understand where our future projections of change are accurate and where they need more work. One way to do this is to look back in the past at models that projected change and compare the projections with the observed changes that have occurred. This offers us a very powerful approach to assess the models. We can tell which regions were predicted well and where predictions did not line up with impacts. This allows us to understand what type of models work well and what changes we need to make to improve future models.

We use this approach in this assessment to assess a model from the Fourth Assessment report (AR4). In AR4 the authors took a Dynamic Global Vegetation Modelling (DGVM) approach to project changes in potential vegetation in 2100 under emissions scenarios B1 and A2. The model presented future changes as shifts in key functional groups e.g., trees, shrubs and grasses. Here we compare vegetation changes up through ~2017 due to climate and non-climate factors with the AR4 projections for 2100. To do so we collated data from existing review papers and papers published in the last 15 years. From these papers we recorded the geographical location of a functional type change and noted the causes, whether climate or non-climate. We plotted each of these changes onto the geographical output form the model in AR4 (Figure Box 2.1.1).

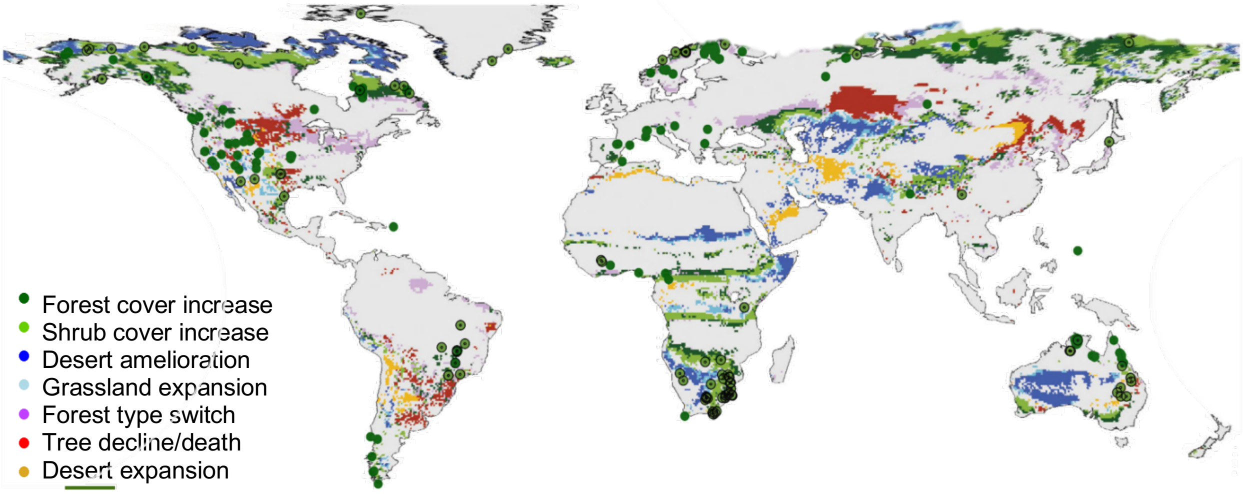


Figure Box 2.1.1: Projections of potential future biome shifts in 2100 under emissions scenario B1 from AR4 are presented as broad bands of colour with the colour representing potential changes in biome distributions. Each biome/functional type is presented as a different colour. The observed changes through ~2017 are presented as coloured dots with the colour of each dot matching the functional type change that has occurred. The observations are due to a mix of factors, climate and non-climate, whereas the AR4 model only included climate factors.

Table Box 2.1.1: Comparison of projections on biome change from AR4 with observed trends since AR4.

Key Projections for AR4 Model	Observed Recent Trends
Boreal forest and Arctic tundra ecosystems would show increased growth due to longer and warmer growing seasons	Observations confirm projection and tree cover and shrub cover densification has occurred widely across these regions
Woody boreal vegetation expected to spread into tundra at higher latitudes and higher elevations	Widespread observations confirming this projection
Contraction of boreal forest at southern ecotone with continental grasslands	
Increase in tree mortality in temperate forests	

Uncertain for the Sahel and other semi-arid regions of increasing drought and aridification vs models projecting more greening due to CO_2 driven increases in water-use efficiency	Extensive greening and encroachment across semi-arid regions of the world. Increase in tree and grass growth across most of the Sahel
In savannas, woody encroachment is projected to be a consequence of enhanced water-use efficiency and increased precipitation in some regions.	Extensive woody encroachment across savannas of the world
The moderate drying, including desert amelioration, was projected in southern Africa, the Sahel region, central Australia, the Arabian Peninsula and parts of central Asia due to a positive impact of rising atmospheric CO_2	Observed – grassland expansion into desert and increase in woody cover in Southern Africa. Increase in greening in Sahel and central Asia.
A general increase of deciduous at the expense of evergreen vegetation is predicted at all latitudes	
Increased tree mortality in forests in both the eastern USA and eastern Asia due to drought stress	

[END BOX 2.1 HERE]

2.5.5 Risk Assessment of Ecosystems and Related Services

2.5.5.1 Biome shifts

Previous spatial analyses of potential future vegetation under climate change have indicated that one-fifth to one-quarter of global terrestrial area is highly vulnerable to biome shifts by 2100 (Scholze et al., 2006; Alo and Wang; Sitch et al., 2008; Gonzalez et al., 2010a). Since the Fifth Assessment Report, at least three additional publications projected potential future global vegetation, estimating that up to one-third of global terrestrial area is highly vulnerable to biome shifts by 2100 due to climate change under high emissions (SRES A2, RCP8.5) (Warszawski et al., 2013; Park et al., 2015; Li et al., 2018a) (Figure 2-N, Table 2-N). At least seven publications since the Fifth Assessment Report have projected potential biome shifts for particular regions (Chakraborty et al., 2013; Langdon and Lawler; Moncrieff et al., 2015; Boit et al., 2016; Lyra et al., 2016; Rasquinha and Sankaran, 2016; Rowland et al., 2016), projecting biome changes on one-tenth to four-fifths of regional areas due to climate change under high emissions (SRES A2, RCP8.5). Global and regional biome projections consistently indicate potential biome shifts upslope or towards the Poles or the Equator. Generally, projections indicate high risks of biome shifts on ~5-10% of global terrestrial area under RCP2.6, increasing as a function of temperature to high risks on ~10-35% of global terrestrial area under RCP8.5 (Figure 2-N, Table 2-N). Models project high risks of poleward shifts of boreal conifer forest into tundra across the Arctic, upslope shifts of forest biomes into alpine biome areas of mountains, and dieback and shifts of Amazon tropical rainforest to shrubland or grassland (Figure 2-N). The globally high biodiversity in tropical forests (Kreft and Jetz, 2007) and on mountains (Rahbek et al., 2019) suggest that biome shifts may disproportionately affect biodiversity. Relatively slow rates of soil development at high-elevations, however, may constrain the pace of upslope biome shifts on mountains (Rahbek et al., 2019). The current sharp transitions at both the northern and southern limits of boreal conifer biome, to temperate woodland in the south and tundra in the north, suggest an abrupt tipping point for potential boreal biome shifts under climate change (Scheffer et al., 2012).

Previous research has identified massive forest dieback and a biome shift to grassland as a serious risk to the integrity of Amazon rainforest and its humid climate (Oyama and Nobre, 2003; Lenton et al., 2008; Nepstad et al., 2008; Salazar and Nobre, 2010; Settele et al., 2014). Increased temperature, increased fire from climate change and local human action, combined with extreme aridity in drought years would drive the Amazon forest dieback. Abrupt tipping points could occur at 4ºC for the biome shift (Salazar and Nobre, 2010) and 40% deforestation for the humid climate (Sampaio et al., 2007). Research since the IPCC Fifth Assessment Report has continued to indicate high risks of Amazon forest dieback from continued climate change and deforestation, acting through wildfire and drought (Anadon et al., 2014; Lyra et al., 2016; Nobre et al., 2016; Boulton et al., 2017; Zemp et al., 2017; Marengo et al., 2018). Under RCP8.5, half of Amazon tropical evergreen forest could shift to grassland, but lower emissions (RCP4.5) could limit the loss to ~5% (Lyra et al., 2016). Precipitation declines from reduced evapotranspiration inputs after forest loss and other vegetation-atmosphere feedbacks could cause additional Amazon forest loss of one-quarter to one-third

(Zemp et al., 2017). More dieback could occur after 2100 than before (Boulton et al., 2017). Experimental burns in Amazon rainforest found abrupt doubling and quadrupling of tree mortality in drought years in plots burned annually and every three years, compared to background mortality (Brando et al., 2014). Flammable grasses invaded burned areas and forest edges. This experiment provides a field example of what vegetation models project.

Due to the coarse resolution of global data, most global models omit island areas. A review of temperature sensitivities of tropical montane cloud forests on 93 islands around the world indicates an elevated risk that climate change could lift the condensation level and the lower range limit on island cloud forests but would affect a smaller area than deforestation and plant species invasions (Pouteau et al., 2018).

Biome shift projections come from equilibrium models, which analyze risk through static representations based on thresholds of vegetation to climate and edaphic factors, or dynamic global vegetation models, which dynamically calculate interactions of biogeography, nutrient cycling, and fire (Sitch et al., 2008). The main limitations of both of these types of models include coarse spatial scale made necessary by the computing power needed to run complex models, reference vegetation data often affected by human land use change and other non-climate factors, and difficulties in accurately modelling dispersal or other ecosystem processes that do not have comprehensive global databases. Many authors publish only maps of their results and do not report the fraction of surface area at different levels of vulnerability or risk.

Research since the Fifth Assessment Report has explored new approaches to analyzing the combined risk of biome shifts due to climate change and habitat fragmentation due to human land use change (Eigenbrod et al., 2015; Boit et al., 2016; Ostberg et al., 2018). These analyses indicate that agriculture, urbanization, and roads generate barriers to dispersal that can hinder shifts of natural vegetation. Consequently, the fraction of area at high risk to biome shifts from climate change and land use change combined can increase to double or triple the risks from climate change alone (Figure 2-N). Analysis of the combined risk of biome shifts and low national economic activity indicates that less materially wealthy countries in southern Africa may experience biome shifts earlier than materially wealthier countries in other parts of the world (Park et al., 2015). Biome shifts and wildfire under emissions scenario A2 generate a high risk of establishment of invasive species on one-sixth of global terrestrial land (Early et al., 2016).

2.5.5.2 Wildfire

Continued climate change under high emissions (SRES A2) could increase future wildfire frequency on one-third to two-thirds of global land by 2100 and decrease fire frequency on one-fifth of global land, with a net global fire frequency increase of ~30% per century (Gonzalez et al., 2010a; Moritz et al., 2012). Since the IPCC Fifth Assessment Report (Settele et al., 2014), additional modelling of indicators of wildfire activity other than fire frequency projects increases at the global level (Flannigan et al., 2013; Knorr et al., 2016a; Burton et al., 2018; Abatzoglou et al., 2019). Under RCP8.5, increases in extreme fire weather days (Canadian Forest Fire Weather Index) due to anthropogenic climate change emerge from natural variability for one-third to two-thirds of burnable global land by 2050 (Abatzoglou et al., 2019). Under emissions scenario A2, the potential for severe fire (Cumulative Severity Rating) could increase across approximately four-fifths of global land and not decrease anywhere (Flannigan et al., 2013). Under RCP2.6, the potential for severe fire (McArthur Forest Fire Danger Index) could increase for approximately half of global land but decrease on approximately one-tenth of global land (Burton et al., 2018). Under RCP8.5 and shared socio-economic pathway (SSP3 - high population growth, slow urbanization), the number of people living in fire-prone areas could increase by three-quarters, to 720 million people in 2100, in a global population of 12.4 billion people (Knorr et al., 2016a). Lower greenhouse gas emissions under RCP4.5 could reduce the number of people at risk by 70 million people. Those projections indicate, however, that human exposure to wildfires results from human population growth more than increase in burned area (Knorr et al., 2016b).

Geographic areas of high fire risk differ among the various global analyses. Some regions identified at high risk by multiple analyses include: Amazon (Gonzalez et al., 2010a; Knorr et al., 2016a; Burton et al., 2018; Abatzoglou et al., 2019), Mediterranean Europe (Gonzalez et al., 2010a; Burton et al., 2018; Abatzoglou et al., 2019), southern Africa (Gonzalez et al., 2010a; Burton et al., 2018; Abatzoglou et al., 2019), Tibet (Moritz et al., 2012; Flannigan et al., 2013; Abatzoglou et al., 2019), tundra (Moritz et al., 2012; Flannigan et al., 2013), western Australia (Gonzalez et al., 2010a; Burton et al., 2018; Abatzoglou et al., 2019), western US (Gonzalez et al., 2010a; Moritz et al., 2012; Knorr et al., 2016a).

In some regions of high risk, higher-resolution spatial analyses have projected wildfire patterns under climate change. In the Brazilian Amazon, climate change under RCP8.5 and high deforestation could double the area with high fire relative probability (Fonseca et al., 2019), increase burned area 400% to 2800% by 2100 (Le Page et al., 2017), and increase fire intensity 90% (De Faria et al., 2017). Lower greenhouse gas emissions (RCP4.5) and reduced deforestation could substantially reduce fire risk to a one-fifth increase in the area with high fire relative probability (Fonseca et al., 2019) and a 100% to 500% increase in burned area by 2100 (Le Page et al., 2017). Fire is the mechanism for potential biome shifts of half of Amazon rainforest to grassland (Lyra et al., 2016; Zemp et al., 2017). Experimental burns in Amazon rainforest found abrupt tree mortality in drought years in frequently burned plots and extensive invasion of burned areas and forest edges by flammable grasses (Brando et al., 2014). In the Bolivian Amazon, burning experiments found a substantial shift of species composition from fire-intolerant to fire-tolerant tree species (Devisscher et al., 2016).

In the Brazilian Cerrado, climate change under RCP8.5 could double burned area by 2100 while emissions reductions to meet the Paris Agreement goal (RCP2.6) could cause a one-tenth reduction of burned area from 2050 to 2100 (Silva et al., 2019). In Mediterranean Europe, climate change of 3ºC could double or triple burned area but keeping the temperature increase to 1.5ºC could limit the burned area increase to 40-50% (Turco et al., 2018). The most severe increases could occur in Spain and Portugal. Severe fire followed by drought could cause Mediterranean biome shifts of forest to non-forest (Batllori et al., 2019). In Australia, climate change under RCP8.5 increases risks of pyroconvective fire by 20 to 40 days in rangelands of Western Australia, South Australia, and the Northern Territory (Dowdy et al., 2019). Pyroconvective fire conditions could reach into the more populated areas of New South Wales, Australia, particularly in November and December, the start of austral summer (Di Virgilio et al., 2019). Increases in heat and potential increases in wildfire threaten the existence of temperature montane rainforest on Tasmania, Australia (Mariani et al., 2019). In India, increases in wildfire could substantially reduce the range of three tree species endemic to the Himalayas (Chitale and Behera, 2019). In boreal ecosystems, climate change under RCP8.5 could double the number of fires in Finland (Lehtonen et al., 2016), increase lightning-driven burned area by one-third in the Northwest Territories, Canada, and by half in Alaska, USA, by 2100 (Veraverbeke et al., 2017), push half of the area of tundra and boreal forest in Alaska, above the burning threshold temperature, and double burned area in Alaska (Young et al., 2017a). Increased fire in Alaska could reduce the extent of lichen tundra by up to one-quarter under RCP 8.5 (Pastick et al., 2017), reduce lichen tundra habitat for caribou up to one-fifth under emissions scenario A1B (Gustine et al., 2014), and increase deciduous forest area 200% to 600% (Pastick et al., 2017).

For the U.S., climate change under RCP8.5 could increase the number of fires with an area >50 km2by 300-400% by 2070 in the lower 48 U.S. states (Barbero et al., 2015). For the western U.S., burned area in climate-limited ecosystems, mainly forests, could double by 2080 under emissions scenario A1B (Littell et al., 2018), climate change under RCP8.5 makes one-quarter of forest area highly vulnerable to burning (Buotte et al., 2019). In Rocky Mountain forests in the U.S., climate change under RCP8.5 increases the risk of fire-facilitated conversion of ~7% of forest to non-forest by 2050 (Parks et al., 2019). In California, climate change under emissions scenario A2 could double fire frequency in some areas (Mann et al., 2016), but emissions reductions (emissions scenario B1) could keep fire frequency from increasing (Westerling et al., 2011). In the Sierra Nevada, California, climate change under RCP8.5 could double fire frequency (Parks et al., 2018; Syphard et al., 2018), although topography and winds could foster cold air pools that would serve as refugia from fire (Wilkin et al., 2016). Higher fire frequencies in the Mediterranean ecosystems of southern California could cause a biome shift of chaparral (evergreen temperate woodland) to coastal scrub (deciduous temperate shrubland) (Lippitt et al., 2013). Carbon dioxide fertilization and warmer conditions could increase invasive grasses and wildfire in desert ecosystems of the south-western United States where wildfire has historically been absent or infrequent, increasing mortality of any sparse tree cover (Horn and St. Clair, 2017; Klinger and Brooks, 2017; Syphard et al., 2017; Moloney et al., 2019; Sweet et al., 2019).

[START FAQ 2.3 HERE]

FAQ2.3: Is climate change increasing wildfire?

[END FAQ 2.3 HERE]

2.5.5.3 Tree Mortality

Globally, the risk of drought-induced mortality is similar in magnitude for angiosperms and gymnosperms (Anderegg et al., 2016). Boreal and temperate forests possess greater diversity of physiological traits related to plant hydraulics, so they are more buffered against drought than tropical forests (Anderegg et al., 2018).

This lack of buffering capacity for plant moisture during drought increases the risk of tree mortality in Amazon rainforests and the possibility of a tipping point of massive mortality and conversion to grassland (Oyama and Nobre, 2003; Lenton et al., 2008; Nepstad et al., 2008; Salazar and Nobre, 2010) (described in more detail above in section 2.4.3.3.1, biome shifts). Under RCP8.5, half of Amazon tropical evergreen forest could shift to grassland through drought-induced tree mortality and wildfire, but lower emissions (RCP4.5) could limit the loss to ~5% (Lyra et al., 2016). Precipitation declines from reduced evapotranspiration inputs after forest loss and other vegetation-atmosphere feedbacks could cause additional Amazon forest loss of one-quarter to one-third (Zemp et al., 2017). Similarly, in Guinean tropical deciduous forest in Africa, climate change under RCP8.5 could increase mortality 700% by 2100 or 400% under lower emissions (RCP4.5) (Claeys et al., 2019).

For temperate conifer forests, drought-induced tree mortality from climate change under RCP8.5 could cause the loss of half of Northern Hemisphere conifer forest area by 2100 (McDowell et al., 2016). For the western U.S., one-tenth of forest area, including conifer and broadleaf forests, is highly vulnerable to drought-induced mortality under RCP8.5 by 2050 (Buotte et al., 2019). In California, increased evapotranspiration in Sierra Nevada conifer forests increases the potential fraction of the area at risk of tree mortality 15-20% per degree Celsius (Goulden and Bales, 2019). In Alaska, U.S., fire-induced tree mortality from climate change under RCP8.5 could reduce the extent of spruce forest (*Picea sp.*) 8-44% by 2100 (Pastick et al., 2017). In Alaska, U.S., and British Columbia, Canada, yellow-cedar (*Callitropsis nootkatensis*) could lose half of its suitable climate (Buma et al., 2017). In Sonoran Desert ecosystems in the U.S., climate change could cause substantial mortality even of drought-adapted tree and shrub species (Munson et al., 2012; Munson et al., 2016).

In Tasmania, Australia, projected increases in wildfire (Fox-Hughes et al., 2014) increase the risks of mortality in mesic vegetation (Harris et al., 2018b) and threaten the disappearance of the long-lived endemic pencil pine (*Athrotaxis cupressoides*) (Holz et al., 2015; Worth et al., 2016) and temperate montane rainforest (Mariani et al., 2019). The area began moving 4000 years ago out of a climate suitable for temperate rainforest (Mariani et al., 2019) and the most severe fires in 250 years burned in 1960-1961 (Holz et al.) and 2016 (Worth et al., 2016). Fire intolerant tree and shrub species, particularly in Australia, are at risk of an interval squeeze resulting from imposition of increased fire frequency on reduced reproductive success from more frequent fire (Enright et al., 2015; Henzler et al., 2018).

2.5.5.4 Terrestrial Ecosystem Carbon Stocks

Continued climate change increases risks of crossing tipping points of conversion of Amazon rainforest to grassland and melting of Arctic permafrost (Settele et al., 2014) that could release substantial amounts of ecosystem carbon to the atmosphere. IPCC AR6, Working Group 1, Chapter 5 examines the projections of these and other changes in global biogeochemistry. IPCC AR6, Working Group 3, Chapter 7 examines potential carbon sequestration measures in agriculture, forestry, and other land use. This section provides information on ecological aspects of projected changes in carbon stocks and flows.

A temperature increase of 2ºC could melt 8 ± 2 million km^2 of permafrost globally, 40% of the total extent (Chadburn et al., 2017). Since the IPCC Fifth Assessment Report, more projections indicate that continued melting of Arctic tundra permafrost could release substantial carbon emissions (Koven et al., 2015b; Schuur et al., 2015; Comyn-Platt et al., 2018; Gasser et al., 2018). Under RCP8.5, models project potential permafrost carbon losses by 2100 of 28-113 Gt (Koven et al., 2015b), 11-143 Gt (Gasser et al., 2018), 42-

141 Gt (von Deimling et al., 2015), 37–170 Gt (Schuur et al., 2015), or 35-205 Gt (Schaefer et al., 2014), potentially increasing global average temperatures 0.29 ± 0.21°C (Schaefer et al., 2014). Limiting the global temperature increase to 2°C could reduce projected permafrost losses by 2100 to 20-58 Gt (von Deimling et al., 2015), 46–51 Gt (Comyn-Platt et al., 2018), 27-100 Gt (Schaefer et al., 2014). Globally, most soil carbon emissions would come from Arctic tundra, with climate change under RCP8.5 causing a soil carbon loss of 55 ± 50 Gt by 2050, increasing atmospheric CO$_2$ by 25 ppm (Crowther et al., 2016). Ultimate carbon losses from permafrost could total 160-310 Gt by 2300 under RCP8.5 or 40-100 Gt under RCP2.6 (von Deimling et al., 2015). The potentially abrupt nature of this and its fundamental impact on global biogeochemistry mark the melting of permafrost as a tipping point (Schaefer et al., 2014).

Despite the projected permafrost losses, models project that increased shrub growth from increased temperatures that could maintain a net sink in the North American Arctic through 2100 (McGuire et al., 2018; Mekonnen et al., 2018).

The IPCC Fifth Assessment Report estimated that, in the Amazon, continued deforestation and tree mortality from drought and wildfire could reduce forest area by half and release 20 ± 10 Gt carbon (Settele et al., 2014), a substantial portion of the current carbon stock of 45-60 Gt (Baccini et al., 2012; Mitchard et al., 2014; Englund et al., 2017). This could cross a tipping point of self-reinforcing feedbacks between reduced vegetation and reduced precipitation (described in more detail in section 2.4.3.2 and 2.5.5.1), leading to an additional forest loss of one-quarter to one-third (Zemp et al., 2017). No new estimate has been published of the ecosystem carbon loss from a climate change-induced dieback of Amazon tropical evergreen forest. Under RCP8.5, carbon emissions from Amazon wildfire could double by 2100 (De Faria et al., 2017). In the Pastaza-Marañon peatland in the Peruvian Amazon, climate change under RCP8.5 could shift the ecosystem from a carbon sink to a source of 0.4 ± 0.1 Gt yr^{-1} by 2100, but emissions reductions under RCP2.6 could keep the system a sink of 70 ± 30 Mt yr^{-1} (Wang et al., 2018). In Indonesia, future clearing of peatland for agriculture could emit 1.2-3.1 Gt C by 2130 (Wijedasa et al., 2018).

Modelling indicates that the current climate allows a potential carbon stock in global vegetation of 920 Gt, double the current stock (Erb et al., 2018a). This difference indicates a potential opportunity to increase ecosystem carbon removals from the atmosphere through afforestation, reforestation, and other land management actions, which could sequester 6.5 Gt y^{-1} of carbon (5.5-10 Gt y^{-1}) (Griscom et al., 2017). Theoretically, enough land exists for restoring 9 9 million km^2 of forest containing 200 Gt carbon (Bastin et al., 2019). Active reforestation, depending on species and methods, could cause serious environmental and social impacts (Heck et al., 2018; Krause et al., 2018) (see Cross-Chapter Box MITIG in this Chapter).

Reduced deforestation and degradation could sequester 1.5 Gt y^{-1}, totalling 130 Gt by 2100 (Le Quéré et al., 2018). Improved fire management, including prescribed burning in North America to facilitate growth of large native trees (Hurteau, 2017) and prescribed burning in Australia earlier in the dry season (Lipsett-Moore et al., 2018) could substantially reduce the 0.6 Gt y^{-1} of net carbon emissions from wildfires.

For protected areas, climate change under a scenario of no emissions reductions could reduce the rate of carbon storage 40%, but removal of protected status from one-third of the area to meet expanded land needs for agriculture and livestock could nearly convert the system of protected areas from a sink to a source (Melillo et al., 2016).

[START FAQ 2.4 HERE]

FAQ2.4: Will planting trees stop climate change?

[END FAQ 2.4 HERE]

2.5.5.5 Primary Productivity

Analyses of atmospheric inversion model output and spatial climate data indicate a sensitivity of net ecosystem productivity to CO$_2$ fertilization of 3.1 ± 0.1 to 8.1 ± 0.3 Gt per 100 ppm CO$_2$ (~1 °C increase) and a sensitivity to temperature of -0.5 ± 0.2, to -1.1 ± 0.1 Gt per degree Celsius (Fernandez-Martinez et al., 2019). These indicate that CO_2 enhances the terrestrial sink but temperature increases constrain it. Climate change under RCP8.5 could reduce by 90% global vegetated area where growth is limited by temperature, mainly at high latitudes (Keenan and Riley, 2018). Under RCP8.5, land-use change could reduce projected carbon increases ~25% (Quesada et al., 2018). Furthermore, emerging nitrogen and phosphorus constraints on growth may slow down CO_2 fertilization to the extent that drought and aridity would limit NPP (Penuelas et al., 2017; Zhou et al., 2019).

An additional risk that CO_2 fertilization poses to ecosystems is an increase in invasive alien plant species, which, under experimental conditions, perform better under increased CO_2 and higher temperatures than do native plant species (Liu et al., 2017).

2.5.5.6 *Climate Regulation*

In addition to comprising a principal component of the global carbon cycle, ecosystem serve to regulate climate by providing moisture inputs for precipitation across long distances from vegetation transpiration and by moderating local temperatures in tropical and temperate areas through the cooling effect of transpiration. Due to vegetation-atmosphere feedbacks, reductions of vegetation cover in one region can reduce evapotranspiration inputs to the atmosphere, leading to drier conditions elsewhere, often in drylands (Avissar and Werth, 2005; Devaraju et al., 2015; Wang et al., 2015; Swann et al., 2018). The long-term decline of rainfall in the African Sahel was initiated by warmer sea surface temperatures due to anthropogenic climate change (Giannini et al., 2003; Shanahan et al., 2009; Suarez-Moreno et al., 2018; Villamayor et al., 2018), while reduction of vegetation cover in Guinean forests, Sudanian woodland, and Sahel savanna amplify the decline via reduced evapotranspiration (Zeng et al., 1999; Yu et al., 2017a; Liu et al., 2019a) and increased albedo (Charney et al.; Yosef et al., 2018). Across Africa, actions that increase forest cover could store carbon and moderate projected future temperature increases (Wu et al., 2016; Diba et al., 2018). In another case, tree death in the Sierra Nevada, California, USA, attributed to anthropogenic climate change (van Mantgem et al., 2009), can reduce water to the atmosphere, reducing vegetation growth on the other side of North America (Swann et al., 2018). Research has demonstrated the local cooling effect of forests through transpiration (Ellison et al., 2017) across the northern mid-latitudes (Lejeune et al., 2018), in Brazil (Cohn et al., 2019), and the Czech Republic (Hesslerova et al., 2018). In Brazil, the local heating effect of deforestation extends outward 50 km Brazil (Cohn et al., 2019).

Projections of the global change in evapotranspiration under climate change, however, do not agree (Doll et al., 2016). Increasing atmospheric CO_2 produces counteracting effects, namely, reduced transpiration due to increased water use efficiency or increased transpiration due to increased plant growth (Doll et al., 2016).

2.5.5.7 *Freshwater Supply from Ecosystems*

The provision of freshwater as habitat for aquatic plants and animals and as drinking water for people depends on climate and vegetation cover (Ellison et al., 2017). Forest ecosystems and freshwater wetland ecosystems maintain water underground in the saturated zone for runoff into streams and rivers and recharge of groundwater aquifers over time. This delayed water release provides the baseline flow in dry seasons. Globally, 4 billion people depend on forested watersheds for drinking water (Mekonnen and Hoekstra, 2016). IPCC AR6 Chapter 4 assesses the physical science aspects of water supply, including precipitation, runoff, and hydrology, and social aspects of human water use. This section assesses ecological aspects of risks to freshwater supply, particularly the integrity of watersheds under continued climate change.

Reduction of vegetation cover can either decrease or increase the water yield of a watershed, depending on local geology and water storage (Evaristo and McDonnell, 2019) and increase sediment flows from erosion, degrading drinking water quality. Increased wildfire under continued climate change could increase these risks to water supplies. Under current fire conditions, almost one-half of global area is at moderate to high risk of water scarcity due to earlier runoff and increased sediment after wildfires, including India, northeast Brazil, Ethiopia, the African Sahel, Spain, Turkey, Russia, and Canada (Robinne et al., 2018). In the arid south-western U.S., projected increases of wildfire due to climate change under RCP8.5 could convert forest to non-forest cover at one site to the extent that the vegetation change alone could reduce runoff up to 10% by 2100 (O'Donnell et al., 2018). Across the western U.S., post-fire erosion under emissions scenario A2 could double sedimentation in one-third of watersheds by 2050, degrading drinking water quality (Sankey et al., 2017). In Oaxaca, Mexico, the combination of climate change under RCP8.5 and conversion of forest to urban land could decrease groundwater stocks more than projected recharge could maintain them (Olivares et al., 2019). In a watershed in southern Thailand, half of which is protected by a national park, the combination of conversion of forest to rubber plantations outside the park and a one-third increase in rainfall could increase erosion and sediment load 15%, degrading water quality for drinking and irrigation (Trisurat et al., 2016). In a watershed in northern Thailand, even though precipitation did not increase from 1975 to 2015, low-flow discharge increased, possibly reflecting 40% deforestation (Tebakari et al., 2018).

2.5.5.8 Risk to Ecosystem Services from Freshwater Systems

Climate-change has direct and indirect impacts on freshwater ecosystem services (or nature's contribution to people (Woodward et al., 2010; IPBES, 2019). Projected increases in water temperature and prolongation of thermal stratification periods will promote algal blooms dominated by cyanobacteria thereby reducing biodiversity in temperate productive lakes (Domis, 2013) (Adrian and Hessen, 2016). Given that eutrophication of Central European lakes have wiped out a significant proportion of the endemic fish fauna (Vonlanthen et al., 2012) climate induced further eutrophication may put additional threat on the fish fauna.

Direct effects of warming have been shown to affect terrestrial community trends, while the effects in aquatic communities (freshwater, marine) were more variable (Bowler et al., 2017). Such variable effects in freshwaters can be attributed to the paucity of data regarding temperature measurements across the range-wide distribution of species (as opposed to e.g. remotely-sensed land surface temperatures for terrestrial ecosystems, often requiring estimates based on literature (Comte and Olden, 2017).

Warming temperatures and altered precipitation patterns across the globe, depending on the region, result in an increased frequency of floods and extended drought periods (Milly et al., 2005). Increased evapotranspiration rates have shown to increase the risk of reduced (ground) water levels, stream drying and stream intermittencies, having direct effects on ES such as water provisioning (Woodward et al., 2010), navigation, and food security through cascading effects in food webs and community turn-over due to exceeding organisms physiological tolerances and range shifts of organisms (Daufresne et al., 2004). Such ecological impacts on ecosystem services, and especially those of changing biodiversity have shown to have indirect impacts on ES through altering ecosystem processes (Chapin et al., 2000). For instance, changes in species population dynamics, species interactions, and the introduction and persistence of non-indigenous species have shown to change freshwater ecosystem characteristics e.g. by replacing or introducing new key or keystone species (Chapin et al., 2000), e.g., mussels in the genus *Dreissena*). Such altered ecosystem processes not only impact biodiversity *per se* but can have impacts on ecosystem services that are contingent on biodiversity (while effects are dependent on the regional context (IPBES, 2018a; IPBES, 2018b; IPBES, 2018c; IPBES, 2018d).

While biodiversity and several ecosystem services can be considered synergistic (food webs, tourism, aesthetical / spiritual value (Langhans et al., 2019), others can be considered antagonistic in case of a strong ecosystem service demand (such as water abstraction, water use, food security in terms of over-exploitation). Here the balance between biodiversity and ecosystem services is key; (spatial prioritization (Langhans et al., 2019).

2.5.5.9 Risk to Ecosystem Services from Changes in Carbon Fluxes via the Terrestrial/Freshwater Interface

Changes in climate and land use modulate the close interactions between terrestrial and freshwater carbon transport and their role as carbon sink or source to the atmosphere. Lakes and rivers bury more carbon than the world's oceans combined. According to metabolic theory of ecology, respiration increases more than photosynthesis with rising temperature. As a result CO_2 emissions from terrestrial soils and freshwaters are likely to increase with global warming. While estimations of global carbon fluxes have been refined in the past decade, especially with respect to the so far underestimated role of freshwaters as substantial sinks and sources of carbon at a global scale, uncertainty in the quantity of carbon fluxes between terrestrial and freshwater ecosystems and subsequent emissions remain very high (Tranvik et al., 2009; Drake et al., 2018; Seekell et al., 2018; Sanches et al., 2019). The mechanism by which moisture controls gaseous C fluxes is a trade–off between stimulation of microbial respiration and limitation of gas diffusivity (Gomez-Gener et al., 2016). In lakes a shift toward heterotrophy would imply a reduction in the carbon sequestration capacity – and thereby a reduction of lakes acting as carbon sinks. Given the exponential response of metabolic rate to temperature (Jackson et al., 2017) found that warming had a greater direct effect on lake metabolism in tropical as compared to lakes in temperate / mid elevation and lakes in artic / high elevation areas. How much warming a lake can tolerate before it switches from net autotrophy to net heterotrophy depends on trophic state; lower under mesotrophic than under eutrophic conditions (Scharfenberger et al., 2019).

The exposure of lake and river sediments during droughts reactivates decomposition of buried organic carbon (Tesi et al., 2016). For dry river beds mineralization of buried organic matter as anoxic sediments are oxygenated during drydown along with pulses of microbial activity following rewetting of desiccated sediment are likely to increase with CC (Bernhardt et al., 2018). Conservative estimates indicate that adding emissions from exposed sediments of dry inland waters to current global estimates of CO_2 emissions from inland waters could result in a 10% increase of total inland water CO_2 emission rates covering streams and rivers (334 mmol m_{-2} day$_{-1}$), lakes and reservoirs (320 mmol m^{-2} day^{-1}) and small ponds (148 mmol m^{-2} day^{-1}). CH_4 fluxes from exposed sediments of dry inland waters are low (Marcé et al., 2019).

Methane (CH_4)emissions in freshwaters mainly originate from lakes particularly from ebullition in shallow lakes (bubble flux from sediments) (Sanches et al., 2019). In 297 lakes distributed globally, CH_4 emissions were lowest in lakes of boreal and north temperate zones whilst higher values were found in the tropics and the south temperate zones, probably related to temperature(Sanches et al., 2019). Aben (2017) found a strong relationship between CH_4ebullition and temperature across a wide range of shallow freshwater ecosystems on different continents using multi-seasonal CH_4ebullition data from the literature. Freshwaters with high primary production and those that receive substantial loads of allochthonous carbon turned out more likely to have high CH_4 ebullition rates. In combination with controlled year round mesocosm experiments their findings suggest that global warming will strongly enhance freshwater CH_4emissions through a disproportionate increase in ebullition by 6–20% per 1 °C increase in water temperature (Aben et al., 2017). It can be expected that ongoing eutrophication enhanced by climate change-related increases in sediment nutrient release and organic carbon and nutrient loading from catchments (see 3.2.1; 2.3.5.3) will enhance CH_4 ebullition at a global scale (Aben et al., 2017; Sanches et al., 2019). Given that small ponds and shallow lakes, are the most abundant freshwaters globally they may become hot spots of CH_4ebullition in the future (Aben et al., 2017). Uncertainty in CH_4 flux estimates stem from different measuring techniques - as such lake CH_4 flux measurement techniques require thorough re-evaluation (Sanches et al., 2019).

Figure 2.10: [PLACEHOLDER FOR SECOND ORDER DRAFT: burning embers figure of key risks - summary will feed into Chapter 16 burning embers figure]

Table 2.6: [PLACEHOLDER FOR SECOND ORDER DRAFT: Summary of key risks for terrestrial and freshwater ecosystems. [This table is based on the structure of IPCC AR5, Table 4-3, page 302, which IPCC AR6, Chapter 16; additional key risks will be added as literature review continues and evidence base builds]

Confidence, Evidence, Agreement	**Climate factors**	**Non-climate factors**	**Already Detected and Attributed**	**Adaptation options**	**Risk with current adaptation**	**Risk with high adaptation**
Species extinctions. Estimates of risk of species' extinctions from model projections varies considerably (0-54%), but some species are already showing severe declines as well as high risk of extinction from projected loss of climate space and/or habitat, making some level of species' extinctions inevitable. Higher rates of extinctions may be avoided through adaptation measures at low emissions scenarios, but limits to adaptation will become increasingly dominate with higher emissions scenarios (see Sections 2.4.2.6; 2.5.3.3; 2.6.1; 2.6.3)						
High, high, medium	Shifts in geographic placements of climate space; loss	land use change, habitat degradation from	Yes	Habitat restoration, habitat creation, increased	[PLACEHOLDER FOR SECOND ORDER DRAFT]	[PLACEHOLDER FOR SECOND ORDER DRAFT]

	of climate space globally; emergence of non-analogue climates, increases in extreme climate events	pollution, fertilization, invasive species		connectivity of habitats and protected areas, increase in protected areas, assisted colonization		
Tree mortality from drought, wildfire, and pest infestations. Increased drought, wildfire, and pest infestation, due to anthropogenic climate change, and deforestation, forest degradation, and wildfires, due to human land use change, have caused tree mortality in tropical, temperate, and boreal ecosystems. Under continued climate change, models project more extensive mortality in tropical rainforests and temperate conifer forests, releasing carbon dioxide, degrading habitat for plant and animal species, and reducing water supplies for people (see Sections 2.3.6; 2.4.3.2).						
High, High, High	Heat, drought, aridity	Deforestation, land-use change	Yes	Reduce deforestation, reduce use of fire in tropical forests, use prescribed burning in fire-dependent temperate forests	Today: medium 2ºC: high 4ºC: high	Today: medium 2ºC: medium 4ºC: high
Ecosystem carbon loss. Tropical forests and Arctic tundra permafrost contain the greatest stocks of aboveground and belowground ecosystem carbon, respectively. Tropical forests currently emit more carbon to the atmosphere than they remove due to deforestation and forest degradation. Under continued climate change, models project possible tipping points of conversion of Amazon rainforest to grasslands and melting of Arctic permafrost that would release enough carbon dioxide to substantially exacerbate climate change. (see Sections 2.3.6.3; 2.4.3.3.4)						
Medium, Medium, High	Heat, drought, aridity	Deforestation, land-use change	No	Reduce deforestation, reduce use of fire in tropical forests	Today: low 2ºC: medium 4ºC: high	Today: low 2ºC: low 4ºC: high

[START FAQ 2.5 HERE]

FAQ2.5: How do climate-related changes in wildlife affect society and human well-being?

[END FAQ 2.5 HERE]

2.6 Climate Change Adaptation for Terrestrial and Freshwater Ecosystems

This section focuses on human interventions to build the resilience of ecosystems or to adjust management to climate change, in the context of climate resilient pathways and the Sustainable Development Goals (SDGs).

2.6.1 Limits to Autonomous (Natural) Adaptation

Natural ecosystems often have a high degree of resilience and can to some extent adjust to change; species can adjust through evolutionary adaptation, distribution change, behavioural change and developmental plasticity. There are, however, limits to autonomous adaptation, both because of intrinsic limitations and the degraded state of many ecosystems.

None of the evolutionary changes either documented or theorized would enable that species to survive and reproduce in climate spaces that it does not already inhabit. Evolutionary responses are *very unlikely* to prevent species extinctions in the case of that species losing its climate space entirely on a regional or global scale (Parmesan and Hanley, 2015). At highest risk are the world's most cold-adapted species (whose habitats are restricted to sea ice, high boreal and high mountaintop areas). Examples include the polar bear (Regehr et al.), "sky-island" plants in the tropics (Kidane et al., 2019), mountain-top amphibians in Spain (Enriquez-Urzelai et al., 2019), mountain-top lichens in the Appalachians (USA) (Allen and Lendemer, 2016), and silverswords in Hawaii (Krushelnycky et al.).

However, there is a conservation potential for using evolutionary changes to enhance the adaptive capacity of target species, such as is being done in the Great Barrier reef by translocating symbionts and corals that have survived recent intense heat-induced bleaching events into areas that have had large die-off (Rinkevich, 2019). Hoffman and Sgro (Ratnam et al., 2011; Sgro et al., 2011) assessed when and how evolution might be able to help wild species adapt to climate change.

Some of the reasons cited in the literature as limits to autonomous adaptation are:

1) Genetic changes in populations require many generations and for many species operate on longer time scales than those on which the climate is currently changing.

2) Many species are moving to higher latitudes as the climate warms, but not all are keeping pace with changes in suitable climate space (Valladares et al., 2014; Mason et al., 2015). Such climate debt indicates an inability for non-genetic autonomous adaptation (e.g. evidence limited ability for plastic responses, such as stemming from dispersal limitations, or behavioural restrictions, or physiological constraints).

3) Some species have low capacity for dispersal, which, combined with increased fragmentation of habitats, creates barriers to range shifts to match climate warming. Studies have shown that changes in distribution of species and composition of communities are limited by the presence of intensively managed agricultural land fragmenting natural habitats (Oliver et al., 2017).

2.6.2 Adaptation Strategies and Programmes

Ecosystems are specifically included in the adaptation goals set out in the Paris agreement and are addressed in most national adaptation programmes. There are also now a large number of adaptation programmes and plans for local governments and governmental and non-governmental organisations working on ecosystems.

In the context of this chapter there is an important distinction to be made between adaptation measures to maintain biodiversity and ecosystems in their own right and what is often termed Ecosystem-based Adaptation (EbA). EbA is the use of biodiversity and ecosystem services as part of an overall adaptation strategy to help people to adapt to the adverse effects of climate change (CBD, 2009). EbA aims to maintain and increase the resilience and reduce the vulnerability of ecosystems and people in the face of the adverse effects of climate change (Vignola et al., 2009). EbA can contribute to many aspects of sustainable development (Figure 2.11)

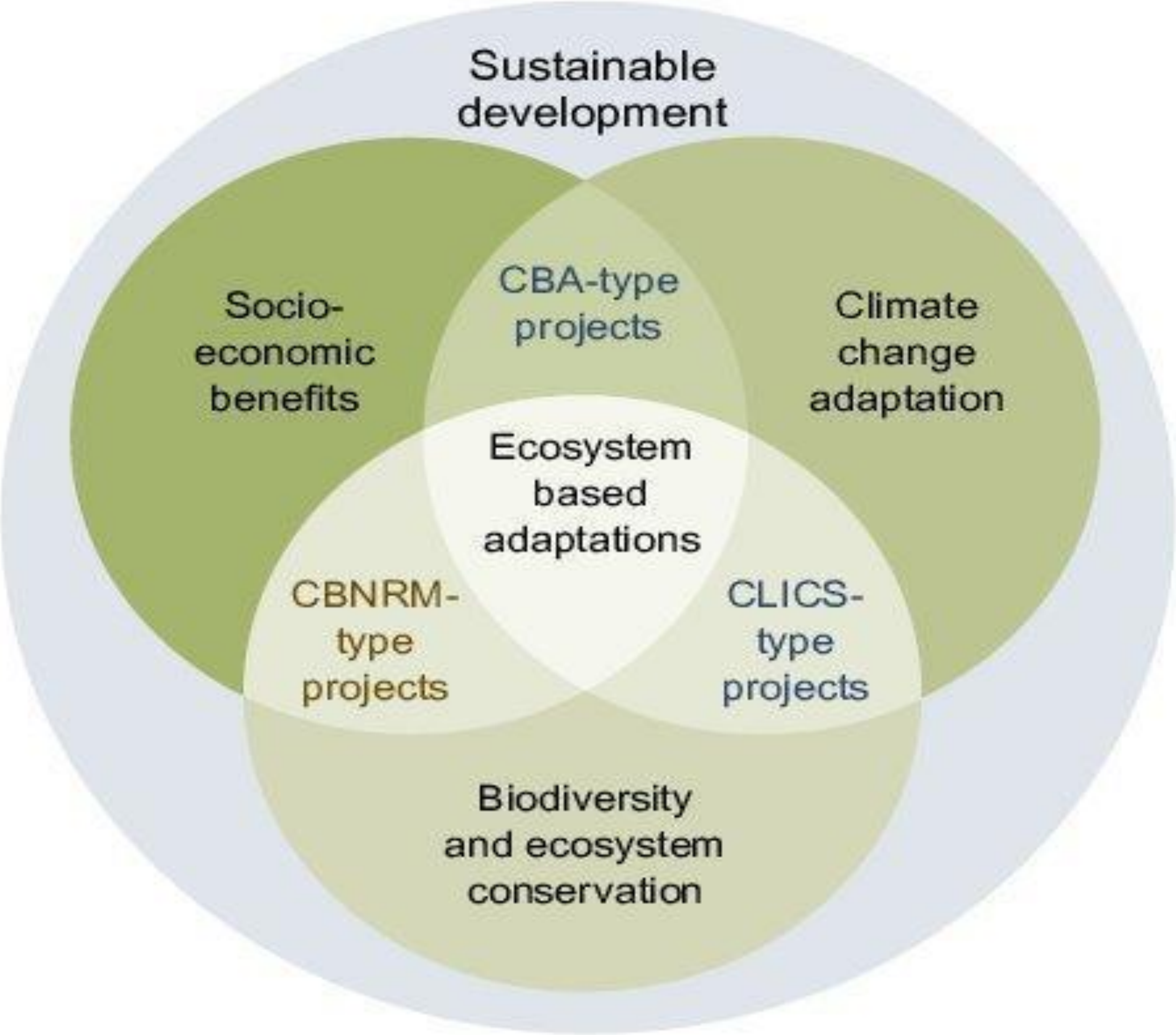


Figure 2.11: Figure adapted from Midgley (2012). Adaptation options relevant to biodiversity and ecosystem conservation within the broader context of sustainable development. CBNRM refers to community based natural resource management [PLACEHOLDER FOR SECOND ORDER DRAFT: citation], while CLICS refers to climate change integrated conservation strategies.

Seddon (2016) reported that of the 162 Intended Nationally Determined Contributions (covering 189 countries) submitted to the United Nations at the time of the research, 109 indicate ‘ecosystem-orientated visions’ for adaptation, although only 23 use the term ‘ecosystem based adaptation.’

Since AR5 a number of studies have considered the wider factors that are important for adaptation programmes and projects. These include:

1) Integration of adaptation planning with other sectors, including agriculture and water resources
2) Partnership working with diverse stakeholders and local communities
3) Good monitoring and evaluation. See the following for a synthesis of 170 case studies: https://unfccc.int/sites/default/files/resource/docs/2015/sbsta/eng/04.pdf

Adaptation is usually context specific and a one size fits all approach is not ideal e.g. forest restoration can be beneficial in forest biomes but not in open grassy ecosystems (see Cross-Chapter Box MITIG in this Chapter).
Better integration of traditional, local & indigenous knowledge & capacity will increase chances of community acceptance and success.

2.6.3 Adaptation for Biodiversity Conservation

A variety of approaches have been identified as potential adaptation measures, over the last 3 decades. Heller and Zaveleta (2009) (quoted in AR5) identified 113 categories of recommendation for adaptation from a survey of 112 papers and reports (Heller and Zavaleta, 2009). Since this time the scientific literature on adaptation for ecosystems has increased dramatically (as illustrated by the fact that Heller and Zavelleta has been cited 1381 times; Google Scholar, 5 April 2019).

The early scientific literature on adaptation was based on ecological theory rather than observations or practical experience. A number of studies since AR5 have investigated the extent to which adaptation has been integrated into conservation planning and is being implemented (e.g. (Macgregor and van Dijk, 2014) (Prober et al., 2019)). There are however relatively few assessments of progress compared to the number of adaptation plans and programmes which have been produced in the academic literature. A recent review (Prober et al., 2019) concluded that out of 473 papers on adaptation, only 16% presented new empirical evidence and there are very few indeed which assess the effectiveness of interventions. There are very few instances where the effect of a planned intervention on the climate sensitivity of an ecosystem is monitored from the start and compared to a control. Nevertheless, since AR5, there have been an increasing number of analyses of how different land use and management influences the vulnerability of species and habitats on the basis of empirical data. This allows at least some of the proposed adaptation measures to be tested for their likely effectiveness. Table 2.7 summarises the evidence that supports the main categories of adaptation measures which have been proposed. We have taken a broad approach and included studies that address extreme weather events such as droughts, which may be exacerbated by climate change as well as long term changes in climate variables. We have not distinguished between studies in which climate change adaptation was an explicit focus and those in which lessons for adaptation can be learnt from studies conducted for other reasons but inform the assessment impacts of actions identified as potential adaptation measures.

[INSERT TABLE 2.7 HERE]

Table 2.7: Evidence to support proposed climate change adaptation measures for biodiversity.

Table 2.7 reflects the fact that adaptation for ecosystems is a broad concept, including a wide range of different actions. It includes targeted interventions to change the microclimate for particular species (for example by shading) through to changing national conservation objectives to take account of changing distributions of species and communities. It includes targeted actions specifically addressing climate change but also protection and restoration of ecosystems, with multiple additional benefits including reducing vulnerability to climate change. As climate change often interacts with other factors including ecosystem degradation and fragmentation (Oliver et al., 2015b), actions to address these other interacting factor is expected to build resilience to climate change. Whilst the evidence base to identify these interactions, for example to show the circumstances in which vulnerability to climate change, or extreme weather events, is reduced, there are few studies testing whether a change in these factors has caused a change in vulnerability. This remains an important evidence gap. There is also a major limitation in the available evidence in that those studies which have been published testing adaptation measures, whether directly or indirectly (through contrasting impacts in different areas) are predominantly from Europe, North America and Australasia.

Many of the actions to adapt to climate change for biodiversity operate at the landscape scale. The total area of habitat, how fragmented it is, the size of habitat patches and the connectivity between them are interlinked properties at this scale. A growing number of studies have investigated how these properties affect species ability both to persist in situ and colonise new areas. Overall larger areas of semi-natural habitat are associated with both increased resilience to ongoing climate change and extreme events and the capacity to colonise new areas (Haslem et al., 2015; Oliver et al., 2017; Papanikolaou et al., 2017). A larger area of habitat enables greater connectivity between patches and / or larger patches and individual patch size and connectivity can also be identified. Large patch size has been found to increase resilience of some populations of species to extreme events such as droughts (Oliver et al., 2015b).

The most consistently cited climate change adaptation measure for species is increasing connectivity to facilitate colonisation of new areas with suitable conditions and there have been advances and innovations in modelling techniques to support this (Littlefield et al., 2019). This reflects the fact many species' habitats are highly fragmented which prevents them naturally changing their range to tracking changing climatic conditions. There is evidence from empirical as well as modelling studies that species can disperse more effectively in better connected areas in terrestrial habitats (Keeley et al., 2018). There are a variety of different approaches to increasing connectivity ranging from increasing overall area of suitable habitat through to 'corridors' and 'stepping stones', with different strategies likely to be more effective for different species and circumstances (Keeley et al., 2018). Connectivity can also be important in increasing resilience of populations to extreme climatic events (Newson et al., 2014; Oliver et al., 2015b). If fluvial corridors are not connected migrating fish populations are not surviving even without climate change but with climate change however connectivity is even more important for organisms to migrate further upstream to colder areas. Connectivity is also important for benthic invertebrates to be able to drift downstream, hence to disperse (Brooks et al., 2018). For adults of benthic invertebrates, a variety of different riparian and terrestrial habitat features can potentially affect dispersal.

The evidence for microclimatic refugia has demonstrated the potential to incorporate this into conservation planning (Jones et al., 2016; Morelli et al., 2016) and this is starting to happen, for example in targeting management actions (Sweet et al., 2019). It is also possible to manipulate microclimate for example by creating shelters for nests (Patino-Martinez et al., 2012) or planting trees to shade water courses (Thomas et al., 2016).

2.6.4 Ecosystem Based Adaptation

Restoration and better management of ecosystems can help people to adapt to climate change as well as the ecosystems themselves. At the same time many of the pressures on ecosystems which prevent adaptation are the result of human activities, so EbA offers the potential to benefit both people and nature. EbA includes a range of different approaches which can benefit both people and nature, examples include restoring coastal and river systems to reduce flood risk and the creation of urban green space to reduce temperatures through shading and evaporative cooling. Since AR5 the evidence base to support ecosystem based adaptation has increased and it is closely linked with a variety of other concepts such as ecosystem services, natural capital and nature-based solutions. Table 2.13 presents an assessment of the evidence supporting the main EbA approaches.

[INSERT TABLE 2.8 HERE]

Table 2.8: Examples of widely advocated Ecosystem Based Adaptation measures with assessments of confidence.

Ecosystem based adaptation was becoming a well recognised concept at the time of AR5 but implementation was still at an early stage in many cases. Since then pilot studies have been assessed and EbA projects have been initiated around the world. The evidence base continues to grow (Table 2.8) and this has consequently led to increasing confidence in approaches which have been shown to work leading to further expansion.

The risks of flooding have increased with climate change as a result of more intense rain storms, hurricanes and rising sea levels and restoring coasts, rivers and wetlands to reduce flood risk have probably seen the largest investment in EbA. This encompasses a wide range of techniques. In tropical and sub-tropical areas, the restoration of mangroves to reduce the risk of coastal flooding is a widely advocated approach supported by evidence (for example (Høye et al., 2013; Sierra-Correa and Kintz, 2015; Powell et al., 2019)). In temperate regions salt marsh is a similarly important habitat (Spalding et al., 2014). Both provide buffering against rising sea levels and storm surges. In river systems (Iacob et al., 2014), the management of catchments and the channel itself is important: restoring natural meanders in canalised water courses and allowing the build up of woody debris can slow flows rates; creating wetlands can store water during flood events.

There are a variety of circumstances in which microclimate and local climate temperatures can be lowered using ecosystem based approaches (Table 2.8) and these techniques are starting to be used more widely. In both urban and agricultural shade trees are a traditional technique which can be applied to contemporary climate change adaptation. Similar shading of water course can lower temperatures to support continuation of important fisheries, including those of Salmonid fish. Within cities, green space, including parks, local nature reserves and green roofs can also provide cooling as a result of evapo-transpiration.

Despite the growth in EbA applications there remain constraints (Nalau et al., 2018) to wider use of it, including in terms of knowledge, governance and integration with society and it is important to approach EbA in the wider context of sustainable development.

2.6.5 Adaptation in Practice: Case Studies and Lessons Learned

Adaptation plans for biodiversity and EbA have been adopted in many countries and by a variety of organisations, land owners and managers; it is however difficult to get a systematic overview of adaptation that is taking place from the academic literature. We have therefore reviewed a sample of case studies, chosen to cover different regions, address a range of issues and at different scales from local to national and using published material in both the journal literature and reports, combined with updates from authors. We present 9 of these case studies below to illustrate the breadth of adaptation and some key issues.

These case studies range from specific local interventions to national strategies and international initiatives. Action on the ground often depends on all of these, for example international funding, directed by a national strategy to support a local project. Within the city of Durban (Roberts and O'Donoghue, 2013), partnership between local communities, local authorities and the academic community were found to be essential, together with an international context. Nevertheless there are examples of local communities using traditional or local knowledge to adapt to changing circumstances as they find them, with little or no external input e.g. (Harvey et al., 2017).

Adaptation for biodiversity conservation ranges from small targeted projects to strategies to integrate adaptation into wider conservation planning. Small scale projects which are specifically adaptation focussed, as for example the transplantation of butterflies in response to shifting ranges or protection of climate sensitive species (Harris, 2018), are rare. However, in countries where nature reserves are actively managed or where ecosystem restoration projects are progressing, local practitioners may well use local knowledge to adapt to weather conditions and their associated effects (fire or water shortage for example). Whilst this is good practice, there is a risk that this will not be sufficiently proactive to address likely future changes in climate. Training and resources to support conservation practitioners in climate change adaptation (for example the Climate Change Adaptation Manual produced by Natural England and the RSPB) are becoming available to help address this.

Adaptation is widely recognised as important for national conservation policy and is being considered in a variety of countries as evidenced by the UK and Thailand being considered in a strategic way in developing protection for National Parks, as shown by the case studies in South Africa and the USA. Adaptation in this strategic context includes a variety of decisions about the selection and objectives for protected areas, for example identifying places where the climate is likely to remain suitable for threatened species, which can act as refugia. It can also mean recognising where a protected area remains important but will support different species or habitats compared to those for which it was originally designated. This is important in terms of directing resources to the places where they can be most effective and ensuring that site management is appropriate in a changing climate. There are however often significant uncertainties and the need for more radical measures will depend on success in reducing greenhouse gas emissions globally. A global rise of 1.5–2 °C would require relatively incremental adjustments to conservation management and spatial targeting of interventions in many parts of the world, but a 3-4°C rise would require radical, transformational changes in approach to maintain many species and habitats and the services they provide to people.

Whilst adaptation strategies for conservation are relatively common, at least at an outline level, implementation is slow in most places. This may partly reflects lack of resources for conservation in many parts of the world, however, one barrier to considering adaptation is that conservationists, local communities and visitors value protected sites in their present form and with their current species. Actions which might jeopardise this are inevitably a last resort. A pathways approach in which alternative adaptation strategies can be adopted with decision points recognised, may be helpful in addressing this but there is little evidence that this approach is being adopted. Initiatives to engage wider communities in discussions about climate change are rare in conservation but are likely to be essential in gaining support for such an approach.

We have distinguished EbA from adaptation for biodiversity alone, but it is important to recognise that the two are intrinsically linked and the largest scale interventions for adaptation in ecosystem have tended to bring together both elements. This is particularly true in the case adaptation to reduce flood risk by habitat creation and using natural processes, including increased risks of coastal flooding from storm surges in relation to rising sea levels and flooding in river systems resulting from more intense storms (Cross-Chapter Box SLR in Chapter 3). Integrating natural ecosystem processes into wider flood risk management, for example by re-naturalising straightened river systems or creating wetlands for water storage (Cross-Chapter Box WATER in Chapter 3), offers the potential more effectively to meet multiple objectives, including nature conservation and has increased overall funding available for ecosystem restoration. Natural flood management has achieved a wide level of acceptance across different countries although it is still used less widely than engineered solutions. Managed realignment projects at the coast and river system restoration are some of the largest adaptation projects in terms of area covered, financial investment and the number of parties involved. They are also often the projects with the most rigorous monitoring and evaluation of effectiveness, this can be done empirically in the case of current conditions and by modelling for potential future impacts. Natural flood management requires a combination of expertise and knowledge on climate, ecology and hydrology together with good local knowledge as many solutions are place specific. This requires interdisciplinary partnership and partnership between scientists, policy makers and practitioners, together with local communities.

EbA also includes a wide range of other solutions to different climate related issues, including providing local cooling for people, crops and livestock. In many farming systems, farmers use local and indigenous knowledge to take advantage of the benefits of shade trees for example in central America (Harvey et al.). This is a relatively straight forward approach with wide potential applicability and there is good potential for knowledge exchange between farmers. There is also the potential to adopt systems like agro-forestry in areas where it has not previously been practised and there are examples of this being trialled. Within urban systems there is increasing interest in the potential for EbA to protect against natural hazards such as drought, flood and heatwave whilst at the same time improving the quality of life. There are examples form cities around the world, such as Durban where this approach is being used.

Across all these areas of adaptation, there is an urgent need for more systematic monitoring and evaluation to allow lessons to be learnt and good practice shared.

2.6.5.1 Case study: Climate Change Adaptation in Conservation in the UK

Scale: National
Issue: Nature conservation (overarching approach)

Climate change poses a number of threats to biodiversity in the UK including: range retraction of cold adapted species, increased coastal erosion and changes in habitats resulting from changing rainfall patterns. All of these are exacerbated by human actions: ecosystems have been heavily modified in the UK with habitats fragmented, land often drained and natural roll back of the coast prevented by hard sea defences.

There is a strong statutory framework for adaptation and the natural environment is a major element of the National Adaptation Programme. Awareness of the need for adaptation to climate change is well established in the UK conservation community, with research on ecosystems and climate change going back to the 1980s, research on adaptation since the 1990s and climate change adaptation principles have been recognised since 2007 (Hopkins et al., 2007). Different conservation agencies cover the different component nations of the United Kingdom (England, Wales, Scotland and Northern Ireland). All have recognised the need for climate change adaptation and have produced extensive information, evidence and resources to support adaptation. For example Natural England has published a Climate Change Adaptation Plan (2015), an Adaptation Manual jointly with the Royal Society for the Protection of Birds – a major conservation NGO (Natural England and RSPB, 2015) and spatial mapping tool for climate change vulnerability (Taylor et al., 2014). The NGO community similarly recognises the need for adaptation.

Surveys by Burch (2014) and (Macgregor and van Dijk, 2014) indicate that in many cases there is an awareness of the need for climate change adaptation amongst conservation practitioners and policy makers, but relatively actions tend to focus on ‘low regrets’ measures which deliver a number of benefits including building resilience to climate change. Nevertheless there are some examples of adaptation in action in the UK, for example managed realignment of the coastline to remove hard sea defences create new coastal habitats. Using natural flood management on river systems is being used in a number of local studies, normally on a pilot study basis. Both of these examples have normally been funded largely on the basis of reducing flood risk to people, in ways that also deliver biodiversity benefits and a growing evidence base to support their effectiveness (https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/654440/Working_with_natural_processes_one_page_summaries.pdf). There has also been restoration of wetlands, especially peatlands which would be expected to build resilience to climate change. Broader adoption of climate change adaptation within conservation is however proceeding more slowly. Approaches include integration within the development of a new Nature Recovery Network encompassing both core conservation sites and small scale habitat patches, increasing connectivity and restoring natural processes; this is still at the planning stage. There is a recognition that it will be necessary to change management objectives of protected sites to reflect changing species distributions and unavoidable changes in vegetation but so far this is at a preliminary stage.

2.6.5.2 Assisted Colonization / Managed Relocation in Practice

Scale: global
Issue: helping species move to track shifting climate space

Managed relocation (assisted migration, assisted colonization) is the movement of species, populations, or genotypes to places outside the areas of their historical distributions to maintain biological diversity or ecosystem functioning with changing climate (Hoegh-Guldberg et al., 2008). It requires careful consideration of scientific, ethical, and legal issues between the object of relocation and the receiving ecosystem (Hoegh-Guldberg et al., 2008; Richardson et al., 2009; Schwartz et al., 2012).

Among species that risk losing their climate spaces if they remain in situ, those least likely to achieve the needed range shifts under their own power are the most obvious candidates for assisted migration. However, even species that are physically capable of long-distance dispersal, such as butterflies, may fail to do so across human-dominated landscapes (Daily and Ehrlich, 1996), in which case expected loss of climate space requires assisted migration or provision of corridors. Because of the number of stepping-stones required for construction of corridors, assisted migration was judged the less expensive of the two options for grassland butterflies in Finland (Tainio et al., 2016).

Since assisted migration beyond current range limits introduces species to habitats where they have not previously been recorded, the potential for negative effects has raised ethical questions (Minteer and Collins, 2010). In ethical support of assisted migration, Siipi and Ahteensuu (2016) defend the moral value of helping species to attain their "natural" ranges in a shifting climate. Lunt (2013) point out that assessments of costs and benefits should include expected positive effects of species translocated poleward in maintaining ecosystem function when they replace other species that are leaving the same communities, while (Thomas, 2011) opines that negative effects are likely to be minimal. Overall, a majority of Conservation biologists have supported incorporating assisted migration into their repertoire of tools, while assessing both the need for it and its expected likelihood of success (Hallfors et al., 2017).

Individual cases show that assisted migration can be successful. Anich & Ward (2017) extended the geographic breeding range of a rare bird, Kirtland's warbler (*Setophaga kirtlandii*), by 225km by using song playbacks to attract migrating individuals. Wadgymar (2015) successfully transplanted an annual legume, *Chamaecrista fasciculata,* to sites beyond its current poleward range limit, while Liu (2012) found that all but one of 20 orchid species survived when transplanted to higher elevations than their current range limits. After introducing two British butterfly species to sites ~65 and ~35 km beyond their poleward range margins, Willis (2009) observed that both introduced populations grew, expanded their ranges and survived for at least 8 years.

Butterflies have been favoured subjects for assisted migration in response to regional climate warming, since they are easy to move and their range dynamics have been extensively studied. The Chequered Skipper butterfly (*Carterocephalus palaemon*) became locally extinct in England in the 1970's, in an area not close to either the species' poleward or equatorial range limits. Nonetheless, Maes (2019) consider climate a crucial parameter for re-introduction, using SDMs both for choosing the source population in Belgium and for predicting that the species should increase at the introduction site between the present and 2070.

Overall, success of assisted migration for conservation purposes has been variable. Bellis (2019), using their own criteria of success, identified 56 successes and 33 failures among 107 translocations of insects that had been undertaken explicitly for Conservation purposes. A useful conclusion from their meta-analysis is that failure was most strongly associated with low numbers of individuals released. Another potential source of failure is local adaptation: there is good evidence that adaptive differences among potential source populations can be important. For example, the transplants of *C. fasciculata* were more successful when sourced from the most poleward existing sites, while individuals from more equatorial habitats performed poorly even when artificially warmed (Wadgymar et al., 2015).

2.6.5.3 Case Study: Conservation of climate change refugia for the Joshua tree (Yucca brevifolia) in Joshua Tree National Park, California, USA

Scale: Local
Issue: Possible extirpation of a plant species within its projected area

Increased aridity with continued climate change under RCP8.5 (Sweet et al., 2019) or A1B (Cole et al., 2011) could nearly eliminate suitable habitat for the Joshua tree (*Yucca brevifolia*), a unique Mojave Desert plant, from Joshua Tree National Park, established for the conservation of the species. Joshua Tree National Park staff have adapted conservation of the Joshua tree by using spatial analyses of suitable habitat under climate change to identify and protect potential refugia for the species (Barrows and Murphy-Mariscal, 2012; Sweet et al., 2019). Climate refugia are locations that potentially retain suitable conditions for a species under climate change. Joshua Tree National Park prioritizes the refugia for control of invasive species and suppression of wildfires in an ecosystem where wildfire, which can kill Joshua trees, has been absent.

2.6.5.4 Case Study: Effects of Climate Change on Tropical High Andean Social Ecological Systems

Scale: local
Issue: complex ramifications of glacial retreat on vegetation, animals and herders

Climate change is transforming high elevation tropical landscapes. Accelerated warming is swiftly reducing tropical glaciers, at rates unseeing since the middle of the Little Ice Age (Thompson et al., 2017). The retreat of the tropical Andes, between 1976 and 2010, is estimated at -0.76 m water equivalent (Vuille et al., 2018). Glacial area in the Cordillera Blanca has shrunk from 723 km^2 to 482 km^2 from 1970s to 2010 (Mark et al., 2017). Large declines in Andean glacier mass and extent have altered surface and ground water flows (Mark et al., 2017). Warming and glacier retreat allow colonization of plants and upward shifts in vegetation communities (Zimmer et al., 2018). In the Andes, the climate-driven upward migration of species has modified its distribution and richness, and the community composition along the Andes altitudinal gradient (Carilla et al., 2018) (Seimon et al., 2017; Zimmer et al., 2018). Drying wetlands has modified alpine plant communities, which are relevant to storing carbon, regulating water, and providing food for local livestock. Glacier retreat substantially impacts water resources and other hydrological ecosystem services including both urban and rural populations, and multiple forms of water use, ranging from human and livestock consumption, to farming, mining and industry (Vuille et al., 2018; Orlove et al., 2019). Less available water has negatively impacted irrigation, and the increasing variability of precipitation has compromised rain-fed agriculture and power generation, particularly in the dry season (Bradley et al., 2006; Bury et al., 2013; Buytaert et al., 2017).

At higher altitudes, climate-driven glacier retreat impacts Andean pastoralists (Postigo, 2013; López-i-Gelats et al., 2016). This retreat has increased meltwater flow regimes (runoff and groundwater) in the short-term

(Polk, 2016) (Young et al., 2017c), with an initial increase in water flow following the glacier melting. However, water flow is already decreasing in some basins and diminishing flows are expected in the near future in many other basins (Baraer et al., 2011) (Vuille et al., 2018; Somers et al., under review). Wetlands are mimicking the pattern of water fluctuation, initially growing and then contracting, leading to negative impacts on herders' livelihoods (Dangles et al., 2017) (Polk et al., 2017). Vicuña (*Vicugna vicugna*) and alpaca (*Vicugna pacos*) are two key elements of these livelihoods because their fibers are the main income sources for the peasant communities and the households respectively. Climate-driven impacts on wetlands, however, are differently affecting the wild vicuña and the domesticated alpaca. The former is adjusting its feeding behaviour to new plant communities and, likely, shifting its spatial distribution as vegetation migrates upwards. In doing so, its vulnerability increases as vicuñas may end up roaming outside the boundaries of the protected areas designed for its conservation and protection from illegal poaching. The alpaca´s responses are driven by the Andean herders. Pastoralist households have been responding to the drying up of grazing areas due to climate change by increasing livestock mobility within their pastures, creating and expanding wetlands through building of several kilometers long irrigation canals, limiting the allocation of wetlands to new households, and sometimes actually cultivating grasses (Postigo, 2013) (López-i-Gelats et al., 2015). These responses, though enabled by adaptive local Andean institutions, are severely compromised by other socio-economic pressures on the resources and capabilities needed for adapting (Valdivia et al., 2010; Postigo, 2019). For instance, mining concessions (<30% and <55% above 4000 m a.s.l. and 3000 m a.s.l. respectively) on the headwaters of the watersheds threaten the quality and limit the access and control over water (Bebbington and Bury, 2009). Further, there is competition for water that is desired for irrigation of agribusiness on the lowland coastal desert (Mark et al., 2017).

2.6.5.5 Case Study: Protecting Gondwanan refugia against fire in Tasmania

Scale: local
Issue: protection of rare endemic species

The Tasmanian Wilderness World Heritage Area (TWWHA) has a high concentration of 'paleo-endemic' plant species restricted to cool, wet climates and fire free environments, but recent wildfires have burnt substantial stands, which are unlikely to recover (Harris et al., 2018b). The fires led to government inquiries and a fire-fighting review, which have suggested changes on the assumption that climate change will make such fires likely to keep happening. During fires in 2019, significant effort and resources were spent trying to protect the remaining stand, including by installing dripline irrigation around stands of the relictual vegetation – an interesting case study because it raises the question of management intervention in wilderness areas and an example where changing policy and planning can be definitely linked to climate change. Could also be linked to indigenous knowledge as there is an ongoing discussion of using Aboriginal burning techniques in the wilderness area.

The TWHHA is managed as a Wilderness Zone, where management is carried out in manner that allows natural processes to predominate. The exclusion of fire from stands of fire-sensitive trees such as the Pencil pine (*Athrotaxis cupressoides*) is part of this management strategy. However, in recent years, the threat posed by extensive and repeated wildfires, and an increasing awareness that fire risk is likely to continue increasing with climate change (Fox-Hughes et al., 2014; Turco et al., 2018) have meant that more direct management intervention has been implemented. There has been a realisation that a "hands-off" approach to managing the threat will not be sufficient to protect the paleo-endemics. Not only is fire-fighting difficult in the remote wilderness area, but limited resources mean that fire managers must prioritise where fires will be fought when many fires are threatening towns and lives across the state simultaneously. After wildfires in 2016 caused extensive damage, new approaches were used in 2019, including the strategic application of long-term fire retardant and the installation of kilometres of sprinkler lines, approaches which are thought to have been effective at halting the fire and protecting the high value vegetation. Impact reports are currently being finalised to quantify the extent of fire-sensitive vegetation communities that have been affected. However, there is concern that these interventions may have adverse effects on the values of the TWHHA if applied widely, so while research is ongoing, these will only be applied in strategic areas (e.g., fire retardant is not being applied to some areas).

The TWWHA Management Plan (2016) emphasises Aboriginal fire management as an important value of the TWWHA, along with their knowledge of plants, animals, marine resources, minerals (ochre and rock sources), and their connection with the area as a living and dynamic landscape. Fire management planning aims to protect important sites from fire and ensure that management does not impact Aboriginal cultural values. Increasingly, there is an acknowledgement that the cessation of traditional fire uses has led to changes in vegetation and a call to incorporate Aboriginal burning knowledge into fire management of the TWHHA.

2.6.5.6 Case Study: Protected areas planning in response to climate change in Thailand

scale: national
issue: protected area network planning

The existing protected areas in Thailand cover approximately 21% of the country land area, which is greater than the average percentage of protected areas of in ASEAN countries (13%). In addition, it is one of the few tropical countries that quantitatively passes the Aichi Target 11 of at least 17% of the terrestrial area protected. However, most protected areas were established on an *ad hoc* basis to protect remaining forest cover, therefore they may not be resilient to the interacting impacts of land use and climate change by 2050. Recent research conducted in northern Thailand indicated that the existing protected areas (31% of the region area) cannot secure viability of many medium-and large-mammal species. Most species would substantially shift their current distributions and will have higher risk to extinction. The model results based on the spatial distribution model and network flow determined there was a need for expansion areas of 5,200 km^2 or 3% of the region to substantially minimize the risk level and increase the average coping capacity of the protection of suitable habitats from 82% as the current plan to 90% (Trisurat, 2018).

A similar approach was also conducted in Yunnan, China (Zomer et al., 2014) and in three tropical bio-geographic realms (Neotropics; Afrotropics; Indo-Malayan tropics) regional and global level under the Spatial Planning for Protected Areas in Response to Climate Change (SPARC) project executed by the Conservation International (CI). These two studies also indicated that future climate change (2050) will decrease the effectiveness of existing protected areas network.

2.6.5.7 Case Study: Ecosystem based adaptation in Durban, South Africa

Scale: local
Issue: Ecosystem based adaptation in a city and surrounding area

Durban was an early pioneer of EbA in a city context, establishing a Municipal Climate Protection Programme (MCPP) in 2004 (Roberts et al., 2012). Durban has a population of approximately 3.5 million, which is growing rapidly and experiences a vareity of serious challenges including high rates of poverty, unemployment and health problems (Roberts and O'Donoghue, 2013). The adaptation work stream within the MCPP is composed of three separate components: municipal adaptation (adaptation activities linked to the key line functions of local government); community-based adaptation, focused on improving the adaptive capacity of local communities and a series of urban management interventions that address specific challenges associated with climate change such as the urban heat island, increased stormwater runoff, water conservation and sea-level rise. EbA elements are included in all of these elements. An approach. The approach which has been taken has had a philosophy of 'learning by doing'. Local governments, including Durban do not typically have all of the necessary skills to implement EbA and partnerships have been essential, including with the local university. Roberts (Roberts et al., 2012) noted that 'While EBA may provide a cost-effective approach to climate change adaptation, some level of capital and operational funding is still required' and that 'the protection of the required ecosystems will necessitate tough decision-making and significant political and administrative will'. Evidence of the success of green infrastructure measures included measurements showing green roof reduces stormwater run-off by approximately 60 ml/m2/minute during a rainfall event, releases water slowly over time and reduces temperature on the surface of the roof by up to 30°C (Roberts et al., 2012). Despite being one of the better examples of monitoring and evaluation, (Roberts et al., 2012) still note the need for better monitoring of the effectiveness of adaptation measures. The engagement of the local community is a key element of EbA in the city and the concept of Community Ecosystem Based Adaptation has been developed.

2.6.5.8 Case Study: Adaptation Management in South African National Parks

Scale: national
Issue: adaptation planning across national park network

South African National Parks have recorded mean temperature increases in excess of 1.5°C over the last 60 years, and the number of days in which temperatures exceeded 35°C and fell lower than 0°C have markedly increased and decreased respectively (van Wilgen and Wannenburgh, 2016). Understanding how such climatic changes are impacting their effectiveness, and how negative implications can be minimised is recognised as essential.

A framework for climate change adaptation planning is being developed for South African National Parks. An iterative process guides setting objectives, assessing vulnerabilities (based on predicted exposure, sensitivity and adaptive capacity), designing adaptation strategies, implementing and reviewing them. To date, SANParks' assessment of protected area vulnerability has focused on three approaches. Firstly, current and predicted climate change have been quantified within each national park and how these interact with other global change drivers has been assessed. Secondly predictions of species and ecosystem vulnerability to climate change have been developed through Spatial Planning for Protected Areas Species in Response to Climate Change (SPARC) project. This has included species distribution models, dynamic global vegetation models, trait-based assessments and network flow. Thirdly, park-level climate change vulnerabilities have been assessed, including consideration of climate change impacts on biodiversity, infrastructure and tourism/income generation, as well as on park-specific adaptation capacity. Several key vulnerabilities have been identified including biodiversity losses, bush encroachment, infrastructure at risk from floods, revenue loss and possible increases in human-wildlife conflict.

SANParks has begun using these vulnerability assessments and consultation with park management to prepare a draft climate change preparedness strategy (van Wilgen and Wannenburgh, 2016). The potential actions being explored include improving landscape connectivity, minimising non-climatic stressors on climate change vulnerable species, disaster risk planning and management, Ecosystem-based Adaptation, assisted translocation and minimising greenhouse gas emissions. Since effectiveness of both novel and existing management approaches is unknown in the climate change context, empirical assessment and knowledge-sharing of their impacts and effectiveness is essential. Through proactive planning for climate change, SANParks hopes to minimise negative impacts, seize emerging opportunities, and pave the way for protected area custodians to prepare for climate change.

2.6.5.9 Cross Cutting Themes

Adaptation is taking place in a wide variety of contexts for a wide variety of reasons. This includes actions focussed on biodiversity and ecosystem based adaptation. Adaptation can take place at a range of scales with specific projects nested within overarching national strategies. Small scale projects can be largely adaptation focussed, but in larger scale projects, adaptation is often integrated into wider restoration or environmental land management objectives. To be effective both approaches are likely to be necessary. Ecosystem based adaptation often focuses on flooding issues. Integrating environmental land and catchment management into wider flood management offers the potential to use resources more effectively to meet multiple objectives. This has increased overall funding available for ecosystem restoration.

Adaptation often requires a combination of expertise and knowledge on climate, ecology and hydrology together with good local knowledge as many solutions are place specific. This requires interdisciplinary partnership and partnership between scientists, policy makers and practitioners. There is an urgent need to evaluate the effectiveness of interventions as adaptation measures. This needs to cover a range of outcomes for species, ecosystems and species and include assessments of costs and benefits.

2.6.6 Limits to Adaptation

Much emphasis has been given to building resilience to reduce vulnerability of ecosystems to climate change. However some changes are inevitable and as described above have already happened *(strong evidence, high agreement)*. An important element of adaptation is therefore to recognise inevitable change and incorporate it into planning conservation and land management. The extent to which this has happened is not known.

There is a conservation potential for using evolutionary changes to enhance the adaptive capacity for target species, such as is being done in the Great Barrier reef by translocating symbionts and corals that have survived recent intense heat-induced bleaching events into areas that have had large die-off. However, known limitations to genetic adaptation preclude species-level adaptation to climates beyond their ecological and evolutionary history (see Sections 2.2.4.6; 2.6.1)

2.6.7 Climate Resilient Pathways

The interactions between social and ecological systems is essential for developing climate resilient pathways and progressing towards the Sustainable Development Goals.

In addition to the direct effects of climate change on ecosystems there are a wide range of human responses to climate change that will have an impact of ecosystems, some positive, some negative.
If mitigation or adaptation are approached in a siloed way, there is the potential for maladaptation with adverse consequences for ecosystems and people. For example the use of 'hard' engineering to reduce flood risk in response to increasing storm events, may prevent river systems from naturally adapting, leading to a loss of biodiversity and not prove effective in the long-term as climate change progresses. As discussed above EbA which works with natural processes is increasingly widely advocated as an alternative, although knowledge gaps remain as to the extent to which this can deliver effective flood protection (Ngai et al., 2017).

To keep global temperature rise well below 2°C and pursue efforts to limit it to 1.5°C, it will almost certainly be necessary to take advantage of the potential for ecosystems to sequester carbon. However, the way in which this is done will make a big difference to natural and semi-natural ecosystems. Most scenarios for meeting the Paris Agreement targets include the deployment of Bioenergy with Carbon Capture and Storage (BECCS) whilst this both replaces fossil fuels and removes carbon from the atmosphere, it will increase pressure on land which could alternatively be used to grow food or provide other ecosystem services. It may also directly or indirectly lead to the cultivation of natural areas which support biodiversity. Similarly plantation forest managed for high timber productivity will contribute to mitigation goals but support less biodiversity than more natural forests. It is important that decision making takes account of the full range of ecosystem services and the benefits which protection and restoration of natural forests and other carbon rich ecosystems can have for climate change mitigation (Lewis et al., 2019) (Griscom et al., 2017). It is also important to ensure that forests are planted in naturally forested biomes – the effects of afforestation in savannah can be very maladaptive with adverse consequences for biodiversity and people (Cross-Chapter Box MITIG in this Chapter). As well as forest, peatlands are particularly important because of the high amount of carbon they store and are priorities for protection and restoration (Leifeld and Menichetti, 2018). There is increasing evidence that intertidal habitat restoration as well as providing adaptation through reducing coastal flood protection also sequesters carbon (Burden et al., 2019) and provides a range of benefits for local communities.

There is a linkage between many of these restoration approaches to climate change adaptation and mitigation and the concept of rewilding. The concept of rewilding is very broad, including the restoration of natural hydrological processes in coastal and river system wetlands. There is also emerging evidence of a range of benefits from rewilding approaches involving the reintroduction of megafauna. For example large-scale megafrugivore rewilding programmes in forests can support the recruitment of many hardwood forest species is dependent on large mammals for dispersal and seed germination (Blake et al., 2009; Cromsigt et al., 2018). The loss of these hardwood species would cause carbon losses as large as 2–12% in the majority of the world's tropical forests. Studies from a rewilding project in the Russian Arctic by Zimov (2009) show that restoring the megafauna of the far north reduce woody encroachment, keeping albedo low and reducing melting of permafrost and preventing carbon emissions (te Beest et al., 2016; Buotte et al., 2017; Cromsigt et al., 2018).

[START BOX 2.2 HERE]

Box 2.2: Risks to Biodiversity of Mitigation

[PLACEHOLDER FOR SECOND ORDER DRAFT: new literature to be added]

Many of the models showing pathways to reach global temperatures not exceeding 1.5°C require large amounts of bioenergy with carbon capture and storage (IPCC 1.5). This sets up a competition between land for bioenergy, food, and biodiversity (Smith et al., 2018).

Article 2 compliant mitigation is defined as mitigation that does not interfere with the efforts to reach a global temperature level "within a time frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure that food production is not threatened and to enable economic development to proceed in a sustainable manner" (Smith et al., 2018). However, studies of the potential competition between land for bioenergy, food and biodiversity show that biodiversity is impacted. One study that examined the combined impacts of climate change and land use change for bioenergy on biodiversity found that larger proportions of species, and smaller range species would be impacted more than by climate change alone (Hof et al., 2018). A study on the potential impacts of bioenergy production and climate change on European birds found that land conversion for biodiversity to meet a 2°C target would have greater impacts on species range loss than a global temperature increase of 4°C, if bioenergy were the only mitigation option (Meller et al., 2015).

[END BOX 2.2 HERE]

2.7 Research Gaps and Priorities

The detection of changes in biodiversity and ecosystems and their attribution to anthropogenic climate change comprised a substantial gap in ecological research identified in the Fifth Assessment Report (Cramer et al., 2014; Settele et al., 2014) and the Special Report on 1.5ºC Warming (Hoegh-Guldberg et al., 2018). While many changes in biodiversity and ecosystems are well-documented, attribution to anthropogenic climate change remains an important research need, particularly since many effects are caused by deforestation, agricultural expansion, urbanization, overfishing, and human actions other than climate change (Jones et al., 2018). A meta-analysis of published species range shifts suggests that extirpations of populations of over 400 species were related to climate change (Wiens, 2016). Projections of species extinctions under projected future climate change have been quantified (Urban, 2015; Warren et al., 2018). This reflects a general abundance of research on future vulnerabilities and risks of ecosystems to climate change relative to research on historical impacts.

There are a number of research shortfalls based on current state of the art methods relating to distributions of species range shifts under climate change for both flora and fauna such as: investigation on distribution changes in tropical lowlands and tropical waters; investigation of lowland range shifts of terrestrial plants; investigation of distribution changes of prokaryotes; use of multi-facetted approaches for more comprehensive assessments, only 10% have used a multidimensional approach focusing on at least two geographical dimensions (e.g. latitude and elevation), to assess distribution shifts or abundance changes; use of local climate velocities to improve expectations for biological shifts (Lenoir and Svenning, 2015). Tree mortality presents another area of ecology exhibiting a research gap in detection and attribution. While drought-induced tree mortality has become widespread (Allen et al., 2010; Hartmann et al., 2018), just a few cases, in western North America (van Mantgem et al., 2009), the African Sahel (Gonzalez et al., 2012), and North Africa (le Polain de Waroux and Lambin, 2012), have been formally attributed to anthropogenic climate change. Global trends in tree mortality and their potential to have ecological and climatological consequences remain highly uncertain at both community levels and relevant temporal scales (hundreds of years for trees) (Hartmann et al., 2018).

Due to climate change, insect pests are likely to increase, resulting in crop loss (Deutsch et al., 2018) and decrease in native biodiversity. However, most of the studies research has been restricted to aboveground pests, whereas little attention has been given to potential impact of climate change on below-ground process including soil-borne pests (Pritchard, 2011; Chakraborty et al., 2013; Juroszek and Von Tiedemann, 2013).

There were multiple gaps in the body of literature assessing the impacts of climate change on ecosystem services. Cultural services were particularly under-represented. Although uncertainty was usually incorporated, there were substantial gaps with respect to underlying causes, along with the methods used to incorporate them (Runting et al., 2017). A key research question is understanding the mechanisms of change for multiple drivers as linkages between the combined impacts of changes in climate, land-water use, and biodiversity on the future supply of ecosystem services remain largely unknown (Elmhagen et al., 2015). Other questions included evaluating the extent and trend of climate change impacts on ES considering different climate change scenarios; assessing vulnerability of important ES that are likely to be sensitive to climate change and; developing ecosystem and ES-specific adaptations to climate change that involve different stakeholders (Alamgir et al., 2014).

The challenges included low availability of data at matching temporal and spatial scales and; uncertainties in future projections, in particular connected to uncertainties regarding future local-to-global social-ecological development (Elmhagen et al., 2015). There are few studies relating to integrated decision making, and even fewer studies aimed to identify solutions that were robust to uncertainty (Runting et al., 2017).

The predominance of research on biodiversity and climate change in Europe and North America highlights a gap in the geographic coverage of research (Tydecks et al., 2018). Potentially substantial climate change risks in tropical forests (Cusack et al., 2016) and tropical mountains (Cronin et al., 2014; Costion et al., 2015) suggest the need for greater coverage in future research. Impact of climate change on interaction between the variability of groundwater and ecosystem productivity of tropical forests is not known.

The response of plant transpiration to increased CO2, climate warming and changes in soil moisture and groundwater elevation must be understood and included in recharge models. Most studies of climate change effects on surface hydrology in alpine, mountainous and snow-dominated regions do not explore subsurface hydrological responses. The impacts of frost on soil hydraulic conductivity and recharge are large, but not fully understood. The quantification of climate change impacts on groundwater and groundwater dependent ecosystem can be explored by running groundwater models with future meteorological boundary conditions (Kløve et al., 2014).

The emergence of the use of paleoecological information to inform contemporary biodiversity conservation under anthropogenic climate change suggest the need for research to fill this disciplinary knowledge gap (Barnosky et al., 2017; Nolan et al., 2018). Moreover, since the case of rapid evolution of one owl species in Finland in response to climate change (Karell et al., 2011), not much research has examined genetic adaptation to climate change.

[INSERT TABLE 2.9 HERE]

Table 2.9: Proposed research priorities terrestrial ecosystems in the face of global change.

Economic losses from weather extremes are particularly in the spotlight. Over the last decades, losses from natural disasters including those from events related to extreme weather have strongly increased (Mechler and Bouwer, 2015). There is a need for better assessment of global adaptation costs, funding and investment (Micale et al., 2018). Potential synergies between international finance for disaster risk management and adaptation have not yet been fully realized. Few researchers have paid attention in incorporating the climate risks factor in the spatial development planning; i.e. many efforts have been made towards the climate-change hazard modeling but limited research in the risk analysis and adaptation strategy (Zhou et al., 2019). Research has almost exclusively focused on normalizing losses for changes in exposure, yet not for vulnerability, which appears a major gap given the dynamic nature of vulnerability (Mechler and Bouwer, 2015).

Decision makers need updated results of scientific studies on certain species or ecosystems of societal value. Observational studies and experiments are indispensable, but they can hardly cover the entire diversity and space. In consequence, research activities have to concentrate on key taxa, ecosystems, and processes. Conceptual links to future modelling approaches would be helpful as only models can deliver rapid approximations of ecological climate change impacts and therewith the identification of adaptation needs (Karell et al., 2011; Anja, 2014; Micale et al., 2018).

[START CROSS-CHAPTER BOX MITIG HERE]

Cross-Chapter Box MITIG: Forests as Mitigation: Positive and Negative Consequences for Adaptation

Authors: Nicola Stevens, Mike Morecroft, Camille Parmesan, Dave Schoeman, Aditi Mukherji, Timon McPhearson, Chris Trisos, Brendan Mackey, Daniela Schmidt, Jeff Hicke, Polly Buotte, Rodel Lasco, Lisa Schipper, Minal Pathak

Natural ecosystems are important contributors to climate change adaptation and mitigation (Griscom et al., 2017; Lewis et al., 2019)(IPCC Land report, 2019) and many future terrestrial emission reduction scenarios include reduced deforestation (Griscom et al., 2017), large-scale reforestation and afforestation efforts (Laestadius et al., 2011; Griscom et al., 2017; Bastin et al., 2019). Protecting existing forests and preventing further degradation is a key starting point: the IPCC Special Report on Land (IPCC, 2019) reported that this had the potential to reduce global emissions by 0.4-5.8 Gt CO_{2e} yr^{-1}. There is also significant potential for forest restoration, reforestation and afforestation: IPCC (2019) estimated the potential carbon uptake as 1.4-10.1 GtCO2e yr^{-1} for restoration and reforestation and 0.5-8.9 Gt CO_{2e} yr^{-1}for afforestation of previously unforested land. There are already programs including The Bonn Challenge, which propose to reforest degraded and deforested lands in biomes around the world. There are potential co-benefits to woodland creation for adaptation in many cases, including soil stabilisation (Thomas et al., 2016) and reducing flood risk through slowing water flow (Dixon 2016).

However, for these adaptation benefits to be realised it is important to locate appropriate types of forest in the right places as forestation can be maladaptive. Not every site identified for reforestation is similarly suitable (Veldman et al., 2017; Bond et al., 2019). Many areas of low tree cover are classified as 'deforested' and 'degraded which have been identified by comparing the vegetation biomass climatic potential of a system against the actual vegetation biomass. Regions where actual biomass is below the potential are regarded as degraded/deforested (Laestadius et al., 2011; Veldman et al., 2017). This bioclimatic type approach assumes low tree cover is evidence of deforestation/degradation and mistakenly misidentifies ~ 1 billion hectares of the world's grassy biomes as products of recent anthropogenic origin (Veldman et al., 2017; Aleman et al., 2018) and therefore suitable for "reforestation". Instead these are ancient non-forested landscapes which support important biodiversity and ecosystem services (Veldman et al., 2015c; Murphy et al., 2016). If these regions erroneously earmarked for tree planting programs and widespread afforestation the loss of biodiversity and ecosystem services of the landscapes could be profound (Veldman et al., 2017; Bond et al., 2019).

We highlight that before implementing widespread reforestation/afforestation plans, trade-offs between adaptation and mitigation need to be considered. Increasing the area of forests will have impacts, and we highlight that impacts on albedo, biodiversity, water balances, other ecosystem services, and land-ownership need to be considered to ensure co-benefits to both adaptation and mitigation.

When does forestry as mitigation to support positive adaptation measures? Avoiding forest loss

Preventing forest loss and managing existing forest serve as a win-win for both adaptation and mitigation. Primary forests harbour irreplaceable biological diversity(Gibson et al., 2011), regulate hydrological regimes (Ellison et al., 2017), moderate regions against climate change impacts (Locatelli et al., 2015; Ellison et al., 2017) and critically stabilize the terrestrial carbon sink (Pan et al., 2011; Mackey et al., 2013). Prioritising the principle of avoided forest lost will maximise benefits to adaptation and biodiversity

Reforestation

Reforestation, especially in areas that retain existing native forest fragments, offers multiple synergistic benefits with adaptation and mitigation. Reforestation options extend the range from natural regeneration, assisted restoration, enrichment planting, native tree plantations, commercial plantations, and agroforestry systems in areas that have been deforested. Arguments indicate that reforestation of native forest, as opposed to commercial plantations, can result in the highest in accumulation of carbon. Additionally it can promote biodiversity recovery (Lewis et al., 2019), enhance hydrological processes (Ellison et al., 2017) particularly through improved water filtration, and groundwater recharge (Ellison et al., 2017) and can reduce risk of soil erosion and flood risk [[PLACEHOLDER FOR SECOND ORDER DRAFT: List of chapter boxes that contain examples to be included] and enhance other ecosystem services (Locatelli et al., 2015). In these cases management of reforestation the case for incorporating a climate-smart reforestation strategy has been highlighted. This process involves planting a species mix to minimise impacts of future climate change on reforestation success (Locatelli et al., 2015).

In urban contexts, where adaptation is needed to protect some of the most vulnerable human populations, forests including street trees are important for climate adaptation in urban contexts especially for providing local cooling through shading and evapotranspiration. Shading reduces mean radiant temperature, which is the dominant influence on outdoor human thermal comfort under warm, sunny conditions (Weydmann et al., 2018). Apart from lowering temperature, urban trees may also contribute to lower energy costs by reducing demand for conventional sources of cooling (e.g. air conditioning), especially during peak- demand periods. Homes with shade trees that are located in cities where air conditioning systems are common can save over 30% of residential peak cooling demand, reducing energy use. However, local cooling by urban trees depends on regional climate context, geographic setting of the city, urban form, the density and placement of the trees, in addition to a variety of other ecological, technical, and social factors. For example, forest fragments less than 0.5-2.0 ha may have negligible cooling effects, beyond the shaded area itself.

Water implications of afforestation and reforestation

Afforestation and reforestation (A/R) are often considered one of the most cost effective ways of storing carbon (Hoegh-Guldberg et al., 2018; Smith et al., 2018; Bastin et al., 2019); and as a result, current discourses on A/R has been centered around carbon storage and sequestration. Yet, the water benefits (both water quantity and quality) and water footprint (water costs) of A/R are not always considered. A new global assessment on forest-water interactions underscored that forests influences the entire water cycle at local, regional and global scales-- both downstream -- through rainfall-evapotranspiration (ET)-runoff dynamics; and downwind via recycled rainfall effects (Creed and van Noordwijk, 2018). The downstream impacts of forests, particularly native forests, on local hydrology are relatively well understood, and comprises of reduction in surface runoff in the immediate vicinity of the forests; more infiltration to groundwater and improved water quality (Bruijnzeel, 2004; Zhou et al., 2015; Ellison et al., 2017; Alvarez-Garreton et al., 2019; Evaristo and McDonnell, 2019). The science of downwind impacts of forests in terms of moisture recycling is still being explored and recent evidence shows that downwind precipitation depend on upwind ET from forests and other vegetation (Keys et al., 2016; Ellison et al., 2017). An important implication of both downstream and downwind influence of forests on local hydrology and global/regional water cycle is that this knowledge can be used to influence the location of AR interventions. Sites where water supply is abundant (to reduce negative local impacts on reduced streamflow) or where transpired water can be recirculated as rainfall due to downwind effect (Creed and van Noordwijk, 2018).

When do forestry approaches as mitigation result in maladaption?

Many of the targeted areas for tree planting programs (see http://www.wri.org/applications/maps/flr-atlas/#. Bastin (2019)) erroneously assume that low tree cover, in climates that can support forests, are deforested and 'degraded' (Figure MITIG1). This misreading of the landscape means that many of the world's grassy ecosystems (grasslands, savannas, shrublands and woodlands) are earmarked for afforestation. Afforestation of grasslands prioritises carbon sequestration services at the cost of multiple ecosystem services that grasslands provide (water, grazing land, nature based tourism, biodiversity). These grassy ecosystems are ancient (Bond and Zaloumis, 2016) with an evolutionary history shaped by interactions between grasses, fires, and herbivores (Scheiter and Higgins, 2009; Maurin et al., 2014; Charles-Dominique et al., 2016). They already are threatened by policies of fire exclusion (Durigan and Ratter, 2016; Abreu et al., 2017) and will be severely impacted by tree planting efforts. Carbon-focused tree planting will exacerbate biodiversity threats and result in widespread biodiversity losses, particularly of species adapted to open environments (Bremer and Farley, 2010; Veldman et al., 2015a; Veldman et al., 2015b; Murphy et al., 2016; Abreu et al., 2017; Veldman et al., 2017). These efforts also threaten important ecosystem service provision like forage for livestock on which many people rely (Parr et al., 2014; Veldman et al., 2015c; Ryan et al., 2016). Converting grassy ecosystems to forests, most of which are in worlds drylands, will reduce stream flow and deplete ground water (Jackson et al., 2005; Cusack et al., 2016; Zhang et al., 2016; Veldman et al., 2017). Swapping grassy ecosystems for forests may also serve to change the albedo and contribute to warming as forests absorb more incoming radiation than grasslands (Creutzig et al., 2015; Baldocchi and Penuelas, 2019). Aside from the negative impacts to adaptation it is also questionable just how much carbon can be sequestered in these landscapes (Baldocchi and Penuelas, 2019). Afforestation of grassy ecosystems can reduce carbon gains (Veldman et al., 2015b; Veldman et al., 2017) as grassy ecosystems make a significant contribution to the carbon sink and have high rates of belowground carbon sequestration (Veldman et al., 2015b; Zaloumis and Bond, 2016). Furthermore a high belowground carbon store prevents carbon loss to fire in these fire-prone environments.

As with forests, we propose that a win:win outcome in terms of both positive adaptation and positive mitigation can be achieved in these grassy ecosystems by taking a restoration approach and actively restoring degraded grassy ecosystems. As with the restoration of forests, restoration efforts in grassy ecosystems can recreate vegetation structure (i.e. historical tree density and herbaceous ground cover), but species-diverse plant communities, including endemic species, are slow to recover restoration. Restoration approaches that have been most successful include: 1) removal of invasive and alien trees, 2) management of woody plant encroachment (widespread removal of trees), 3) reintroduction of a prescribed burning regime, 4) grazing management, 5) reseeding of grassy species, and 6) soil management following soil disturbance. These activities have the benefit of improving biodiversity, enhancing ecosystem service provision and contributing to the global carbon store.

Considerations for tree planting schemes

1) Using maps of global biomes or maps of distribution of grassy ecosystems (Figure MITIG1) determine if the region of interest falls into what is defined as a forest or a grassy ecosystem.

2) In cities and urbanized systems, locations for planting new trees for both mitigation and local adaptation depends strongly on local social and infrastructural factors that determine where trees can be added. Choice of species is critical given the combination of pressures on urban trees from insect herbivory, disease, changing climate conditions, and direct human impact.

3) Is there a high level of diversity and endemism in the regionß

4) What do the plant traits of the system tell you: In grassy ecosystems look for a suite of traits that indicate if the system is a fire maintained system or is a fire sensitive close forest (Ratnam et al., 2011) (for a full list see Ratnam et al., 2011). As an example these traits include thick bark, open canopied trees, large underground storage

5) Are open environment specialists present: Many open environments across the world are host to light loving plants which contribute to a rich biodiversity in the region. It is helpful to assess what open environment specialists occur in the area. Open areas can be characterised by a high diversity of forbs. In meadows a high number of orchids may occur. In tropical grassy plants with specialised growth forms may also occur e.g. underground trees and forbs and bulbs with well-developed underground storage systems

6) Consider water implications downstream and downwind impacts of forests on local, regional and global water cycles, which also provides guidance on best possible sites for A/R interventions.

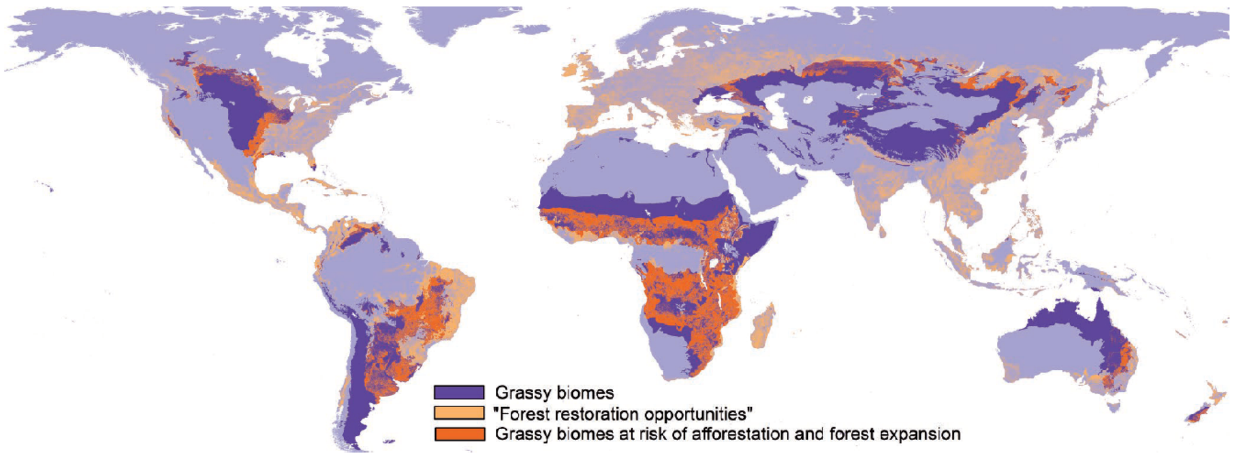


Figure MITIG1: A global map highlighting where open grassy ecosystems are at risk from "reforestation" and "afforestation" activities (Adapted from Veldman et al., 2015b)

References

Abreu, R. C. R. et al., 2017: The biodiversity cost of carbon sequestration in tropical savanna. Science Advances, 3 (8), e1701284.

Aleman, J. C., M. A. Jarzyna and A. C. Staver, 2018: Forest extent and deforestation in tropical Africa since 1900. Nature Ecology & Evolution, 2 (1), 26.

Alvarez-Garreton, C., A. Lara, J. P. Boisier and M. Galleguillos, 2019: The Impacts of Native Forests and Forest Plantation on Water Supply in Chile. Forests, 10 (6), 473.

Baldocchi, D. and J. Penuelas, 2019: The physics and ecology of mining carbon dioxide from the atmosphere by ecosystems. Global Change Biology.

Bastin, J.-F. et al., 2019: The global tree restoration potential. Science, 365 (6448), 76-79, doi:10.1126/science.aax0848.

Bond, W. and N. P. Zaloumis, 2016: The deforestation story: testing for anthropogenic origins of Africa's flammable grassy biomes. Philosophical Transactions of the Royal Society B: Biological Sciences, 371 (1696), 20150170.

Bond, W. J., N. Stevens, G. F. Midgley and C. E. R. Lehmann, 2019: The Trouble with Trees: Afforestation Plans for Africa. Trends in Ecology & Evolution.

Bremer, L. L. and K. A. Farley, 2010: Does plantation forestry restore biodiversity or create green deserts? A synthesis of the effects of land-use transitions on plant species richness. Biodiversity and Conservation, 19 (14), 3893-3915.

Bruijnzeel, L. A., 2004: Hydrological functions of tropical forests: not seeing the soil for the trees? Agriculture, Ecosystems & Environment, 104 (1), 185-228.

Charles-Dominique, T. et al., 2016: Spiny plants, mammal browsers, and the origin of African savannas. Proceedings of the National Academy of Sciences, 113 (38), E5572-E5579.

Creed, I. F. and M. van Noordwijk, 2018: Forest and Water on a Changing Planet: Vulnerability, Adaptation and Governance Opportunities: A Global Assessment Report. International Union of Forest Research Organizations (IUFRO).

Creutzig, F. et al., 2015: Bioenergy and climate change mitigation: an assessment. Global Change Biology Bioenergy, 7 (5), 916-944, doi:10.1111/gcbb.12205.

Cusack, D. et al., 2016: Global change effects on humid tropical forests: Evidence for biogeochemical and biodiversity shifts at an ecosystem scale. Reviews of Geophysics, 54 (3), 523-610, doi:10.1002/2015RG000510.

Durigan, G. and J. A. Ratter, 2016: The need for a consistent fire policy for Cerrado conservation. Journal of Applied Ecology, 53 (1), 11-15.

Ellison, D. et al., 2017: Trees, forests and water: Cool insights for a hot world. Global Environmental Change-Human and Policy Dimensions, 43, 51-61, doi:10.1016/j.gloenvcha.2017.01.002.

Evaristo, J. and J. J. McDonnell, 2019: Global analysis of streamflow response to forest management. Nature, 570 (7762), 455-461, doi:10.1038/s41586-019-1306-0.

Gibson, L. et al., 2011: Primary forests are irreplaceable for sustaining tropical biodiversity. Nature, 478 (7369), 378.

Griscom, B. W. et al., 2017: Natural climate solutions. Proceedings of the National Academy of Sciences, 114, 11645-11650.

Hoegh-Guldberg, O. et al., 2018: Impacts of 1.5ºC global warming on natural and human systems. In: IPCC special report on the impacts of global warming of 1.5 °C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change. http://www.ipcc.ch/report/sr15/.

IPCC, 2019: Summary for Policy Makers: IPCC Special Report on Climate Change and Land [Ipcc (ed.)]. International Panel on Climate Change, Geneva, in press.

Jackson, R. B. et al., 2005: Trading water for carbon with biological carbon sequestration. Science, 310 (5756), 1944-1947.

Keys, P. W., L. Wang-Erlandsson and L. J. Gordon, 2016: Revealing invisible water: moisture recycling as an ecosystem service. PLOS ONE, 11 (3), e0151993.

Laestadius, L. et al., 2011: Mapping opportunities for forest landscape restoration. Unasylva (English ed.), 62 (238), 47-48.

Lewis, S. L., C. E. Wheeler, E. T. A. Mitchard and A. Koch, 2019: Restoring natural forests is the best way to remove atmospheric carbon. Nature Publishing Group.

Locatelli, B. et al., 2015: Tropical reforestation and climate change: beyond carbon. Restoration Ecology, 23 (4), 337-343.

Mackey, B. et al., 2013: Untangling the confusion around land carbon science and climate change mitigation policy. Nature Climate Change, 3 (6), 552.

Maurin, O. et al., 2014: Savanna fire and the origins of the ‘underground forests’ of Africa. New Phytologist, 204 (1), 201-214.

Murphy, B. P., A. N. Andersen and C. L. Parr, 2016: The underestimated biodiversity of tropical grassy biomes. Philosophical Transactions of the Royal Society B: Biological Sciences, 371 (1703), 20150319.

Pan, Y. et al., 2011: A large and persistent carbon sink in the world’s forests. Science, 333 (6045), 988-993.

Parr, C. L. et al., 2014: Tropical grassy biomes: misunderstood, neglected, and under threat. Trends in Ecology & Evolution, 29 (4), 205-213.

Ratnam, J. et al., 2011: When is a ‘forest’ a savanna, and why does it matter? Global Ecology and Biogeography, 20 (5), 653-660, doi:10.1111/j.1466-8238.2010.00634.x.

Ryan, C. M. et al., 2016: Ecosystem services from southern African woodlands and their future under global change. Philosophical Transactions of the Royal Society B: Biological Sciences, 371 (1703), 20150312.

Scheiter, S. and S. I. Higgins, 2009: Impacts of climate change on the vegetation of Africa: an adaptive dynamic vegetation modelling approach. Global Change Biology, 15 (9), 2224-2246, doi:10.1111/j.1365-2486.2008.01838.x.

Smith, P. et al., 2018: Impacts on terrestrial biodiversity of moving from a 2°C to a 1.5°C target. Philosophical Transactions of the Royal Society A, 376.

Thomas, S. M., S. W. Griffiths and S. J. Ormerod, 2016: Beyond cool: adapting upland streams for climate change using riparian woodlands. Global Change Biology, 22 (1), 310-324.

Veldman, J. W. et al., 2015a: Toward an old-growth concept for grasslands, savannas, and woodlands. Frontiers in Ecology and the Environment, 13 (3), 154-162.

Veldman, J. W. et al., 2015b: Tyranny of trees in grassy biomes. Science, 347 (6221), 484-485.

Veldman, J. W. et al., 2015c: Where tree planting and forest expansion are bad for biodiversity and ecosystem services. BioScience, 65 (10), 1011-1018.

Veldman, J. W. et al., 2017: Grassy biomes: An inconvenient reality for large-scale forest restoration? A comment on the essay by Chazdon and Laestadius. American Journal of Botany, 104 (5), 649-651.

Weydmann, A., W. Walczowski, J. Carstensen and S. Kwaśniewski, 2018: Warming of Subarctic waters accelerates development of a key marine zooplankton Calanus finmarchicus. Global Change Biology, 24 (1), 172-183, doi:10.1111/gcb.13864.

Zaloumis, N. P. and W. J. Bond, 2016: Reforestation or conservation? The attributes of old growth grasslands in South Africa. Philosophical Transactions of the Royal Society B: Biological Sciences, 371 (1703), 20150310.

Zhang, C., M. Jansen, L. De Meester and R. Stoks, 2016: Energy storage and fecundity explain deviations from ecological stoichiometry predictions under global warming and size-selective predation. Journal of Animal Ecology, 85 (6), 1431-1441, doi:10.1111/1365-2656.12531.

Zhou, Z. et al., 2015: Responses of alpine grassland to climate warming and permafrost thawing in two basins with different precipitation regimes on the Qinghai-Tibetan Plateaus. Arctic, Antarctic, and Alpine Research, 47 (1), 125-131.

[END CROSS-CHAPTER BOX MITIG HERE]

[START CROSS-CHAPTER BOX EXTREMES HERE]

Cross-Chapter Box EXTREMES: Ramifications of Climatic Extremes for Marine, Terrestrial and Freshwater Systems

Authors: Rebecca Harris), Rita Adrian, Philip Boyd, Sarah Cooley, Simon Donner, Mette Mauritzen, Camille Parmesan, Marie-Fanny Racault, Bjoern Rost, David Schoeman

Extreme weather events (events that are rare within their statistical reference distribution at a particular place (IPCC, 2014) such as heatwaves, droughts, floods, storms, and tropical cyclones have become more frequent and more intense in many regions of the world as a result of anthropogenic climate change (Seneviratne et al., 2012; Ummenhofer and Meehl, 2017) [[PLACEHOLDER FOR SECOND ORDER DRAFT: see WG1 AR6 Chapter 9]). This trend is projected to increase (e.g., Section 3.2.2.1, Cross-Chapter Box SLR in Chapter 3, link to chapters).

In recent years there has been a rash of extreme weather events that have resulted in serious and wide-ranging ramifications on marine (Holbrook et al., 2019), terrestrial (Thornton et al., 2014) and freshwater (Huber et al., 2012; Kuha et al., 2016) systems. It has been suggested that it is the combined impact of long-term climate trends and episodic extreme weather events that is responsible (the 'press and pulse' framework (Harris et al., 2018a). The most well-known examples include the northeast Pacific 'Blob' and the Great Barrier Reef bleaching, but strong biological responses to extreme events are now being recorded in almost all ecosystems of the world (Harris et al., 2018a; Marcé et al., 2019; Maxwell et al., 2019; Smale et al., 2019).

In many cases the magnitude of the transient changes in extreme events exceeds that of projected mean changes for the year 2100, regardless of emissions scenario (Figure EXTREMES1). For example, sea surface temperatures (SSTs) up to 6ºC warmer than the long-term mean have been recorded in the North Pacific 'blob'. Between 1982-2016 the number of days with marine heatwaves has doubled (Frolicher and Laufkotter), even though the ocean warmed only by ~0.42ºC on average (Huang et al.). Extremes may therefore 'short-circuit' longer-term projected change. Although basic research demonstrates that extreme events are drivers of major changes in abundances in wild species (Parmesan et al., 2000), most models of species and ecosystem responses to climate change are based on mean climate conditions and do not consider the combined effects of extremes and changes to the background climate. Model projections may therefore substantially underestimate the potential for biological responses. This has implications for the future condition of marine, terrestrial and freshwater life (e.g., Chapter 2, Sections 3.4.2.1, 3.4.2.3, 3.4.2.5, 3.4.4.1), their ability to adapt, biogeochemical feedbacks and human adaptation to such events.

Recent observed responses to extremes have widespread ramifications for biodiversity, tourism, fisheries, forestry and ecosystem services, such as erosion control, carbon storage, nutrient cycling and water quality. For example, there has been a proliferation of extreme events in the ocean, such as marine heatwaves, in some cases resulting in the tropicalisation of food webs (ref, Section 3.4.2.7, (Hidalgo et al., 2018). In terrestrial systems there is evidence of the role of droughts and extreme floods on the C cycle (i.e., flood events and transport of matter into rivers; dry river beds as carbon dioxide emitters associated with riverine systems (Marcé et al., 2019). Warming induced temperature stress and loss of aerobic habitat has caused fish kills in rivers and lakes. Extremely hot summers have led to unprecedented outbreaks of food poisoning and bacterial infections in Alaska and Fenno-Scandia (see Cross-Chapter Box ILLNESS in Chapter 3). Increased frequency and magnitude of high latitude storms and tropical cyclones over the oceans and land have also had major consequences for ecosystems and livelihoods.

Extreme events are identified as key determinants of change in many (all?) Sectoral and Regional Chapters due to their importance in the marine, terrestrial and freshwater realms and in all regions of the world. This Cross-Chapter Box explores the convergences and divergences in our understanding of how extreme events evolve and propagate, and how biota in terrestrial, freshwater and marine ecosystems respond to these events. The aim is to describe the main features of extreme events that influence biological responses and suggest common metrics of extremes – such as timing, duration, intensity, geographical extent – that will bring together our knowledge from the land, freshwater and ocean systems. It will serve as a common conceptual starting point, pointing to Sections in individual Chapters for further detail.

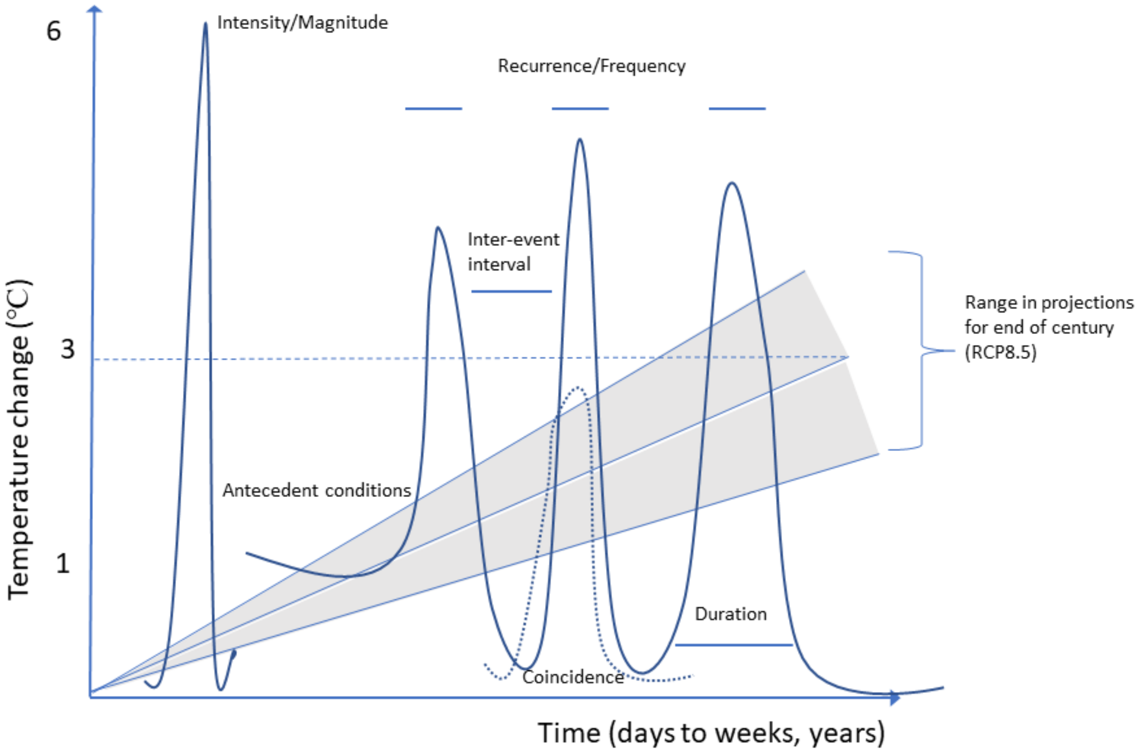


Figure EXTREMES1: Metrics observed for three extreme events that have led to extreme biological responses in marine, terrestrial and freshwater systems. The marine example is the North Pacific Blob, where temperatures of 6ºC above mean conditions have been recorded. The terrestrial example could be repeated fires in obligate seeder forests. The freshwater example could be an episodic storm or flood event (hours to days), a heatwave (weeks to months), or an ice free winter in the Northern Hemisphere (months) in relation to the mean global projections for 2100. Temperature is used to illustrate the concepts, but the same metrics apply to other extremes such as drought, wind storms and floods. [PLACEHOLDER FOR SECOND ORDER DRAFT: This conceptual figure will be replaced with one based on observed metrics for each case study. Figure EXTREMES1 will describe the characteristics of three extreme events that have led to biological responses (1 marine, 1 terrestrial and 1 freshwater example) in relation to the mean global projections for 2100. It will highlight the fact that the magnitude of recent extreme events exceeds the conditions that are projected to occur by the end of the century under the high emissions scenario and the characteristics of extreme events that are associated with increased impact on natural systems.]

The impact of an extreme event is a function of the characteristics of the event as well as those of the ecosystem being exposed to the event. So the timing, frequency, intensity and geographic extent of the extreme event, relative to the lifecycle, resistance and resilience of the natural abiotic and biotic systems, all determine the severity of the biological response (Hillebrand et al., 2018). These concepts are equally applicable to human systems, offering opportunities to elucidate some of the hidden limits to adaptive capacity in social systems.

There are many features in common across the marine, terrestrial and freshwater realms. Parallels in the observed responses of biota and ecosystems across the three systems allow the identification of characteristics that may be associated with greater vulnerability. These include the spatial scale of the ecosystem and dispersal ability of species relative to the extreme event and connectivity, which influence recolonization and recovery; the level of habitat or diet specificity in the community and the complexity of the food web; and the taxonomic, phylogenetic and functional diversity and redundancy of the system, which affect key functions such as habitat provision and trophic flow (e.g., few species at critical mid-trophic levels in pelagic systems). In all systems, impacts on engineering species (e.g., kelp, corals in the marine realm; dominant tree species in the terrestrial; keystone species such as filter feeders or picivorous fish, macrophytes in lakes) and interactions between invasive and range-shifting species affect recovery, and can lead to ecosystem tipping points, beyond which the system may not recover. [Section 2.5.3; SROCC Chapter 6; AR6 sectoral chapters].

Finally, in all systems, it is possible to generalise about the factors that increase the likelihood of adverse impacts on natural systems. Impacts appear to be greater [add confidence statement] when i) extreme events occur with greater frequency, particularly when the interval between events is insufficient to allow recovery to previous population sizes (e.g., frequent fire, even when populations are adapted to cope with fire, or frequent bleaching, in the case of corals); ii) events occur over large spatial areas, reducing the potential for recolonization from nearby populations (e.g., regional droughts that cause large declines in most populations); iii) the magnitude of an extreme event exceeds historical levels, so organisms are less likely to be adapted to them; iv) several extremes coincide (e.g., low sea level, high surface and sea water temperatures, drought) (add refs to seagrass and mangroves in W Australia); v) ecosystems are already under stress or disturbed by habitat fragmentation, land use change, nutrient pollution or water extraction from river systems; vi) extreme events coincide with vulnerable life cycle stages; vii) cascading interactions and feedbacks occur within a system (e.g.. range shifting leads to changes to competitive interactions; run-off pulses from the watershed leads to changes in competitive interaction in lake phytoplankton due to changes in light (browning) and nutrient ratios; (Adrian and Hessen, 2016); extreme events occur near range edges, where species are already close to one limiting factors (Arafeh-Dalmau et al., 2019) and viii) fundamental physiological thresholds are exceeded (e.g., during heat events) (refs to be added for each).

Extreme events are defined in all systems in both absolute and relative terms. The number of days exceeding a physiological threshold is useful when the thermal tolerance of species is known (e.g., 42°C for flying foxes (Welbergen et al., 2007), but such tolerances are not often known. Definitions in relative terms are useful because organisms are adapted to local levels of climate variability, so the magnitude of the deviation from the mean has the greatest biological impact.

While many similarities exist in the characteristics of extreme events and biological systems across the realms, there are also differences that affect sensitivity to extremes and our understanding and ability to predict responses. Heatwaves are the most commonly recorded extreme in marine ecosystems, with a few instances of corrosive ocean acidification extremes. In contrast, on land there is a wider range of extreme event types reported in addition to heatwaves, such as wildfire, drought and cyclones, while freshwater systems are heavily impacted by heat waves, storms, drought and floods, which tend to interact with non-climate drivers such as eutrophication and storms (sections 2.3.1, 2.3.5). As a result, there is a greater range in extreme event indices and methods to assess extreme events against the background of climate trends and variability in the terrestrial and freshwater worlds. Similar metrics are applied across the realms for heatwave, although the terrestrial may be more embedded because the fire literature has developed many of the concepts (e.g., inter-fire interval, intensity).

In lakes, heat waves cause extended and more stable thermal stratification with subsequent oxygen depletion in the deep water and subsequent release of nutrients from the sediments. Storm events cause changes in the depth of the thermocline causing nutrient entrainment from the deep water body to the upper water body. In both cases nutrient pulses are likely to positively affect algal growth (Adrian and Hessen, 2016). Coastal areas are also facing strong and interacting stressors, including wastewater discharge, chemical pollution and habitat degradation, which, in combination with floods and cyclones, are causing extreme responses in coastal systems (through river runoffs, mixing of sediments).

The relative importance of local, regional and global drivers of extreme events differs across marine, terrestrial and freshwater systems, affecting the predictability of potential biological impact. The marine system is more integrated at the global scale, with greater inertia in the drivers related to oceanic forcing compared to atmospheric forcing, and substantially longer lag times. In contrast, freshwater systems are more likely to be affected by regional drivers, and, depending on the characteristics of the waterbody (e.g., depth, size), may respond on shorter timescales (e.g.. Kuha et al., 2016).

Large scale extremes in climatic signals such as the North Atlantic Oscillation and El Niño Southern Oscillation have been shown to synchronize changes in spring phenology in marine and freshwater (Adrian and Hessen, 2016; Racault et al., 2017a). Such synchrony is not found for weather extremes. Marine ecosystem responses to extreme types of El Niño (Canonical and Modoki) are observed worldwide but with considerable and sometimes opposing regional effects (Racault et al., 2017b). However, extreme events are coupled across realms. For instance, marine heatwaves and associated atmospheric pressure systems have contributed to severe droughts, extreme precipitation, flooding and increased cyclone intensity (SROCC chapter 6).

While our understanding of the propagation of extreme events is improving, our ability to predict where and when an extreme biological response might occur remains limited. Recent events have improved our understanding of extreme responses in natural systems. However, it remains challenging to predict which systems are most at risk to extreme events, for several reasons. First, extreme weather events result in abrupt ecosystem changes over short response times, which can be highly nonlinear when compared with responses to the ongoing climate trend (Harris et al., 2018a; Beaugrand et al., 2019). Recovery times can be very long and there is the potential for lag and legacy effects in ecosystem processes. Finally, responses are taxonomically and geographically idiosyncratic, and often there is a lack of biological understanding of the mechanisms driving the responses. Resistance, resilience and recovery times to extreme events is influenced by biological characteristics such as dispersal ability, competitive advantages, reproduction, functional redundancy and growth, as well as general health of the systems (i.e., degraded systems are be more vulnerable to extreme events). However, data to back these general assumptions is not always available, and there are documented exceptions to many of them. The potential for ecological and biogeochemical feedbacks and mechanisms underlying the responses (e.g., individual, coupled or clustered) is also still often not well understood.

Further, not all extreme events have a biological impact (Bailey and van de Pol, 2016). For instance, an extreme wind event may have little impact on phytoplankton in a lake which was fully mixed prior to the event, whereas storm effects on phytoplankton community assembly may compound when lakes are not yet recovered from a previous storm or if periods of drought alternate with periods of intense precipitation, potentially eroding ecosystem resilience. Hence the timing of storm events, and antecedent conditions, may greatly influence the ecological impact of storms (Perga et al., 2018).

Identifying key precepts of the responses to extreme events in marine, terrestrial and freshwater systems and consolidating common concepts, definitions and examples may improve the ability to identify susceptible ecosystems and appropriate management responses.

Table EXTREMES1: State of knowledge of extreme events in marine, terrestrial and freshwater systems. References are grouped as M, marine, T, terrestrial and F, freshwater.

Property	Marine	Terrestrial	Freshwater	Notes/refs
Definition	>5 days duration > 90th percentile of temperature	Absolute and relative indices (> 90th/95th or <10th/5th percentiles) of temperature and precipitation	Absolute and relative indices (> 90th/95th or <10th/5th percentiles) of temperature, wind precipitation, shear stress, sediment resuspension	M: (Hobday et al., 2016) F: (Jennings et al., 2012; Havens et al., 2016) T: to be added
Altered properties	Marine heatwaves, episodic upwelling of corrosive low pH waters; low O_2 levels caused by interaction of biotic and abiotic conditions (e.g., “black tides”); mean changes versus variability	Heatwaves, wildfires, drought, wind, high and low precipitation	Water temperature, light, mixing regime, nutrient dynamics in lakes; flow dynamics, matter transport, metabolism in rivers	M: (Holbrook et al., 2019; Oliver, 2019) F:(Jennings et al., 2012; Bernhardt et al., 2018). T: to be added
Drivers	Climate mode phases (internal variability) for heatwaves (T); coastal upwelling (OA)	Climate mode phases (internal variability) for heatwaves and drought, fire weather	Air temperature-, wind, precipitation- extremes	M: (Holbrook et al., 2019) F: (Jennings et al., 2012; Bernhardt et al., 2018) T: to be added

Predictability	Predictability of (influential) climate modes	Predictability of (influential) climate modes	Predictability of (influential) climate modes	M: (Gauthier et al., 2013; Zhang and Ren, 2017)
Unknowns	Interactions of multiple climate modes; interplay of local and large-scale events; interaction between biotic and abiotic conditions (e.g., "black tides": link between extreme rainfall/ heatwaves, bacterial and plankton growth)	Interactions of multiple climate modes and with other human drivers; physiological thresholds for most species; differences in resilience for degraded vs healthy systems	Role of timing, magnitude, frequency of events and pre-and post event conditions for resistance and resilience in response; interaction of multiple environmental modes	M: (Holbrook et al., 2019) F: (Hampton et al., 2017)

Table EXTREMES2: Characteristics of extreme events that determine biological responses. References are grouped as M, marine, T, terrestrial and F, freshwater. Assessment will incorporate relevant literature and table will be formatted to highlight the commonalities and differences across the realms once assessment is completed.

Property	**Marine**	**Terrestrial**	**Freshwater**	**Notes/refs**
Magnitude of the extreme		Relative to historical distribution to incorporate local adaptation	Relative to historical mean and magnitude	
Abruptness of event onset		Not generally considered		
Duration of event	Days to months to years	Days exceeding physiological threshold (eg. flying fox T) or arbitrary temperature	Hours to months to years (storms, precipitation, droughts)	M: (Smale et al., 2019) T: to be added F: (Jennings et al., 2012)
Spatial extent	Coastal regions to ocean basins Vicinity of 'warm range edges' of resident biota (T)	Potential for recolonization; vicinity of 'warm range edges' of resident biota	Across lake area and depth, across river catchments and river beds	M: (Smale et al., 2019) T: to be added F: (Jennings et al., 2012; Marcé et al., 2019)
Intermittency/heterogeneity of the event (composite metric to capture recurrence and magnitude)	Environmental Heterogeneity of marine heatwave		Intermittency/heterogeneity is the rule rather than the exception	M: to be added T: to be added F: to be added
Recurrence	Increasing frequency in some regions, e.g., the Mediterranean, Back to back heatwaves in GBR	Increased frequency of fire, flood	Frequent storm-precipitation, hot-extremes	M: (Hughes et al.; Hobday et al., 2018; Hughes et al., 2018) T: to be added F: (Perga et al., 2018)
Interval between occurrence	Years	Recovery from the event (obligate seeders and recurrent fires)	Recurring extreme events may erode ecosystem resilience	F: (Perga et al., 2018) M:(Hobday et al.)
Timing of event		Important in relation to life history, phenology,	Change in flood timing, reduction in ice	

	determines susceptibility	development - ice free winters in north temperate zone	
Seasonality – winter versus summer extremes and their implications	Vulnerability of life stages	Loss of ice in winter causes change in phytoplankton trait composition, more intense stratification in summer causes algal bloom formation, propagating effects from winter conditions into the following summer	M: to be added T: to be added F: (Huber et al., 2012; Jennings et al., 2012; Ozkundakci et al., 2016; Hampton et al., 2017)
Coincidence of >1 extreme	Drought, high SST, air temperature and low sea level (e.g., Mangroves);	Storms associated with short term run-off events and physical mixing of water column	M: to be added T: to be added F: to be added
Interactions and feedbacks	With other stressors and disturbance	With other stressors and disturbance (mixing/flow regime, nutrient loads)	
Intensity	Link to thresholds		
Antecedent conditions	Fire weather, Drought leading to dry soils	Freshwater stratification	

References

Adrian, R. and D. O. Hessen, 2016: Environmental Impacts—Lake Ecosystems. In: North Sea Region Climate Change Assessment [Quante, M. and F. Colijn (eds.)]. Springer, Springer International Publishing AG Switzerland.

Arafeh-Dalmau, N. et al., 2019: Extreme Marine Heatwaves Alter Kelp Forest Community Near Its Equatorward Distribution Limit. *Frontiers in Marine Science*, 6, doi:10.3389/fmars.2019.00499.

Bailey, L. and M. van de Pol, 2016: Tackling extremes: challenges for ecological and evolutionary research on extreme climatic events. *Journal of Animal Ecology*, 85 (1), 85-96, doi:10.1111/1365-2656.12451.

Beaugrand, G. et al., 2019: Prediction of unprecedented biological shifts in the global ocean. *Nature Climate Change*, 9 (3), 237-+, doi:10.1038/s41558-019-0420-1.

Bernhardt, E. et al., 2018: The metabolic regimes of flowing waters. *Limnology and Oceanography*, 63, S99-S118, doi:10.1002/lno.10726.

Frolicher, T. L. and C. Laufkotter, 2018: Emerging risks from marine heat waves. *Nature Communications*, 9, doi:10.1038/s41467-018-03163-6.

Gauthier, G. et al., 2013: Long-term monitoring at multiple trophic levels suggests heterogeneity in responses to climate change in the Canadian Arctic tundra. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 368 (1624), 20120482.

Hampton, S. et al., 2017: Ecology under lake ice. *Ecology Letters*, 20 (1), 98-111, doi:10.1111/ele.12699.

Harris, R. M. B. et al., 2018: Biological responses to the press and pulse of climate trends and extreme events. *Nature Climate Change*, 8 (7), 579-587, doi:10.1038/s41558-018-0187-9.

Havens, K. et al., 2016: Extreme Weather Events and Climate Variability Provide a Lens to How Shallow Lakes May Respond to Climate Change. *Water*, 8 (6), doi:10.3390/w8060229.

Hidalgo, M., V. Mihneva, M. Vasconcellos and M. Bernal, 2018: Climate change impacts, vulnerabilities and adaptations: Mediterranean Sea and the Black Sea marine fisheries. In: Impacts of Climate Change on fisheries and aquaculture: Synthesis of current knowledge, adaptation and mitigation options. FAO Fisheries and Aquaculture Technical Paper No. 627 [Manuel Barange, T. B. M. B. S. F.-S. F. P. (ed.)].

Hillebrand, H. et al., 2018: Decomposing multiple dimensions of stability in global change experiments. *Ecology Letters*, 21 (1), 21-30, doi:10.1111/ele.12867.

Hobday, A. et al., 2018: Climate change impacts, vulnerabilities and adaptations: Australian marine fisheries. In: Impacts of Climate Change on fisheries and aquaculture: Synthesis of current knowledge, adaptation and mitigation options [Barange, M. e. s. (ed.)], FAO 1-18. **http://hdl.handle.net/102.100.100/86715?index=1**.

Hobday, A. J. et al., 2016: A hierarchical approach to defining marine heatwaves. *Progress in Oceanography*, 141, 227-238, doi:10.1016/j.pocean.2015.12.014.

Holbrook, N. J. et al., 2019: A global assessment of marine heatwaves and their drivers. *Nature Communications*, 10, doi:10.1038/s41467-019-10206-z.

Huang, B. Y. et al., 2015: Extended Reconstructed Sea Surface Temperature Version 4 (ERSST.v4). Part I: Upgrades and Intercomparisons. *Journal of Climate*, 28 (3), 911-930, doi:10.1175/jcli-d-14-00006.1.

Huber, V., C. Wagner, D. Gerten and R. Adrian, 2012: To bloom or not to bloom: contrasting responses of cyanobacteria to recent heat waves explained by critical thresholds of abiotic drivers. *Oecologia*, 169 (1), 245-256, doi:10.1007/s00442-011-2186-7.

Hughes, T. P. et al., 2017: Global warming and recurrent mass bleaching of corals. *Nature*, 543 (7645), 373-+, doi:10.1038/nature21707.

Hughes, T. P. et al., 2018: Global warming transforms coral reef assemblages. *Nature*, 556 (7702), 492, doi:10.1038/s41586-018-0041-2.

Jennings, E. et al., 2012: Effects of weather-related episodic events in lakes: an analysis based on high-frequency data. *Freshwater Biology*, 57 (3), 589-601, doi:10.1111/j.1365-2427.2011.02729.x.

Kuha, J. et al., 2016: Response of boreal lakes to episodic weather-induced events. *Inland Waters*, 6 (4), 523-534, doi:10.5268/iw-6.4.886.

Marcé, R. et al., 2019: Emissions from dry inland waters are a blind spot in the global carbon cycle. *Earth-Science Reviews*, 188, 240-248, doi:10.1016/j.earscirev.2018.11.012.

Maxwell, S. L. et al., 2019: Conservation implications of ecological responses to extreme weather and climate events. *Diversity and Distributions*, 25 (4), 613-625, doi:10.1111/ddi.12878.

Oliver, E. C. J., 2019: Mean warming not variability drives marine heatwave trends. *Climate Dynamics*, 53 (3-4), 1653-1659, doi:10.1007/s00382-019-04707-2.

Ozkundakci, D. et al., 2016: Winter severity determines functional trait composition of phytoplankton in seasonally ice-covered lakes. *Global Change Biology*, 22 (1), 284-298, doi:10.1111/gcb.13085.

Parmesan, C., T. L. Root and M. R. Willig, 2000: Impacts of extreme weather and climate on terrestrial biota. *Bulletin of the American Meteorological Society*, 81 (3), 443-450.

Perga, M. et al., 2018: Storm impacts on alpine lakes: Antecedent weather conditions matter more than the event intensity. *Global Change Biology*, 24 (10), 5004-5016, doi:10.1111/gcb.14384.

Racault, M. F. et al., 2017a: Impact of El Nino Variability on Oceanic Phytoplankton. *Frontiers in Marine Science*, 4, doi:10.3389/fmars.2017.00133.

Racault, M. F., S. Sathyendranath, N. Menon and T. Platt, 2017b: Phenological Responses to ENSO in the Global Oceans. *Surveys in Geophysics*, 38 (1), 277-293, doi:10.1007/s10712-016-9391-1.

Seneviratne, S. I. et al., 2012: Changes in climate extremes and their impacts on the natural physical environment. Cambridge University Press, Cambridge, UK, and New York, NY, USA, pp. 109-230.

Smale, D. A. et al., 2019: Marine heatwaves threaten global biodiversity and the provision of ecosystem services. *Nature Climate Change*, 9 (4), 306-+, doi:10.1038/s41558-019-0412-1.

Thornton, P. K., P. J. Ericksen, M. Herrero and A. J. Challinor, 2014: Climate variability and vulnerability to climate change: a review. *Global Change Biology*, 20 (11), 3313-3328, doi:10.1111/gcb.12581.

Ummenhofer, C. C. and G. A. Meehl, 2017: Extreme weather and climate events with ecological relevance: a review. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 372 (1723), doi:10.1098/rstb.2016.0135.

Welbergen, J. A., S. M. Klose, N. Markus and P. Eby, 2007: Climate change and the effects of temperature extremes on Australian flying-foxes. *Proceedings of the Royal Society B: Biological Sciences*, 275 (1633), doi:https://doi.org/10.1098/rspb.2007.1385.

Zhang, C. and W. Ren, 2017: Complex climatic and CO2 controls on net primary productivity of temperate dryland ecosystems over central Asia during 1980–2014. *Journal of Geophysical Research: Biogeosciences*, 122 (9), 2356–2374.

[END CROSS-CHAPTER BOX EXTREMES HERE]

References

Abatzoglou, J. T. and A. P. Williams, 2016: Impact of anthropogenic climate change on wildfire across western US forests. *Proceedings of the National Academy of Sciences*, 113 (42), 11770–11775.

Abatzoglou, J. T., A. P. Williams and R. Barbero, 2019: Global emergence of anthropogenic climate change in fire weather indices. *Geophysical Research Letters*, 46 (1), 326–336.

Abatzoglou, J. T. et al., 2018: Global patterns of interannual climate-fire relationships. *Global Change Biology*, 24 (11), 5164-5175, doi:10.1111/gcb.14405.

Abell, R., J. Allan and B. Lehner, 2007: Unlocking the potential of protected areas for freshwaters. *Biological Conservation*, 134 (1), 48-63.

Abell, R., B. Lehner, M. Thieme and S. Linke, 2017: Looking Beyond the Fenceline: Assessing Protection Gaps for the World's Rivers. *Conservation Letters*, 10 (4), 384-394, doi:10.1111/conl.12312.

Abelson, A. et al., 2015: Upgrading marine ecosystem restoration using ecological-social concepts. *BioScience*, 66 (2), 156-163, doi:10.1093/biosci/biv171.

Aben, R. C. H. et al., 2017: Cross continental increase in methane ebullition under climate change. *Nature Communications*, 8, doi:10.1038/s41467-017-01535-y.

Abreu, R. C. R. et al., 2017: The biodiversity cost of carbon sequestration in tropical savanna. *Science Advances*, 3 (8), e1701284.

Ackerman, D., D. Griffin, S. E. Hobbie and J. C. Finlay, 2017: Arctic shrub growth trajectories differ across soil moisture levels. *Global Change Biology*, 23 (10), 4294-4302.

Adams, H. D. et al., 2017: A multi-species synthesis of physiological mechanisms in drought-induced tree mortality. *Nature Ecology & Evolution*, 1 (9), 1285-1291, doi:10.1038/s41559-017-0248-x.

Addis, C. E. and M. S. Bret-Harte, 2019: The importance of secondary growth to plant responses to snow in the arctic. *Functional Ecology*, 33 (6), 1050-1066.

Adrian, R. et al., 2012: Windows of change: temporal scale of analysis is decisive to detect ecosystem responses to climate change. *Marine Biology*, 159 (11), 2533-2542, doi:10.1007/s00227-012-1938-1.

Adrian, R. and D. O. Hessen, 2016: Environmental Impacts—Lake Ecosystems. In: North Sea Region Climate Change Assessment [Quante, M. and F. Colijn (eds.)]. Springer, Springer International Publishing AG Switzerland.

Adrian, R. et al., 2009: Lakes as sentinels of climate change. *Limnology and Oceanography*, 54 (6), 2283-2297, doi:10.4319/lo.2009.54.6_part_2.2283.

AghaKouchak, A., L. Y. Cheng, O. Mazdiyasni and A. Farahmand, 2014: Global warming and changes in risk of concurrent climate extremes: Insights from the 2014 California drought. *Geophysical Research Letters*, 41 (24), 8847-8852, doi:10.1002/2014gl062308.

Aguirre-Gutiérrez, J. et al., 2019: Drier tropical forests are susceptible to functional changes in response to a long-term drought. *Ecol Lett*, 22 (5), 855-865, doi:10.1111/ele.13243.

Ahlström, A., P. A. Miller and B. Smith, 2012: Too early to infer a global NPP decline since 2000. *Geophysical Research Letters*, in press.

Ahlstrom, A. et al., 2015: Importance of vegetation dynamics for future terrestrial carbon cycling. *Environmental Research Letters*, 10 (5), doi:054019 10.1088/1748-9326/10/5/054019.

Aide, T. M. et al., 2019: Woody vegetation dynamics in the tropical and subtropical Andes from 2001 to 2014: Satellite image interpretation and expert validation. *Global Change Biology*, 25 (6), 2112-2126.

Ainsworth, E. A. et al., 2012: The Effects of Tropospheric Ozone on Net Primary Productivity and Implications for Climate Change. In: Annual Review of Plant Biology, Vol 63 [Merchant, S. S. (ed.)], 63, 637-661.

Alamgir, M., P. L. Pert and S. M. Turton, 2014: A review of ecosystem services research in Australia reveals a gap in integrating climate change and impacts on ecosystem services. *International Journal of Biodiversity Science, Ecosystem Services & Management*, 10 (2), 112-127.

Aleixo, I. et al., 2019: Amazonian rainforest tree mortality driven by climate and functional traits. *Nature Climate Change*, 9 (5), 384-+, doi:10.1038/s41558-019-0458-0.

Aleman, J. C. et al., 2017: Tree cover in Central Africa: determinants and sensitivity under contrasted scenarios of global change. *Scientific Reports*, 7, doi:41393 10.1038/srep41393.

Aleman, J. C., O. Blarquez and C. A. Staver, 2016: Land-use change outweighs projected effects of changing rainfall on tree cover in sub-Saharan Africa. *Global Change Biology*, 22 (9), 3013-3025, doi:10.1111/gcb.13299.

Aleman, J. C., M. A. Jarzyna and A. C. Staver, 2018: Forest extent and deforestation in tropical Africa since 1900. *Nature Ecology & Evolution*, 2 (1), 26.

Alencar, A. A., P. M. Brando, G. P. Asner and F. E. Putz, 2015: Landscape fragmentation, severe drought, and the new Amazon forest fire regime. *Ecological Applications*, 25 (6), 1493-1505, doi:10.1890/14-1528.1.

Alkama, R. and A. Cescatti, 2016: Biophysical climate impacts of recent changes in global forest cover. *Science*, 351 (6273), 600-604, doi:10.1126/science.aac8083.

Alkemade, R. et al., 2013: Assessing the impacts of livestock production on biodiversity in rangeland ecosystems. *Proceedings of the National Academy of Sciences of the United States of America*, 110 (52), 20900-20905, doi:10.1073/pnas.1011013108.

Allen, C. D., D. D. Breshears and N. G. McDowell, 2015: On underestimation of global vulnerability to tree mortality and forest die-off from hotter drought in the Anthropocene. *Ecosphere*, 6 (8), 1-55.

Allen, C. D. et al., 2010: A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecology and Management*, 259 (4), 660-684, doi:10.1016/j.foreco.2009.09.001.

Allen, J. L. and J. C. Lendemer, 2016: Climate change impacts on endemic, high-elevation lichens in a biodiversity hotspot. *Biodiversity and Conservation*, 25 (3), 555-568, doi:10.1007/s10531-016-1071-4.

Alo, C. A. and G. Wang, 2008: Potential future changes of the terrestrial ecosystem based on climate projections by eight general circulation models. *Journal of Geophysical Research: Biogeosciences*, 113 (G1).

Alvarez-Garreton, C., A. Lara, J. P. Boisier and M. Galleguillos, 2019: The Impacts of Native Forests and Forest Plantation on Water Supply in Chile. *Forests*, 10 (6), 473.

Ameztegui, A., L. Coll, L. Brotons and J. M. Ninot, 2016: Land-use legacies rather than climate change are driving the recent upward shift of the mountain tree line in the Pyrenees. *Global Ecology and Biogeography*, 25 (3), 263-273, doi:10.1111/geb.12407.

Anadon, J. D., O. E. Sala and F. T. Maestre, 2014: Climate change will increase savannas at the expense of forests and treeless vegetation in tropical and subtropical Americas. *Journal of Ecology*, 102 (6), 1363-1373, doi:10.1111/1365-2745.12325.

Andela, N. et al., 2017: A human-driven decline in global burned area. *Science*, 356 (6345), 1356-1361, doi:10.1126/science.aal4108.

Andela, N. et al., 2019: The Global Fire Atlas of individual fire size, duration, speed and direction. *Earth System Science Data*, 11 (2), 529-552, doi:10.5194/essd-11-529-2019.

Anderegg, W. R. L. et al., 2015: Tree mortality from drought, insects, and their interactions in a changing climate. *New Phytologist*, 208 (3), 674-683, doi:10.1111/nph.13477.

Anderegg, W. R. L. et al., 2016: Meta-analysis reveals that hydraulic traits explain cross-species patterns of drought-induced tree mortality across the globe. *Proceedings of the National Academy of Sciences of the United States of America*, 113 (18), 5024-5029, doi:10.1073/pnas.1525678113.

Anderegg, W. R. L. et al., 2018: Hydraulic diversity of forests regulates ecosystem resilience during drought. *Nature*, 561 (7724), 538-+, doi:10.1038/s41586-018-0539-7.

Anderson, L. O. et al., 2018: Vulnerability of Amazonian forests to repeated droughts. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, 373 (1760), doi:10.1098/rstb.2017.0411.

Anich, N. M. and M. P. Ward, 2017: Using audio playback to expand the geographic breeding range of an endangered species. *Diversity and Distributions*, 23 (12), 1499-1508, doi:10.1111/ddi.12635.

Anja, J., 2014: Extending the climate envelope: Methodological approaches to integrate ecological prerequisites in species distribution models at large spatial extents.

Arafeh-Dalmau, N. et al., 2019: Extreme Marine Heatwaves Alter Kelp Forest Community Near Its Equatorward Distribution Limit. *Frontiers in Marine Science*, 6, doi:10.3389/fmars.2019.00499.

Aragao, L. E. O. C. et al., 2018: 21st Century drought-related fires counteract the decline of Amazon deforestation carbon emissions. *Nature Communications*, 9, doi:10.1038/s41467-017-02771-y.

Aram, F., E. H. García, E. Solgi and S. Mansournia, 2019: Urban green space cooling effect in cities. *Heliyon*, 5 (4), e01339.

Araujo, M. et al., 2019: Standards for distribution models in biodiversity assessments. *Science Advances*, 5 (1), doi:10.1126/sciadv.aat4858.

Archer, S. R. et al., 2017: Woody plant encroachment: causes and consequences. In: Rangeland systems. Springer, Cham, 25–84.

Archibald, S., 2016: Managing the human component of fire regimes: lessons from Africa. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371 (1696), 20150346.

Archibald, S., C. E. R. Lehmann, J. L. Gómez-Dans and R. A. Bradstock, 2013: Defining pyromes and global syndromes of fire regimes. *Proceedings of the National Academy of Sciences*, 110 (16), 6442-6447.

Arneth, A. et al., 2019: *Chapter 1: Framing and Context*. Special Report Climate Change and Land, IPCC, in press.

Arora, V. K. and J. R. Melton, 2018: Reduction in global area burned and wildfire emissions since 1930s enhances carbon uptake by land. *Nature Communications*, 9, doi:10.1038/s41467-018-03838-0.

Arora, V. K. et al., 2016: Potential near-future carbon uptake overcomes losses from a large insect outbreak in British Columbia, Canada. *Geophysical Research Letters*, 43 (6), 2590-2598, doi:10.1002/2015gl067532.

Asner, G. P. et al., 2016a: Progressive forest canopy water loss during the 2012–2015 California drought. *Proceedings of the National Academy of Sciences*, 113 (2), E249–E255.

Asner, G. P., N. Vaughn, I. P. Smit and S. Levick, 2016b: Ecosystem-scale effects of megafauna in African savannas. *Ecography*, 39 (2), 240–252.

Assouma, M. H. et al., 2019: Contrasted seasonal balances in a Sahelian pastoral ecosystem result in a neutral annual carbon balance. *Journal of Arid Environments*, 162, 62-73, doi:10.1016/j.jaridenv.2018.11.013.

Avissar, R. and D. Werth, 2005: Global hydroclimatological teleconnections resulting from tropical deforestation. *Journal of Hydrometeorology*, 6 (2), 134-145, doi:10.1175/jhm406.1.

Avitabile, V. et al., 2016: An integrated pan-tropical biomass map using multiple reference datasets. *Global Change Biology*, 22 (4), 1406-1420, doi:10.1111/gcb.13139.

Baccini, A. et al., 2012: Estimated carbon dioxide emissions from tropical deforestation improved by carbon-density maps. *Nature Climate Change*, 2 (3), 182-185, doi:10.1038/nclimate1354.

Baccini, A. et al., 2017: Tropical forests are a net carbon source based on aboveground measurements of gain and loss. *Science*, 358 (6360), 230-234.

Badgley, G., L. D. L. Anderegg, J. A. Berry and C. B. Field, 2019: Terrestrial gross primary production: Using NIRV to scale from site to globe. *Global Change Biology*, doi:10.1111/gcb.14729.

Badiou, P., B. Page and L. Ross, 2019: A comparison of water quality and greenhouse gas emissions in constructed wetlands and conventional retention basins with and without submerged macrophyte management for storm water regulation. *Ecological Engineering*, 127, 292-301, doi:https://doi.org/10.1016/j.ecoleng.2018.11.028.

Bailey, L. and M. van de Pol, 2016: Tackling extremes: challenges for ecological and evolutionary research on extreme climatic events. *Journal of Animal Ecology*, 85 (1), 85-96, doi:10.1111/1365-2656.12451.

Baillie, J. E. M. and E. R. Butcher, 2012: *Priceless or Worthless? The world's most threatened species*. Zoological Society of London, IUCN, SSC and ZSL, London, 123 [Available at: https://portals.iucn.org/library/sites/library/files/documents/2012-096.pdf].

Bakker, E. S. et al., 2016: Combining paleo-data and modern exclosure experiments to assess the impact of megafauna extinctions on woody vegetation. *Proceedings of the National Academy of Sciences*, 113 (4), 847-855.

Bakker, E. S. and J.-C. Svenning, 2018: *Trophic rewilding: impact on ecosystems under global change*. The Royal Society.

Balanya, J. et al., 2006: Global genetic change tracks global climate warming in Drosophila subobscura. *Science*, 313 (5794), 1773-1775, doi:10.1126/science.1131002.

Balch, J. K. et al., 2015: The susceptibility of southeastern Amazon forests to fire: insights from a large-scale burn experiment. *BioScience*, 65 (9), 893-905.

Baldocchi, D. and J. Penuelas, 2019: The physics and ecology of mining carbon dioxide from the atmosphere by ecosystems. *Global Change Biology*.

Balian, E., H. Segers, C. Leveque and K. Martens, 2008: An introduction to the freshwater animal diversity assessment (FADA) project. *Hydrobiologia*, 595, 3-8, doi:10.1007/s10750-007-9235-6.

Ballantyne, A. P. et al., 2012: Increase in observed net carbon dioxide uptake by land and oceans during the past 50 years. *Nature*, 488 (7409), 70.

Baraer, M., B. G. Mark and J. M. McKenzie, 2011: Past peak water in Peru's Cordillera Blanca: diagnosing the demise of glacier influence on stream discharge. In: *Fall meeting, AGU*, 5-9 December, San Francisco, CA.

Barbero, R. et al., 2015: Climate change presents increased potential for very large fires in the contiguous United States. *International Journal of Wildland Fire*, 24 (7), 892–899.

Barichivich, J. et al., 2013: Large-scale variations in the vegetation growing season and annual cycle of atmospheric CO2 at high northern latitudes from 1950 to 2011. *Global Change Biology*, 19 (10), 3167-3183, doi:10.1111/gcb.12283.

Barnosky, A. et al., 2017: Merging paleobiology with conservation biology to guide the future of terrestrial ecosystems. *Science*, 355 (6325), doi:10.1126/science.aah4787.

Barrows, C. W. and M. L. Murphy-Mariscal, 2012: Modeling impacts of climate change on Joshua trees at their southern boundary: How scale impacts predictions. *Biological Conservation*, 152, 29-36, doi:10.1016/j.biocon.2012.03.028.

Barry, J. P., C. H. Baxter, R. D. Sagarin and S. E. Gilman, 1995: Climate-Related, Long-Term Faunal Changes in a California Rocky Intertidal Community. *Science*, 267 (5198), 672-675, doi:10.1126/science.267.5198.672.

Bartlett, M. K., M. Detto and S. W. Pacala, 2019: Predicting shifts in the functional composition of tropical forests under increased drought and CO. *Ecol Lett*, 22 (1), 67-77, doi:10.1111/ele.13168.

Barton, A. M. and H. M. Poulos, 2018: Pine vs. oaks revisited: Conversion of Madrean pine-oak forest to oak shrubland after high-severity wildfire in the Sky Islands of Arizona. *Forest Ecology and Management*, 414, 28-40, doi:10.1016/j.foreco.2018.02.011.

Bartosiewicz, M. et al., 2019: Hot tops, cold bottoms: Synergistic climate warming and shielding effects increase carbon burial in lakes. *Limnology and Oceanography Letters*, 0 (0), doi:10.1002/lol2.10117.

Basara, J. B. and J. I. Christian, 2018: Seasonal and interannual variability of land-atmosphere coupling across the Southern Great Plains of North America using the North American regional reanalysis. *International Journal of Climatology*, 38 (2), 964-978, doi:10.1002/joc.5223.

Bastin, J.-F. et al., 2019: The global tree restoration potential. *Science*, 365 (6448), 76-79, doi:10.1126/science.aax0848.

Bastviken, D., J. Cole, M. Pace and L. Tranvik, 2004: Methane emissions from lakes: Dependence of lake characteristics, two regional assessments, and a global estimate. *Global Biogeochemical Cycles*, 18 (4), doi:10.1029/2004gb002238.

Batáry, P., L. V. Dicks, D. Kleijn and W. J. Sutherland, 2015: The role of agri-environment schemes in conservation and environmental management. *Conservation Biology*, 29 (4), 1006-1016.

Batjes, N. H., 2016: Harmonized soil property values for broad-scale modelling (WISE30sec) with estimates of global soil carbon stocks. *Geoderma*, 269, 61-68, doi:10.1016/j.geoderma.2016.01.034.

Batllori, E. et al., 2019: Compound fire-drought regimes promote ecosystem transitions in Mediterranean ecosystems. *Journal of Ecology*, 107 (3), 1187-1198, doi:10.1111/1365-2745.13115.

Batllori, E. et al., 2017: Potential relocation of climatic environments suggests high rates of climate displacement within the North American protection network. *Global Change Biology*, 23 (8), 3219-3230, doi:10.1111/gcb.13663.

Battin, T. J. et al., 2009: Biophysical controls on organic carbon fluxes in fluvial networks (vol 1, pg 95, 2008). *Nature Geoscience*, 2 (8), 595-595, doi:10.1038/ngeo602.

Baudena, M. et al., 2015: Forests, savannas, and grasslands: bridging the knowledge gap between ecology and Dynamic Global Vegetation Models. *Biogeosciences*, 12 (6), 1833-1848, doi:10.5194/bg-12-1833-2015.

Bay, R. A. et al., 2018: Genomic signals of selection predict climate-driven population declines in a migratory bird. *Science*, 359 (6371), 83-86, doi:10.1126/science.aan4380.

Beaugrand, G. et al., 2019: Prediction of unprecedented biological shifts in the global ocean. *Nature Climate Change*, 9 (3), 237-+, doi:10.1038/s41558-019-0420-1.

Beaulieu, J. J., T. DelSontro and J. A. Downing, 2019: Eutrophication will increase methane emissions from lakes and impoundments during the 21st century. *Nature Communications*, 10, doi:10.1038/s41467-019-09100-5.

Beaumont, L. et al., 2016: Which species distribution models are more (or less) likely to project broad-scale, climate-induced shifts in species ranges? *Ecological Modelling*, 342, 135-146, doi:10.1016/j.ecolmodel.2016.10.004.

Bebbington, A. J. and J. T. Bury, 2009: Institutional challenges for mining and sustainability in Peru. *Proceedings of the National Academy of Sciences*, 106 (41), 17296-17301, doi:10.1073/pnas.0906057106.

Beer, C. et al., 2010: Terrestrial Gross Carbon Dioxide Uptake: Global Distribution and Covariation with Climate. *Science*, 329 (5993), 834-838, doi:10.1126/science.1184984.

Bell, G. et al., 2019: Trophic structure modulates community rescue following acidification. *Proceedings of the Royal Society B-Biological Sciences*, 286 (1904), 9, doi:10.1098/rspb.2019.0856.

Bell, G. and A. Gonzalez, 2009: Evolutionary rescue can prevent extinction following environmental change. *Ecology Letters*, 12 (9), 942-948, doi:10.1111/j.1461-0248.2009.01350.x.

Bellahirech, A. et al., 2019: Site- and tree-related factors affecting colonization of cork oaks Quercus suber L. by ambrosia beetles in Tunisia. *Annals of Forest Science*, 76 (2), doi:10.1007/s13595-019-0815-1.

Bellard, C. et al., 2012: Impacts of climate change on the future of biodiversity. *Ecology Letters*, 15 (4), 365-377, doi:10.1111/j.1461-0248.2011.01736.x.

Bellard, C. et al., 2014: Vulnerability of biodiversity hotspots to global change. *Global Ecology and Biogeography*, 23 (12), 1376-1386, doi:10.1111/geb.12228.

Bellis, J., D. Bourke, C. Williams and S. Dalrymple, 2019: Identifying factors associated with the success and failure of terrestrial insect translocations. *Biological Conservation*, 236, 29-36, doi:10.1016/j.biocon.2019.05.008.

Bello, C. et al., 2015: Defaunation affects carbon storage in tropical forests. *Science Advances*, 1 (11), doi:10.1126/sciadv.1501105.

Benali, A. et al., 2017: Bimodal fire regimes unveil a global-scale anthropogenic fingerprint. *Global Ecology and Biogeography*, 26 (7), 799-811, doi:10.1111/geb.12586.

Bennett, A. C., N. G. McDowell, C. D. Allen and K. J. Anderson-Teixeira, 2015: Larger trees suffer most during drought in forests worldwide. *Nature Plants*, 1 (10), 15139.

Bentz, B. J. et al., 2010: Climate Change and Bark Beetles of the Western United States and Canada: Direct and Indirect Effects. *BioScience*, 60 (8), 602-613, doi:10.1525/bio.2010.60.8.6.

Bergengren, J. C., D. E. Waliser and Y. L. Yung, 2011: Ecological sensitivity: a biospheric view of climate change. *Climatic Change*, 107 (3-4), 433.

Berner, L. T., B. E. Law, A. J. H. Meddens and J. A. Hicke, 2017: Tree mortality from fires, bark beetles, and timber harvest during a hot and dry decade in the western United States (2003-2012). *Environmental Research Letters*, 12 (6), doi:10.1088/1748-9326/aa6f94.

Bernhardt, E. et al., 2018: The metabolic regimes of flowing waters. *Limnology and Oceanography*, 63, S99-S118, doi:10.1002/lno.10726.

Berteaux, D. et al., 2018: Northern protected areas will become important refuges for biodiversity tracking suitable climates. *Scientific Reports*, 8, doi:10.1038/s41598-018-23050-w.

Berzaghi, F. et al., 2018: Assessing the role of megafauna in tropical forest ecosystems and biogeochemical cycles - the potential of vegetation models. *Ecography*, 41 (12), 1934-1954, doi:10.1111/ecog.03309.

Betts, R. A. et al., 2015: Climate and land use change impacts on global terrestrial ecosystems and river flows in the HadGEM2-ES Earth system model using the representative concentration pathways. *Biogeosciences*, 12, 1317-1338, doi:10.5194/bg-12-1317-2015.

Bhatt, U. S. et al., 2014: Implications of Arctic sea ice decline for the Earth system. *Annual Review of Environment and Resources*, 39, 57-89.

Bird, J. P. et al., 2012: Integrating spatially explicit habitat projections into extinction risk assessments: a reassessment of Amazonian avifauna incorporating projected deforestation. *Diversity and Distributions*, 18 (3), 273-281, doi:10.1111/j.1472-4642.2011.00843.x.

Biskaborn, B. K. et al., 2019: Permafrost is warming at a global scale. *Nature Communications*, 10, doi:10.1038/s41467-018-08240-4.

Bjorkman, A. D. et al., 2019: Status and trends in Arctic vegetation: Evidence from experimental warming and long-term monitoring. *AMBIO*, 1-15.

Bjorkman, A. D. et al., 2018: Plant functional trait change across a warming tundra biome. *Nature*, 562 (7725), 57.

Blair, J., J. Nippert and J. Briggs, 2014: Grassland ecology. *Ecology and the Environment*, 389-423.

Blake, S. et al., 2009: Forest elephants: tree planters of the Congo. *Biotropica*, 41 (4), 459-468.

Blaser, W. J. et al., 2018: Climate-smart sustainable agriculture in low-to-intermediate shade agroforests. *Nature Sustainability*, 1 (5), 234.

Blume-Werry, G., S. D. Wilson, J. Kreyling and A. Milbau, 2016: The hidden season: growing season is 50% longer below than above ground along an arctic elevation gradient. *New Phytologist*, 209 (3), 978-986.

Boelman, N. T. et al., 2015: Greater shrub dominance alters breeding habitat and food resources for migratory songbirds in Alaskan arctic tundra. *Global Change Biology*, 21 (4), 1508-1520.

Boisvert-Marsh, L., C. Perie and S. de Blois, 2019: Divergent responses to climate change and disturbance drive recruitment patterns underlying latitudinal shifts of tree species. *Journal of Ecology*, 107 (4), 1956-1969, doi:10.1111/1365-2745.13149.

Boit, A. et al., 2016: Large-scale impact of climate change vs. land-use change on future biome shifts in Latin America. *Global Change Biology*, 22 (11), 3689-3701, doi:10.1111/gcb.13355.

Bolton, D. K. et al., 2018: Evidence of vegetation greening at alpine treeline ecotones: three decades of Landsat spectral trends informed by lidar-derived vertical structure. *Environmental Research Letters*, 13 (8), doi:10.1088/1748-9326/aad5d2.

Bonai, D. et al., 2016: The response of tropical rainforests to drought-lessons from recent research and future prospects. *Annals of Forest Science*, 73, 27-44, doi:10.1007/s13595-015-0522-5.

Bonal, D. et al., 2016: The response of tropical rainforests to drought—lessons from recent research and future prospects. *Annals of Forest Science*, 73 (1), 27-44.

Bonan, G. B. and S. C. Doney, 2018: Climate, ecosystems, and planetary futures: The challenge to predict life in Earth system models. *Science*, 359 (6375), 533-+, doi:10.1126/science.aam8328.

Bond, W. and N. P. Zaloumis, 2016: The deforestation story: testing for anthropogenic origins of Africa's flammable grassy biomes. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371 (1696), 20150170.

Bond, W. J., N. Stevens, G. F. Midgley and C. E. R. Lehmann, 2019: The Trouble with Trees: Afforestation Plans for Africa. *Trends in Ecology & Evolution*.

Bond, W. J., F. I. Woodward and G. F. Midgley, 2005: The global distribution of ecosystems in a world without fire. *New Phytologist*, 165 (2), 525-538.

Boulton, C. A., B. B. B. Booth and P. Good, 2017: Exploring uncertainty of Amazon dieback in a perturbed parameter Earth system ensemble. *Global Change Biology*, 23 (12), 5032-5044, doi:10.1111/gcb.13733.

Boutin, S. and J. Lane, 2014: Climate change and mammals: evolutionary versus plastic responses. *Evolutionary Applications*, 7 (1), 29-41, doi:10.1111/eva.12121.

Bowler, D. E. et al., 2017: Cross-realm assessment of climate change impacts on species' abundance trends. *Nature Ecology & Evolution*, 1 (3), 0067, doi:10.1038/s41559-016-0067.

Boysen, L., W. Lucht, D. Gerten and V. Heck, 2016: Impacts devalue the potential of large-scale terrestrial CO2 removal through biomass plantations. *Environmental Research Letters*, 11 (9), doi:10.1088/1748-9326/11/9/095010.

Bradley, B. A., C. A. Curtis and J. C. Chambers, 2016: Bromus response to climate and projected changes with climate change. In: Exotic brome-grasses in arid and semiarid ecosystems of the western US. Springer, 257–274.

Bradley, R. S., M. Vuille, H. F. Diaz and W. Vergara, 2006: Threats to water supplies in the tropical Andes. *Science*, 312 (5781), 1755-1756.

Bradshaw, W. E. and C. M. Holzapfel, 2001: Genetic shift in photoperiodic response correlated with global warming. *Proceedings of the National Academy of Sciences of the United States of America*, 98 (25), 14509-14511, doi:10.1073/pnas.241391498.

Bramer, I. et al., 2018: Chapter Three - Advances in Monitoring and Modelling Climate at Ecologically Relevant Scales. In: Advances in Ecological Research [Bohan, D. A., A. J. Dumbrell, G. Woodward and M. Jackson (eds.)]. Academic Press, 58, 101-161.

Brando, P. M. et al., 2014: Abrupt increases in Amazonian tree mortality due to drought–fire interactions. *Proceedings of the National Academy of Sciences*, 111 (17), 6347-6352.

Brandt, M. et al., 2019: Changes in rainfall distribution promote woody foliage production in the Sahel. *Communications biology*, 2 (1), 133.

Brandt, M. et al., 2018: Reduction of tree cover in West African woodlands and promotion in semi-arid farmlands. *Nature Geoscience*, 11 (5), 328-+, doi:10.1038/s41561-018-0092-x.

Bremer, L. L. and K. A. Farley, 2010: Does plantation forestry restore biodiversity or create green deserts? A synthesis of the effects of land-use transitions on plant species richness. *Biodiversity and Conservation*, 19 (14), 3893-3915.

Bridle, J. R., J. Buckley, E. J. Bodsworth and C. D. Thomas, 2014: Evolution on the move: specialization on widespread resources associated with rapid range expansion in response to climate change. *Proceedings of the Royal Society B-Biological Sciences*, 281 (1776), 7, doi:10.1098/rspb.2013.1800.

Brienen, R. J. W. et al., 2015: Long-term decline of the Amazon carbon sink. *Nature*, 519 (7543), 344.

Bright, B. C., J. A. Hicke and A. J. H. Meddens, 2013: Effects of bark beetle-caused tree mortality on biogeochemical and biogeophysical MODIS products. *Journal of Geophysical Research-Biogeosciences*, 118 (3), 974-982, doi:10.1002/jgrg.20078.

Broadmeadow, S. et al., 2011: The influence of riparian shade on lowland stream water temperatures in southern England and their viability for brown trout. *River Research and Applications*, 27 (2), 226-237.

Brodie, J. F., 2016: Synergistic effects of climate change and agricultural land use on mammals. *Frontiers in Ecology and the Environment*, 14 (1), 20-26, doi:10.1002/16-0110.1.

Brooker, R. W. et al., 2018: Tiny niches and translocations: The challenge of identifying suitable recipient sites for small and immobile species. *Journal of Applied Ecology*, 55 (2), 621-630.

Brooks, A. J., B. Wolfenden, B. J. Downes and J. Lancaster, 2018: Barriers to dispersal: The effect of a weir on stream insect drift. *River Research and Applications*, 34 (10), 1244-1253.

Brookshire, E. N. J. and T. Weaver, 2015: Long-term decline in grassland productivity driven by increasing dryness. *Nature Communications*, 6, 7148.

Brouwers, N. C. and N. C. Coops, 2016: Decreasing Net Primary Production in forest and shrub vegetation across southwest Australia. *Ecological Indicators*, 66, 10-19, doi:10.1016/j.ecolind.2016.01.010.

Brovkin, V. et al., 2013: Effect of Anthropogenic Land-Use and Land-Cover Changes on Climate and Land Carbon Storage in CMIP5 Projections for the Twenty-First Century. *Journal of Climate*, 26 (18), 6859-6881, doi:10.1175/jcli-d-12-00623.1.

Brown, D. R. N. et al., 2015: Interactive effects of wildfire and climate on permafrost degradation in Alaskan lowland forests. *Journal of Geophysical Research-Biogeosciences*, 120 (8), 1619-1637, doi:10.1002/2015jg003033.

Bruijnzeel, L. A., 2004: Hydrological functions of tropical forests: not seeing the soil for the trees? *Agriculture, Ecosystems & Environment*, 104 (1), 185-228.

Buckeridge, K. M., E. Zufelt, H. Y. Chu and P. Grogan, 2010: Soil nitrogen cycling rates in low arctic shrub tundra are enhanced by litter feedbacks. *Plant and Soil*, 330 (1-2), 407-421, doi:10.1007/s11104-009-0214-8.

Buisson, L. et al., 2010: Uncertainty in ensemble forecasting of species distribution. *Global Change Biology*, 16 (4), 1145-1157, doi:10.1111/j.1365-2486.2009.02000.x.

Buitenwerf, R., W. J. Bond, N. Stevens and W. S. W. Trollope, 2012: Increased tree densities in South African savannas: > 50 years of data suggests CO2 as a driver. *Global Change Biology*, 18 (2), 675-684, doi:10.1111/j.1365-2486.2011.02561.x.

Buma, B. et al., 2017: Emerging climate-driven disturbance processes: widespread mortality associated with snow-to-rain transitions across 10 degrees of latitude and half the range of a climate-threatened conifer. *Global Change Biology*, 23 (7), 2903-2914, doi:10.1111/gcb.13555.

Bunn, S. E. and A. H. Arthington, 2002: Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. *Environmental Management*, 30 (4), 492-507, doi:10.1007/s00267-002-2737-0.

Buotte, P. C. et al., 2017: Recent and future climate suitability for whitebark pine mortality from mountain pine beetles varies across the western US. *Forest Ecology and Management*, 399, 132–142.

Buotte, P. C. et al., 2019: Near-future forest vulnerability to drought and fire varies across the western United States. *Global Change Biology*, 25 (1), 290–303.

Burch, S., P. Berry and M. Sanders, 2014: Embedding climate change adaptation in biodiversity conservation: A case study of England. *Environmental Science & Policy*, 37, 79-90.

Burden, A., A. Garbutt and C. Evans, 2019: Effect of restoration on saltmarsh carbon accumulation in Eastern England. *Biology Letters*, 15 (1), 20180773.

Burton, C., R. A. Betts, C. D. Jones and K. Williams, 2018: Will Fire Danger Be Reduced by Using Solar Radiation Management to Limit Global Warming to 1.5 degrees C Compared to 2.0 degrees C? *Geophysical Research Letters*, 45 (8), 3644-3652, doi:10.1002/2018gl077848.

Bury, J. et al., 2013: New geographies of water and climate change in Peru: Coupled natural and social transformations in the Santa River watershed. *Annals of the Association of American Geographers*, 103 (2), 363-374.

Bush, A. A. et al., 2014: Continental-scale assessment of risk to the Australian Odonata from climate change. *PLOS ONE*, 9 (2), e88958.

Bush, M. B. et al., 2015: Fire and climate: contrasting pressures on tropical Andean timberline species. *Journal of Biogeography*, 42 (5), 938-950, doi:10.1111/jbi.12470.

Bustamante, M. M. C. et al., 2016: Toward an integrated monitoring framework to assess the effects of tropical forest degradation and recovery on carbon stocks and biodiversity. *Global Change Biology*, 22 (1), 92-109, doi:10.1111/gcb.13087.

Buytaert, W. et al., 2017: Glacial melt content of water use in the tropical Andes. *Environmental Research Letters*, 12 (11), 8, doi:10.1088/1748-9326/aa926c.

Calef, M. P. et al., 2015: Recent Changes in Annual Area Burned in Interior Alaska: The Impact of Fire Management. *Earth Interactions*, 19, 1-17, doi:10.1175/ei-d-14-0025.1.

Caon, L., V. R. Vallejo, C. J. Ritsema and V. Geissen, 2014: Effects of wildfire on soil nutrients in Mediterranean ecosystems. *Earth-Science Reviews*, 139, 47-58, doi:10.1016/j.earscirev.2014.09.001.

Caracciolo, D., E. Istanbulluoglu, L. V. Noto and S. L. Collins, 2016: Mechanisms of shrub encroachment into Northern Chihuahuan Desert grasslands and impacts of climate change investigated using a cellular automata model. *Advances in Water Resources*, 91, 46–62.

Carilla, J. et al., 2018: Vegetation trends over eleven years on mountain summits in NW Argentina. *Ecology and Evolution*, 8 (23), 11554-11567.

Carroll, C., J. J. Lawler, D. R. Roberts and A. Hamann, 2015: Biotic and Climatic Velocity Identify Contrasting Areas of Vulnerability to Climate Change. *PLOS ONE*, 10 (10), doi:10.1371/journal.pone.0140486.

Case, M. F. et al., 2019: Severe drought limits trees in a semi-arid savanna. *Ecology*, doi:10.1002/ecy.2842.

Castaneda, L. E. et al., 2019: Evolutionary potential of thermal preference and heat tolerance in Drosophila subobscura. *Journal of Evolutionary Biology*, 32 (8), 818-824, doi:10.1111/jeb.13483.

Cavanaugh, K. C. et al., 2014: Carbon storage in tropical forests correlates with taxonomic diversity and functional dominance on a global scale. *Global Ecology and Biogeography*, 23 (5), 563-573, doi:10.1111/geb.12143.

Cernusak, L. A. et al., 2013: Tropical forest responses to increasing atmospheric CO2: current knowledge and opportunities for future research. *Functional Plant Biology*, 40 (6), 531-551.

Chadburn, S. E. et al., 2017: An observation-based constraint on permafrost loss as a function of global warming. *Nature Climate Change*, 7 (5), 340-+, doi:10.1038/nclimate3262.

Chae, Y. et al., 2015: Arctic greening can cause earlier seasonality of Arctic amplification. *Geophysical Research Letters*, 42 (2), 536-541, doi:10.1002/2014gl061841.

Chakraborty, A., P. K. Joshi, A. Ghosh and G. Areendran, 2013: Assessing biome boundary shifts under climate change scenarios in India. *Ecological Indicators*, 34, 536-547, doi:10.1016/j.ecolind.2013.06.013.

Chamaille-Jammes, S., M. Massot, P. Aragon and J. Clobert, 2006: Global warming and positive fitness response in mountain populations of common lizards Lacerta vivipara. *Global Change Biology*, 12 (2), 392-402, doi:10.1111/j.1365-2486.2005.01088.x.

Chambers, J. C. et al., 2014: Resilience to stress and disturbance, and resistance to Bromus tectorum L. invasion in cold desert shrublands of western North America. *Ecosystems*, 17 (2), 360–375.

Chapin, F. S., 3rd et al., 2000: Consequences of changing biodiversity. *Nature*, 405 (6783), 234-42, doi:10.1038/35012241.

Chaplin-Kramer, R. et al., 2015: Degradation in carbon stocks near tropical forest edges. *Nature Communications*, 6, doi:10.1038/ncomms10158.

Charles-Dominique, T. et al., 2016: Spiny plants, mammal browsers, and the origin of African savannas. *Proceedings of the National Academy of Sciences*, 113 (38), E5572-E5579.

Charney, J., P. H. Stone and W. J. Quirk, 1975: DROUGHT IN SAHARA - BIOGEOPHYSICAL FEEDBACK MECHANISM. *Science*, 187 (4175), 434-435, doi:10.1126/science.187.4175.434.

Chen, D. et al., 2018: Strong cooling induced by stand-replacing fires through albedo in Siberian larch forests. *Scientific Reports*, 8, doi:10.1038/s41598-018-23253-1.

Chen, G., D. J. Hayes and A. D. McGuire, 2017: Contributions of wildland fire to terrestrial ecosystem carbon dynamics in North America from 1990 to 2012. *Global Biogeochemical Cycles*, 31 (5), 878–900.

Chen, I.-C. et al., 2011: Rapid Range Shifts of Species Associated with High Levels of Climate Warming. *Science*, 333 (6045), 1024-1026, doi:10.1126/science.1206432.

Chenoweth, J. et al., 2011: Impact of climate change on the water resources of the eastern Mediterranean and Middle East region: Modeled 21st century changes and implications. *Water Resources Research*, 47, doi:10.1029/2010wr010269.

Chessman, B. C., 2009: Climatic changes and 13-year trends in stream macroinvertebrate assemblages in New South Wales, Australia. *Global Change Biology*, 15 (11), 2791-2802, doi:10.1111/j.1365-2486.2008.01840.x.

Chessman, B. C., 2015: Relationships between lotic macroinvertebrate traits and responses to extreme drought. *Freshwater Biology*, 60 (1), 50-63, doi:10.1111/fwb.12466.

Chin, A., P. M. Kyne, T. I. Walker and R. B. McAULEY, 2010: An integrated risk assessment for climate change: analysing the vulnerability of sharks and rays on Australia's Great Barrier Reef. *Global Change Biology*, 16 (7), 1936-1953, doi:10.1111/j.1365-2486.2009.02128.x.

Chitale, V. and M. D. Behera, 2019: How will forest fires impact the distribution of endemic plants in the Himalayan biodiversity hotspot? *Biodiversity and Conservation*, 28 (8-9), 2259-2273, doi:10.1007/s10531-019-01733-8.

Choat, B. et al., 2018: Triggers of tree mortality under drought. *Nature*, 558 (7711), 531-539, doi:10.1038/s41586-018-0240-x.

Christmas, M. J., M. F. Breed and A. J. Lowe, 2016: Constraints to and conservation implications for climate change adaptation in plants. *Conservation Genetics*, 17 (2), 305-320.

Chuang, A. and C. R. Peterson, 2016: Expanding population edges: theories, traits, and trade-offs. *Global Change Biology*, 22 (2), 494-512, doi:10.1111/gcb.13107.

Ciais, P. et al., 2013: Carbon and Other Biogeochemical Cycles. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. [Stocker, T. F., D. Qin, G. K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P. M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom.

Ciais, P. et al., 2019: Five decades of northern land carbon uptake revealed by the interhemispheric CO2 gradient. *Nature*, 568 (7751), 221-+, doi:10.1038/s41586-019-1078-6.

Cizauskas, C. et al., 2017: Parasite vulnerability to climate change: an evidence- based functional trait approach. *Royal Society Open Science*, 4 (1), doi:10.1098/rsos.160535.

Claeys, F. et al., 2019: Climate change would lead to a sharp acceleration of Central African forests dynamics by the end of the century. *Environmental Research Letters*, 14 (4), doi:10.1088/1748-9326/aafb81.

Cline, T., V. Bennington and J. Kitchell, 2013: Climate Change Expands the Spatial Extent and Duration of Preferred Thermal Habitat for Lake Superior Fishes. *PLOS ONE*, 8 (4), doi:10.1371/journal.pone.0062279.

Coffman, J. M. et al., 2014: Restoration practices have positive effects on breeding bird species of concern in the Chihuahuan Desert. *Restoration Ecology*, 22 (3), 336-344.

Cohen, J. M. et al., 2019a: An interaction between climate change and infectious disease drove widespread amphibian declines. *Global Change Biology*, 25 (3), 927-937, doi:10.1111/gcb.14489.

Cohen, J. M., M. J. Lajeunesse and J. R. Rohr, 2018: A global synthesis of animal phenological responses to climate change. *Nature Climate Change*, 8 (3), 224, doi:10.1038/s41558-018-0067-3.

Cohen, J. M. et al., 2019b: Impacts of thermal mismatches on chytrid fungus Batrachochytrium dendrobatidis prevalence are moderated by life stage, body size, elevation and latitude. *Ecology Letters*, 22 (5), 817-825, doi:10.1111/ele.13239.

Cohn, A. S. et al., 2019: Forest loss in Brazil increases maximum temperatures within 50 km. *Environmental Research Letters*, 14 (8), doi:10.1088/1748-9326/ab31fb.

Cole, K. L. et al., 2011: Past and ongoing shifts in Joshua tree distribution support future modeled range contraction. *Ecological Applications*, 21 (1), 137-149, doi:10.1890/09-1800.1.

Collins, M. B. and E. T. A. Mitchard, 2017: A small subset of protected areas are a highly significant source of carbon emissions. *Scientific Reports*, 7, doi:10.1038/srep41902.

Collins, S. L. and Y. Xia, 2014: Long-term dynamics and hotspots of change in a desert grassland plant community. *The American Naturalist*, 185 (2), E30–E43.

Colwell, R. K. et al., 2008: Global warming, elevational range shifts, and lowland biotic attrition in the wet tropics. *Science*, 322 (5899), 258-261, doi:10.1126/science.1162547.

Comte, L., L. Buisson, M. Daufresne and G. Grenouillet, 2013: Climate-induced changes in the distribution of freshwater fish: observed and predicted trends. *Freshwater Biology*, 58 (4), 625-639, doi:10.1111/fwb.12081.

Comte, L. and J. D. Olden, 2017: Climatic vulnerability of the world's freshwater and marine fishes. *Nature Climate Change*, 7, 718, doi:10.1038/nclimate3382
https://www.nature.com/articles/nclimate3382#supplementary-information.

Comyn-Platt, E. et al., 2018: Carbon budgets for 1.5 and 2 degrees C targets lowered by natural wetland and permafrost feedbacks. *Nature Geoscience*, 11 (8), 568-+, doi:10.1038/s41561-018-0174-9.

Conti, L., A. Schmidt-Kloiber, G. Grenouillet and W. Graf, 2014: A trait-based approach to assess the vulnerability of European aquatic insects to climate change. *Hydrobiologia*, 721 (1), 297-315, doi:10.1007/s10750-013-1690-7.

Cook, B. I., E. M. Wolkovich and C. Parmesan, 2012: Divergent responses to spring and winter warming drive community level flowering trends. *Proceedings of the National Academy of Sciences*, 109 (23), 9000-9005, doi:10.1073/pnas.1118364109.

Corlett, R. T., 2016: The impacts of droughts in tropical forests. *Trends in Plant Science*, 21 (7), 584-593.

Cornelissen, J. H. C. et al., 2007: Global negative vegetation feedback to climate warming responses of leaf litter decomposition rates in cold biomes. *Ecology Letters*, 10 (7), 619-627, doi:10.1111/j.1461-0248.2007.01051.x.

Costion, C. et al., 2015: Will tropical mountaintop plant species survive climate change? Identifying key knowledge gaps using species distribution modelling in Australia. *Biological Conservation*, 191, 322-330, doi:10.1016/j.biocon.2015.07.022.

Cramb, R. A., J. F. McCarthy and NUS Press, 2015: *The oil palm complex : smallholders, agribusiness, and the state in Indonesia and Malaysia*. NUS Press, Singapore, xvi, 470 pages pp.

Cramer, W. et al., 2014: Detection and attribution of observed impacts. In: Climate Climate Change 2014: Impacts, Adaptation, and Vulnerability, . Cambridge University Press, 979-1038.

Creed, I. et al., 2018: Global change-driven effects on dissolved organic matter composition: Implications for food webs of northern lakes. *Global Change Biology*, 24 (8), 3692-3714, doi:10.1111/gcb.14129.

Creed, I. F. and M. van Noordwijk, 2018: *Forest and Water on a Changing Planet: Vulnerability, Adaptation and Governance Opportunities: A Global Assessment Report*. International Union of Forest Research Organizations (IUFRO).

Creutzig, F. et al., 2015: Bioenergy and climate change mitigation: an assessment. *Global Change Biology Bioenergy*, 7 (5), 916-944, doi:10.1111/gcbb.12205.

Cromsigt, J. P. et al., 2018: Trophic rewilding as a climate change mitigation strategy? *Philosophical Transactions of the Royal Society B: Biological Sciences*, 373 (1761), 20170440.

Cronin, D., M. Libalah, R. Bergl and G. Hearn, 2014: Biodiversity and conservation of tropical montane ecosystems in the Gulf of Guinea, West Africa. *Arctic Antarctic and Alpine Research*, 46 (4), 891-904, doi:10.1657/1938-4246-46.4.891.

Crowther, T. W. et al., 2016: Quantifying global soil carbon losses in response to warming. *Nature*, 540 (7631), 104.

Cudlin, P. et al., 2017: Drivers of treeline shift in different European mountains. *Climate Research*, 73 (1-2), 135-150, doi:10.3354/cr01465.

Cusack, D. et al., 2016: Global change effects on humid tropical forests: Evidence for biogeochemical and biodiversity shifts at an ecosystem scale. *Reviews of Geophysics*, 54 (3), 523-610, doi:10.1002/2015RG000510.

d'Amour, C. B. et al., 2017: Future urban land expansion and implications for global croplands. *Proceedings of the National Academy of Sciences*, 114 (34), 8939–8944.

da Silva, S. S. et al., 2018: Dynamics of forest fires in the southwestern Amazon. *Forest Ecology and Management*, 424, 312-322, doi:10.1016/j.foreco.2018.04.041.

Daily, G. C. and P. R. Ehrlich, 1996: Nocturnality and species survival. *Proceedings of the National Academy of Sciences of the United States of America*, 93 (21), 11709-11712, doi:10.1073/pnas.93.21.11709.

Dangles, O. et al., 2017: Ecosystem sentinels for climate change? Evidence of wetland cover changes over the last 30 years in the tropical Andes. *PLOS ONE*, 12 (5), e0175814, doi:10.1371/journal.pone.0175814.

Dargie, G. C. et al., 2017: Age, extent and carbon storage of the central Congo Basin peatland complex. *Nature*, 542 (7639), 86-+, doi:10.1038/nature21048.

Daskin, J. H., R. A. Alford and R. Puschendorf, 2011: Short-Term Exposure to Warm Microhabitats Could Explain Amphibian Persistence with Batrachochytrium dendrobatidis. *PLOS ONE*, 6 (10), doi:10.1371/journal.pone.0026215.

Daskin, J. H., M. Stalmans and R. M. Pringle, 2016: Ecological legacies of civil war: 35-year increase in savanna tree cover following wholesale large-mammal declines. *Journal of Ecology*, 104 (1), 79–89.

Datry, T., K. Fritz and C. Leigh, 2016: Challenges, developments and perspectives in intermittent river ecology. *Freshwater Biology*, 61 (8), 1171-1180, doi:10.1111/fwb.12789.

Daufresne, M., P. Bady and J. F. Fruget, 2007: Impacts of global changes and extreme hydroclimatic events on macroinvertebrate community structures in the French Rhone River. *Oecologia*, 151 (3), 544-59, doi:10.1007/s00442-006-0655-1.

Daufresne, M., K. Lengfellner and U. Sommer, 2009: Global warming benefits the small in aquatic ecosystems. - 106 (31).

Daufresne, M., M. C. Roger, H. Capra and N. Lamouroux, 2004: Long-term changes within the invertebrate and fish communities of the Upper Rhone River: effects of climatic factors. *Global Change Biology*, 10 (1), 124-140, doi:DOI 10.1046/j.1529-8817.2003.00720.x.

Davies-Barnard, T. et al., 2015: Quantifying the relative importance of land cover change from climate and land-use in the representative concentration pathways. *Global Biogechemical Cycles*, doi:10.1002/2014gb004949.

Davies, A. B., A. Gaylard and G. P. Asner, 2018: Megafaunal effects on vegetation structure throughout a densely wooded African landscape. *Ecological Applications*, 28 (2), 398–408.

Davis, K. T. et al., 2019: Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. *Proceedings of the National Academy of Sciences*, 116 (13), 6193–6198.

Dawson, T. P. et al., 2011: Beyond predictions: biodiversity conservation in a changing climate. *Science (New York, N.Y.)*, 332 (6025), 53-58, doi:10.1126/science.1200303.

de Andrade, R. B. et al., 2017: Scenarios in tropical forest degradation: carbon stock trajectories for REDD+. *Carbon Balance and Management*, 12 (1), 6, doi:10.1186/s13021-017-0074-0.

de Eyto, E. et al., 2016: Response of a humic lake ecosystem to an extreme precipitation event: physical, chemical, and biological implications. *Inland Waters*, 6 (4), 483-498, doi:10.5268/iw-6.4.875.

De Faria, B. L. et al., 2017: Current and future patterns of fire-induced forest degradation in Amazonia. *Environmental Research Letters*, 12 (9), 095005.

de Paula, M. D., J. Groeneveld and A. Huth, 2015: Tropical forest degradation and recovery in fragmented landscapes - Simulating changes in tree community, forest hydrology and carbon balance. *Global Ecology and Conservation*, 3, 664-677, doi:10.1016/j.gecco.2015.03.004.

De Sy, V. et al., 2015: Land use patterns and related carbon losses following deforestation in South America. *Environmental Research Letters*, 10 (12), doi:10.1088/1748-9326/10/12/124004.

Deere, N. J. et al., 2018: High Carbon Stock forests provide co-benefits for tropical biodiversity. *Journal of Applied Ecology*, 55 (2), 997-1008, doi:10.1111/1365-2664.13023.

DelSontro, T., J. J. Beaulieu and J. A. Downing, 2018: Greenhouse gas emissions from lakes and impoundments: Upscaling in the face of global change. *Limnology and Oceanography Letters*, 3 (3), 64-75, doi:10.1002/lol2.10073.

Dempson, B. et al., 2017: Influence of climate and abundance on migration timing of adult Atlantic salmon (Salmo salar) among rivers in Newfoundland and Labrador. *Ecology of Freshwater Fish*, 26 (2), 247-259, doi:10.1111/eff.12271.

Dennison, P. E., S. C. Brewer, J. D. Arnold and M. A. Moritz, 2014: Large wildfire trends in the western United States, 1984-2011. *Geophysical Research Letters*, 41 (8), 2928-2933, doi:10.1002/2014gl059576.

Deutsch, C. A. et al., 2018: Increase in crop losses to insect pests in a warming climate. *Science*, 361 (6405), 916-919, doi:https://doi.org/10.1126/science.aat3466.

Devaraju, N., G. Bala and A. Modak, 2015: Effects of large-scale deforestation on precipitation in the monsoon regions: Remote versus local effects. *Proceedings of the National Academy of Sciences of the United States of America*, 112 (11), 3257-3262, doi:10.1073/pnas.1423439112.

Devictor, V. et al., 2012: Differences in the climatic debts of birds and butterflies at a continental scale. *Nature Climate Change*, 2 (2), 121-124, doi:10.1038/nclimate1347.

Devisscher, T., Y. Malhi, V. D. R. Landivar and I. Oliveras, 2016: Understanding ecological transitions under recurrent wildfire: A case study in the seasonally dry tropical forests of the Chiquitania, Bolivia. *Forest Ecology and Management*, 360, 273-286, doi:10.1016/j.foreco.2015.10.033.

Dey, C. J. et al., 2017: Increasing nest predation will be insufficient to maintain polar bear body condition in the face of sea ice loss. *Global Change Biology*, 23 (5), 1821-1831, doi:10.1111/gcb.13499.

Di Marco, M. et al., 2018: The extent and predictability of the biodiversity-carbon correlation. *Ecology Letters*, 21 (3), 365-375, doi:10.1111/ele.12903.

Di Virgilio, G. et al., 2019: Climate Change Increases the Potential for Extreme Wildfires. *Geophysical Research Letters*, 46 (14), 8517-8526, doi:10.1029/2019gl083699.

Diba, I., M. Camara, A. B. Sarr and A. Diedhiou, 2018: Potential Impacts of Land Cover Change on the Interannual Variability of Rainfall and Surface Temperature over West Africa. *Atmosphere*, 9 (10), doi:10.3390/atmos9100376.

Diffenbaugh, N. S., D. L. Swain and D. Touma, 2015: Anthropogenic warming has increased drought risk in California. *Proceedings of the National Academy of Sciences of the United States of America*, 112 (13), 3931-3936, doi:10.1073/pnas.1422385112.

Doblas-Miranda, E. et al., 2015: Reassessing global change research priorities in mediterranean terrestrial ecosystems: how far have we come and where do we go from here? *Global Ecology and Biogeography*, 24 (1), 25-43.

Doll, P. et al., 2016: Modelling Freshwater Resources at the Global Scale: Challenges and Prospects. *Surveys in Geophysics*, 37 (2), 195-221, doi:10.1007/s10712-015-9343-1.

Domis, D. S., 2013: Plankton dynamics under different climatic conditions in space and time. *Freshwater Biology*, 58, 463-482.

Domisch, S. et al., 2013: Modelling distribution in European stream macroinvertebrates under future climates. *Global Change Biology*, 19 (3), 752-62, doi:10.1111/gcb.12107.

Domisch, S., S. C. Jahnig and P. Haase, 2011: Climate-change winners and losers: stream macroinvertebrates of a submontane region in Central Europe. *Freshwater Biology*, 56 (10), 2009-2020, doi:DOI 10.1111/j.1365-2427.2011.02631.x.

Donato, D. C., B. J. Harvey and M. G. Turner, 2016: Regeneration of montane forests 24 years after the 1988 Yellowstone fires: A fire-catalyzed shift in lower treelines? *Ecosphere*, 7 (8), doi:10.1002/ecs2.1410.

Donato, D. C. et al., 2011: Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience*, 4 (5), 293-297, doi:10.1038/ngeo1123.

Donohue, R. J., M. L. Roderick, T. R. McVicar and G. D. Farquhar, 2013: Impact of CO2 fertilization on maximum foliage cover across the globe's warm, arid environments. *Geophysical Research Letters*, 40 (12), 3031-3035, doi:10.1002/grl.50563.

Dowdy, A. J., 2018: Climatological Variability of Fire Weather in Australia. *Journal of Applied Meteorology and Climatology*, 57 (2), 221-234, doi:10.1175/jamc-d-17-0167.1.

Dowdy, A. J. et al., 2019: Future changes in extreme weather and pyroconvection risk factors for Australian wildfires. *Scientific Reports*, 9, doi:10.1038/s41598-019-46362-x.

Drake, T. W., P. A. Raymond and R. G. M. Spencer, 2018: Terrestrial carbon inputs to inland waters: A current synthesis of estimates and uncertainty. *Limnology and Oceanography Letters*, 3 (3), 132-142, doi:10.1002/lol2.10055.

Du, E. and W. de Vries, 2018: Nitrogen-induced new net primary production and carbon sequestration in global forests. *Environmental Pollution*, 242, 1476-1487, doi:10.1016/j.envpol.2018.08.041.

Du, H. B. et al., 2018: Warming-induced upward migration of the alpine treeline in the Changbai Mountains, northeast China. *Global Change Biology*, 24 (3), 1256-1266, doi:10.1111/gcb.13963.

du Toit, J. C. O., T. G. O'Connor and L. Van den Berg, 2015: Photographic evidence of fire-induced shifts from dwarf-shrub-to grass-dominated vegetation in Nama-Karoo. *South African Journal of Botany*, 101, 148-152.

du Toit, J. C. O. and T. G. O'Connor, 2014: Changes in rainfall pattern in the eastern Karoo, South Africa, over the past 123 years. *Water Sa*, 40 (3), 453-460.

Duffy, P. B. et al., 2019: Strengthened scientific support for the Endangerment Finding for atmospheric greenhouse gases. *Science*, 363 (6427), eaat5982, doi:10.1126/science.aat5982.

Durigan, G. and J. A. Ratter, 2016: The need for a consistent fire policy for Cerrado conservation. *Journal of Applied Ecology*, 53 (1), 11-15.

Durwall, 2016: The extend of the threat to freshwater fish biodiversity. In: Conservation of Freshwater Fishes [Closs (ed.)]. Cambridge University Press.

Duveiller, G., J. Hooker and A. Cescatti, 2018: The mark of vegetation change on Earth's surface energy balance. *Nature Communications*, 9, doi:10.1038/s41467-017-02810-8.

Earl, N. and I. Simmonds, 2018: Spatial and Temporal Variability and Trends in 2001-2016 Global Fire Activity. *Journal of Geophysical Research (Atmospheres)*, 123, 2524-2536, doi:10.1002/2017JD027749.

Earl, N., I. Simmonds and N. Tapper, 2015: Weekly cycles of global fires-Associations with religion, wealth and culture, and insights into anthropogenic influences on global climate. *Geophysical Research Letters*, 42 (21), 9579-9589, doi:10.1002/2015gl066383.

Early, R. et al., 2016: Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications*, 7, doi:10.1038/ncomms12485.

Early, R. and S. Keith, 2019: Geographically variable biotic interactions and implications for species ranges. *Global Ecology and Biogeography*, 28 (1), 42-53, doi:10.1111/geb.12861.

Eigenbrod, F., P. Gonzalez, J. Dash and I. Steyl, 2015: Vulnerability of ecosystems to climate change moderated by habitat intactness. *Global Change Biology*, 21 (1), 275-286, doi:10.1111/gcb.12669.

Elith, J. and J. R. Leathwick, 2009: Species Distribution Models: Ecological Explanation and Prediction Across Space and Time. *Annual Review of Ecology Evolution and Systematics*, 40, 677-697, doi:10.1146/annurev.ecolsys.110308.120159.

Ellis, E. C. and N. Ramankutty, 2008: Putting people in the map: anthropogenic biomes of the world. *Frontiers in Ecology and the Environment*, 6 (8), 439-447, doi:10.1890/070062.

Ellison, D. et al., 2017: Trees, forests and water: Cool insights for a hot world. *Global Environmental Change-Human and Policy Dimensions*, 43, 51-61, doi:10.1016/j.gloenvcha.2017.01.002.

Elmendorf, S. C. et al., 2015: Experiment, monitoring, and gradient methods used to infer climate change effects on plant communities yield consistent patterns. *Proceedings of the National Academy of Sciences*, 112 (2), 448-452.

Elmhagen, B., J. Kindberg, P. Hellström and A. Angerbjörn, 2015: A boreal invasion in response to climate change? Range shifts and community effects in the borderland between forest and tundra. *AMBIO*, 44 (1), 39–50.

Elsen, P. R., W. B. Monahan and A. M. Merenlender, 2018: Global patterns of protection of elevational gradients in mountain ranges. *Proceedings of the National Academy of Sciences of the United States of America*, 115 (23), 6004-6009, doi:10.1073/pnas.1720141115.

Englund, O. et al., 2017: A new high-resolution nationwide aboveground carbon map for Brazil. *Geo-Geography and Environment*, 4 (2), doi:10.1002/geo2.45.

Enright, N. J. et al., 2015: Interval squeeze: altered fire regimes and demographic responses interact to threaten woody species persistence as climate changes. *Frontiers in Ecology and the Environment*, 13 (5), 265-272, doi:10.1890/140231.

Enriquez-Urzelai, U. et al., 2019: Are amphibians tracking their climatic niches in response to climate warming? A test with Iberian amphibians. *Climatic Change*, 154 (1-2), 289-301, doi:10.1007/s10584-019-02422-9.

Erb, K. H. et al., 2018a: Unexpectedly large impact of forest management and grazing on global vegetation biomass. *Nature*, 553 (7686), 73-+, doi:10.1038/nature25138.

Erb, W. M. et al., 2018b: Wildfire smoke impacts activity and energetics of wild Bornean orangutans. *Scientific Reports*, 8 (1), 7606-7606, doi:10.1038/s41598-018-25847-1.

Espírito-Santo, M. M. et al., 2016: Understanding patterns of land-cover change in the Brazilian Cerrado from 2000 to 2015. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371 (1703), 20150435.

Esquivel-Muelbert, A. et al., 2019: Compositional response of Amazon forests to climate change. *Global Change Biology*, 25 (1), 39-56.

Evaristo, J. and J. J. McDonnell, 2019: Global analysis of streamflow response to forest management. *Nature*, 570 (7762), 455-461, doi:10.1038/s41586-019-1306-0.

Exbrayat, J. F., Y. Y. Liu and M. Williams, 2017: Impact of deforestation and climate on the Amazon Basin's above-ground biomass during 1993-2012. *Scientific Reports*, 7, doi:10.1038/s41598-017-15788-6.

Exbrayat, J. F. and M. Williams, 2015: Quantifying the net contribution of the historical Amazonian deforestation to climate change. *Geophysical Research Letters*, 42 (8), 2968-2976, doi:10.1002/2015gl063497.

Fadrique, B. et al., 2018: Widespread but heterogeneous responses of Andean forests to climate change. *Nature*, 564 (7735), 207-212, doi:10.1038/s41586-018-0715-9.

FAO, 2018: *The state of the World´s forests 2018 - Forest pathways to sustainable development*. Food and Agriculture Organization of the United Nations (FAO), FAO, Rome, 118 [Available at: http://www.fao.org/3/I9535EN/i9535en.pdf].

Farmer, T., E. Marschall, K. Dabrowski and S. Ludsin, 2015: Short winters threaten temperate fish populations. *Nature Communications*, 6, doi:10.1038/ncomms8724.

Fauset, S. et al., 2015: Hyperdominance in Amazonian forest carbon cycling. *Nature Communications*, 6, doi:10.1038/ncomms7857.

Feeley, K. J., J. T. Stroud and T. M. Perez, 2017: Most ‘global’ reviews of species’ responses to climate change are not truly global. *Diversity and Distributions*, 23 (3), 231-234, doi:10.1111/ddi.12517.

Fei, S. et al., 2018: Impacts of climate on the biodiversity-productivity relationship in natural forests. *Nature Communications*, 9, doi:10.1038/s41467-018-07880-w.

Fernandez-Martinez, M. et al., 2019: Global trends in carbon sinks and their relationships with CO2 and temperature. *Nature Climate Change*, 9 (1), 73-+, doi:10.1038/s41558-018-0367-7.

Fernandez, M., H. H. Hamilton and L. M. Kueppers, 2015: Back to the future: using historical climate variation to project near-term shifts in habitat suitable for coast redwood. *Global Change Biology*, 21 (11), 4141-4152, doi:10.1111/gcb.13027.

Ferreira, J. et al., 2018: Carbon-focused conservation may fail to protect the most biodiverse tropical forests. *Nature Climate Change*, 8 (8), 744, doi:10.1038/s41558-018-0225-7.

Fettig, C. J., L. A. Mortenson, B. M. Bulaon and P. B. Foulk, 2019: Tree mortality following drought in the central and southern Sierra Nevada, California, US. *Forest Ecology and Management*, 432, 164-178, doi:10.1016/j.foreco.2018.09.006.

Ficetola, G. F. and L. Maiorano, 2016: Contrasting effects of temperature and precipitation change on amphibian phenology, abundance and performance. *Oecologia*, 181 (3), 683-693, doi:10.1007/s00442-016-3610-9.

Field, C. B. et al., 2014: Summary for Policymakers. In: Climate Change 2014: Impacts, Adaptation and Vulnerability - Contributions of the Working Group II to the Fifth Assessment Report. Cambridge University Press, Cambridge and New York, 1-32.

Finstad, A. et al., 2016: From greening to browning: Catchment vegetation development and reduced S-deposition promote organic carbon load on decadal time scales in Nordic lakes. *Scientific Reports*, 6, doi:10.1038/srep31944.

Fischlin, A. et al., 2007: Ecosystems, their properties, goods, and services. In: Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Parry, M. L., O. F. Canziani, J. P. Palutifok, P. Van der Linden and C. E. Hanson (eds.)]. Cambridge University Press, Cambridge, UK, 211-272.

Flannigan, M. et al., 2013: Global wildland fire season severity in the 21st century. *Forest Ecology and Management*, 294, 54-61, doi:10.1016/j.foreco.2012.10.022.

Fleischer, K. et al., 2019: Amazon forest response to CO2 fertilization dependent on plant phosphorus acquisition. *Nature Geoscience*, 12 (9), 736-741, doi:10.1038/s41561-019-0404-9.

Foden, W. B. et al., 2013: Identifying the World's Most Climate Change Vulnerable Species: A Systematic Trait-Based Assessment of all Birds, Amphibians and Corals. *PLOS ONE*, 8 (6), e65427, doi:10.1371/journal.pone.0065427.

Foden, W. B. et al., 2019: Climate change vulnerability assessment of species. *Wiley Interdisciplinary Reviews: Climate Change*, 10 (1), e551, doi:10.1002/wcc.551.

Fonseca, M. G. et al., 2019: Effects of climate and land-use change scenarios on fire probability during the 21st century in the Brazilian Amazon. *Global Change Biology*, 25 (9), 2931-2946, doi:10.1111/gcb.14709.

Fontes, C. G. et al., 2018: Dry and hot: the hydraulic consequences of a climate change-type drought for Amazonian trees. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, 373 (1760), doi:10.1098/rstb.2018.0209.

Fordham, D. et al., 2018: How complex should models be? Comparing correlative and mechanistic range dynamics models. *Global Change Biology*, 24 (3), 1357-1370, doi:10.1111/gcb.13935.

Forkel, M. et al., 2019: Recent global and regional trends in burned area and their compensating environmental controls. *Environmental Research Communications*, 1 (5), 051005, doi:10.1088/2515-7620/ab25d2.

Fox-Hughes, P. et al., 2014: Future fire danger climatology for Tasmania, Australia, using a dynamically downscaled regional climate model. *International Journal of Wildland Fire*, 23 (3), 309-321, doi:10.1071/wf13126.

Frank, D. C. et al., 2015: Water-use efficiency and transpiration across European forests during the Anthropocene. *Nature Climate Change*, 5 (6), 579.

Franks, S. J. and A. A. Hoffmann, 2012: Genetics of Climate Change Adaptation. In: Annual Review of Genetics, Vol 46 [Bassler, B. L. (ed.)]. Annual Reviews, Palo Alto, 46, 185-208.

Friedlingstein, P. et al., 2014: Uncertainties in CMIP5 Climate Projections due to Carbon Cycle Feedbacks. *Journal of Climate*, 27 (2), 511-526, doi:10.1175/jcli-d-12-00579.1.

Friend, A. D. et al., 2014: Carbon residence time dominates uncertainty in terrestrial vegetation responses to future climate and atmospheric CO2. *Proceedings of the National Academy of Sciences of the United States of America*, 111 (9), 3280-3285, doi:10.1073/pnas.1222477110.

Frolicher, T. L. and C. Laufkotter, 2018: Emerging risks from marine heat waves. *Nature Communications*, 9, doi:10.1038/s41467-018-03163-6.

Frost, G. V. et al., 2018: Seasonal and long-term changes to active-layer temperatures after tall shrubland expansion and succession in Arctic tundra. *Ecosystems*, 21 (3), 507-520.

Fulbright, T. E., K. W. Davies and S. R. Archer, 2018: Wildlife responses to brush management: a contemporary evaluation. *Rangeland Ecology & Management*, 71 (1), 35-44.

Fullerton, A. et al., 2018: Longitudinal thermal heterogeneity in rivers and refugia for coldwater species: effects of scale and climate change. *Aquatic Sciences*, 80 (1), doi:10.1007/s00027-017-0557-9.

Fuss, S. et al., 2018: Negative emissions-Part 2: Costs, potentials and side effects. *Environmental Research Letters*, 13 (6), doi:10.1088/1748-9326/aabf9f.

Gang, C. et al., 2017: Modeling the dynamics of distribution, extent, and NPP of global terrestrial ecosystems in response to future climate change. *Global and Planetary Change*, 148, 153-165, doi:10.1016/j.gloplacha.2016.12.007.

Gang, C. et al., 2015: Comparative assessment of grassland NPP dynamics in response to climate change in China, North America, Europe and Australia from 1981 to 2010. *Journal of Agronomy and Crop Science*, 201 (1), 57-68.

Gao, Q. Z. et al., 2016a: Climate change and its impacts on vegetation distribution and net primary productivity of the alpine ecosystem in the Qinghai-Tibetan Plateau. *Science of the Total Environment*, 554, 34-41, doi:10.1016/j.scitotenv.2016.02.131.

Gao, Q. Z. et al., 2016b: Changes in Global Grassland Productivity during 1982 to 2011 Attributable to Climatic Factors. *Remote Sensing*, 8 (5), doi:10.3390/rs8050384.

Gao, Q. Z. et al., 2016c: Climatic change controls productivity variation in global grasslands. *Scientific Reports*, 6, doi:10.1038/srep26958.

Garcia, R. et al., 2014: Matching species traits to projected threats and opportunities from climate change. *Journal of Biogeography*, 41 (4), 724-735, doi:10.1111/jbi.12257.

Gardner, T. A. et al., 2009: Prospects for tropical forest biodiversity in a human-modified world. *Ecology Letters*, 12 (6), 561-582, doi:10.1111/j.1461-0248.2009.01294.x.

Gasser, T. et al., 2018: Path-dependent reductions in CO2 emission budgets caused by permafrost carbon release. *Nature Geoscience*, 11 (11), 830-+, doi:10.1038/s41561-018-0227-0.

Gatti, L. V. et al., 2014: Drought sensitivity of Amazonian carbon balance revealed by atmospheric measurements. *Nature*, 506 (7486), 76.

Gatti, R. C. et al., 2019: Accelerating upward treeline shift in the Altai Mountains under last-century climate change. *Scientific Reports*, 9, doi:10.1038/s41598-019-44188-1.

Gauthier, G. et al., 2013: Long-term monitoring at multiple trophic levels suggests heterogeneity in responses to climate change in the Canadian Arctic tundra. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 368 (1624), 20120482.

Gaveau, D. L. A. et al., 2014: Major atmospheric emissions from peat fires in Southeast Asia during non-drought years: Evidence from the 2013 Sumatran fires. *Scientific Reports*, 4, doi:10.1038/srep06112.

Ghimire, B. et al., 2015: Large carbon release legacy from bark beetle outbreaks across Western United States. *Global Change Biology*, 21 (8), 3087-3101, doi:10.1111/gcb.12933.

Giannini, A., R. Saravanan and P. Chang, 2003: Oceanic forcing of Sahel rainfall on interannual to interdecadal time scales. *Science*, 302 (5647), 1027-1030, doi:10.1126/science.1089357.

Gibson, L. et al., 2011: Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature*, 478 (7369), 378.

Giersch, J. et al., 2015: Climate-induced range contraction of a rare alpine aquatic invertebrate. *Freshwater Science*, 34 (1), 53-65, doi:10.1086/679490.

Giesen, W. and G. Z. Anshari, 2016: Danau Sentarum National Park (Indonesia). *The Wetland Book: II: Distribution, Description and Conservation*, 1-10.

Giglio, L. et al., 2018: The Collection 6 MODIS burned area mapping algorithm and product. *Remote Sensing of Environment*, 217, 72-85, doi:10.1016/j.rse.2018.08.005.

Giglio, L., J. T. Randerson and G. R. van der Werf, 2013: Analysis of daily, monthly, and annual burned area using the fourth-generation global fire emissions database (GFED4). *Journal of Geophysical Research-Biogeosciences*, 118 (1), 317-328, doi:10.1002/jgrg.20042.

Gillingham, P. K. et al., 2015: The effectiveness of protected areas in the conservation of species with changing geographical ranges. *Biological Journal of the Linnean Society*, 115 (3), 707-717, doi:10.1111/bij.12506.

Giuntoli, I., J. P. Vidal, C. Prudhomme and D. M. Hannah, 2015: Future hydrological extremes: the uncertainty from multiple global climate and global hydrological models. *Earth System Dynamics*, 6 (1), 267-285, doi:10.5194/esd-6-267-2015.

Gleason, K. E. et al., 2019: Four-fold increase in solar forcing on snow in western U.S. burned forests since 1999. *Nature Communications*, 10 (1), 2026, doi:10.1038/s41467-019-09935-y.

Gomez-Gener, L. et al., 2016: When Water Vanishes: Magnitude and Regulation of Carbon Dioxide Emissions from Dry Temporary Streams. *Ecosystems*, 19 (4), 710-723, doi:10.1007/s10021-016-9963-4.

Gonzalez, C. et al., 2010a: Climate Change and Risk of Leishmaniasis in North America: Predictions from Ecological Niche Models of Vector and Reservoir Species. *Plos Neglected Tropical Diseases*, 4 (1), doi:10.1371/journal.pntd.0000585.

Gonzalez, P. et al., 2015: Aboveground live carbon stock changes of California wildland ecosystems, 2001–2010. *Forest Ecology and Management*, 348, 68-77, doi:10.1016/j.foreco.2015.03.040.

Gonzalez, P., R. P. Neilson, J. M. Lenihan and R. J. Drapek, 2010b: Global patterns in the vulnerability of ecosystems to vegetation shifts due to climate change. *Global Ecology and Biogeography*, 19 (6), 755-768, doi:10.1111/j.1466-8238.2010.00558.x.

Gonzalez, P., C. Tucker and H. Sy, 2012: Tree density and species decline in the African Sahel attributable to climate. *Journal of Arid Environments*, 78, 55-64, doi:10.1016/j.jaridenv.2011.11.001.

Gonzalez, P. et al., 2018: Disproportionate magnitude of climate change in United States national parks. *Environmental Research Letters*, 13 (10), doi:10.1088/1748-9326/aade09.

Gotanda, K. M. et al., 2015: Linking macrotrends and microrates: Re-evaluating microevolutionary support for Cope's rule. *Evolution*, 69 (5), 1345-1354, doi:10.1111/evo.12653.

Goulden, M. L. and R. C. Bales, 2019: California forest die-off linked to multi-year deep soil drying in 2012-2015 drought. *Nature Geoscience*, 12 (8), 632-+, doi:10.1038/s41561-019-0388-5.

Greenwood, O. et al., 2016: Using in situ management to conserve biodiversity under climate change. *Journal of Applied Ecology*, 53 (3), 885-894, doi:10.1111/1365-2664.12602.

Greenwood, S. et al., 2017: Tree mortality across biomes is promoted by drought intensity, lower wood density and higher specific leaf area. *Ecology Letters*, 20 (4), 539-553, doi:10.1111/ele.12748.

Griscom, B. W. et al., 2017: Natural climate solutions. *Proceedings of the National Academy of Sciences*, 114, 11645-11650.

Grosse, G. et al., 2016: Changing permafrost in a warming world and feedbacks to the Earth system. *Environmental Research Letters*, 11 (4), 040201.

Gustine, D. D. et al., 2014: Climate-Driven Effects of Fire on Winter Habitat for Caribou in the Alaskan-Yukon Arctic. *PLOS ONE*, 9 (7), doi:10.1371/journal.pone.0100588.

Haddad, N. M. et al., 2015: Habitat fragmentation and its lasting impact on Earth’s ecosystems. *Science Advances*, 1 (2), e1500052.

Haensler, A., J. Cermak, S. Hagemann and D. Jacob, 2011: Will the southern African west coast fog be affected by future climate change? Results of an initial fog projection using a regional climate model. *Erdkunde*, 261-275.

Hallfors, M. H., S. Aikio and L. E. Schulman, 2017: Quantifying the need and potential of assisted migration. *Biological Conservation*, 205, 34-41, doi:10.1016/j.biocon.2016.11.023.

Hallgren, W. et al., 2016: The Biodiversity and Climate Change Virtual Laboratory: Where ecology meets big data. *Environmental Modelling & Software*, 76, 182-186, doi:10.1016/j.envsoft.2015.10.025.

Halupka, L. and K. Halupka, 2017: The effect of climate change on the duration of avian breeding seasons: a meta-analysis. *Proceedings of the Royal Society B-Biological Sciences*, 284 (1867), doi:10.1098/rspb.2017.1710.

Hampton, S. et al., 2017: Ecology under lake ice. *Ecology Letters*, 20 (1), 98-111, doi:10.1111/ele.12699.

Hanberry, B. B. and M. H. Hansen, 2015: Advancement of tree species across ecotonal borders into non-forested ecosystems. *Acta Oecologica-International Journal of Ecology*, 68, 24-36, doi:10.1016/j.actao.2015.07.002.

Handmer, J., Y. et al., 2012: Changes in impacts of climate extremes: human systems and ecosystems. In: Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change (IPCC) [Field, C. B., C. Barros, T. F. Stocker, D. Qin, D. J. Dokken, K. Ebi, M. D. Mastrandrea, K. J. Mach, G. K. Plattner, C. D. Allen, M. Tignor and P. M. Midgley (eds.)]. Cambridge University Press, Cambridge and New York, 231-290.

Hanes, C. C. et al., 2019: Fire-regime changes in Canada over the last half century. *Canadian Journal of Forest Research*, 49 (3), 256-269, doi:10.1139/cjfr-2018-0293.

Hanke, H., L. Borjeson, K. Hylander and E. Enfors-Kautsky, 2016: Drought tolerant species dominate as rainfall and tree cover returns in the West African Sahel. *Land Use Policy*, 59, 111-120, doi:10.1016/j.landusepol.2016.08.023.

Hansen, M. C. et al., 2013: High-resolution global maps of 21st-century forest cover change. *Science*, 342 (6160), 850-853.

Hansen, W. D. et al., 2016: Forest-landscape structure mediates effects of a spruce bark beetle (Dendroctonus rufipennis) outbreak on subsequent likelihood of burning in Alaskan boreal forest. *Forest Ecology and Management*, 369, 38-46, doi:10.1016/j.foreco.2016.03.036.

Hao, Z., F. Hao, V. P. Singh and X. Zhang, 2018: Quantifying the relationship between compound dry and hot events and El Niño–southern Oscillation (ENSO) at the global scale. *Journal of Hydrology*, 567, 332-338, doi:https://doi.org/10.1016/j.jhydrol.2018.10.022.

Harris, R. M. B. et al., 2018a: Biological responses to the press and pulse of climate trends and extreme events. *Nature Climate Change*, 8 (7), 579-587, doi:10.1038/s41558-018-0187-9.

Harris, R. M. B. et al., 2018b: Exploring the Future of Fuel Loads in Tasmania, Australia: Shifts in Vegetation in Response to Changing Fire Weather, Productivity, and Fire Frequency. *Forests*, 9 (4), doi:10.3390/f9040210.

Hart, S. J., T. T. Veblen, D. Schneider and N. P. Molotch, 2017: Summer and winter drought drive the initiation and spread of spruce beetle outbreak. *Ecology*, 98 (10), 2698-2707, doi:10.1002/ecy.1963.

Hartmann, H. et al., 2018: Research frontiers for improving our understanding of drought-induced tree and forest mortality. *New Phytologist*, 218 (1), 15-28, doi:10.1111/nph.15048.

Harvey, C. A. et al., 2017: The use of Ecosystem-based Adaptation practices by smallholder farmers in Central America. *Agriculture, Ecosystems & Environment*, 246, 279-290.

Haslem, A., D. G. Nimmo, J. Q. Radford and A. F. Bennett, 2015: Landscape properties mediate the homogenization of bird assemblages during climatic extremes. *Ecology*, 96 (12), 3165-3174.

Hasler, C. T., D. Butman, J. D. Jeffrey and C. D. Suski, 2016: Freshwater biota and rising pCO2? *Ecology Letters*, 19 (1), 98-108, doi:doi:10.1111/ele.12549.

Haussmann, N. S., J. M. Kalwij and S. Bezuidenhout, 2016: Some ecological side-effects of chemical and physical bush clearing in a southern African rangeland ecosystem. *South African Journal of Botany*, 102, 234-239.

Havens, K. et al., 2016: Extreme Weather Events and Climate Variability Provide a Lens to How Shallow Lakes May Respond to Climate Change. *Water*, 8 (6), doi:10.3390/w8060229.

Hayden, B. et al., 2019: From clear lakes to murky waters - tracing the functional response of high-latitude lake communities to concurrent 'greening' and 'browning'. *Ecology Letters*, 22 (5), 807-816, doi:10.1111/ele.13238.

He, C., L. Zhou, W. Ma and Y. Wang, 2019: Spatial Assessment of Urban Climate Change Vulnerability during Different Urbanization Phases. *Sustainability*, 11 (8), 2406.

Heck, V., D. Gerten, W. Lucht and A. Popp, 2018: Biomass-based negative emissions difficult to reconcile with planetary boundaries. *Nature Climate Change*, 8 (2), 151-+, doi:10.1038/s41558-017-0064-y.

Hegerl, G. C. et al., 2010: Good practice guidance paper on detection and attribution related to anthropogenic climate change. In: *Meeting Report of the Intergovernmental Panel on Climate Change Expert Meeting on Detection and Attribution of Anthropogenic Climate Change*, IPCC Working Group I Technical Support Unit, University of Bern, Bern ….

Heller, N. E. and E. S. Zavaleta, 2009: Biodiversity management in the face of climate change: a review of 22 years of recommendations. *Biological Conservation*, 142 (1), 14-32.

Hempson, G. P., S. Archibald and W. J. Bond, 2017: The consequences of replacing wildlife with livestock in Africa. *Scientific Reports*, 7 (1), 17196.

Hennon, P. E. et al., 2012: Shifting Climate, Altered Niche, and a Dynamic Conservation Strategy for Yellow-Cedar in the North Pacific Coastal Rainforest. *BioScience*, 62 (2), 147-158, doi:10.1525/bio.2012.62.2.8.

Henzler, J. et al., 2018: A squeeze in the suitable fire interval: Simulating the persistence of fire-killed plants in a Mediterranean-type ecosystem under drier conditions. *Ecological Modelling*, 389, 41-49, doi:10.1016/j.ecolmodel.2018.10.010.

Hessburg, P. F. et al., 2016: Tamm Review: Management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. *Forest Ecology and Management*, 366, 221-250, doi:10.1016/j.foreco.2016.01.034.

Hesslerova, P., H. Huryna, J. Pokorny and J. Prochazka, 2018: The effect of forest disturbance on landscape temperature. *Ecological Engineering*, 120, 345-354, doi:10.1016/j.ecoleng.2018.06.011.

Heubes, J. et al., 2011: Modelling biome shifts and tree cover change for 2050 in West Africa: Biome shifts and tree cover change in West Africa. *Journal of Biogeography*, 38 (12), 2248-2258, doi:10.1111/j.1365-2699.2011.02560.x.

Hicke, J. A., A. J. H. Meddens and C. A. Kolden, 2016: Recent Tree Mortality in the Western United States from Bark Beetles and Forest Fires. *Forest Science*, 62 (2), 141-153, doi:10.5849/forsci.15-086.

Hickler, T. et al., 2012: Projecting the future distribution of European potential natural vegetation zones with a generalized, tree species-based dynamic vegetation model: Future changes in European vegetation zones. *Global Ecology and Biogeography*, 21 (1), 50-63, doi:10.1111/j.1466-8238.2010.00613.x.

Hidalgo, M., V. Mihneva, M. Vasconcellos and M. Bernal, 2018: Climate change impacts, vulnerabilities and adaptations: Mediterranean Sea and the Black Sea marine fisheries. In: Impacts of Climate Change on fisheries and aquaculture: Synthesis of current knowledge, adaptation and mitigation options. FAO Fisheries and Aquaculture Technical Paper No. 627 [Manuel Barange, T. B. M. B. S. F.-S. F. P. (ed.)].

Higgins, S. I. and S. Scheiter, 2012: Atmospheric CO2 forces abrupt vegetation shifts locally, but not globally. *Nature*, advance online publication, 10.1038/nature11238, doi:http://www.nature.com/nature/journal/vaop/ncurrent/abs/nature11238.html#supplementary-information.

Hillebrand, H. et al., 2018: Decomposing multiple dimensions of stability in global change experiments. *Ecology Letters*, 21 (1), 21-30, doi:10.1111/ele.12867.

Hobday, A. et al., 2018: Climate change impacts, vulnerabilities and adaptations: Australian marine fisheries. In: Impacts of Climate Change on fisheries and aquaculture: Synthesis of current knowledge, adaptation and mitigation options [Barange, M. e. s. (ed.)], FAO 1-18. **http://hdl.handle.net/102.100.100/86715?index=1**.

Hobday, A. J. et al., 2016: A hierarchical approach to defining marine heatwaves. *Progress in Oceanography*, 141, 227-238, doi:10.1016/j.pocean.2015.12.014.

Hodgkins, S. B. et al., 2018: Tropical peatland carbon storage linked to global latitudinal trends in peat recalcitrance. *Nature Communications*, 9, doi:10.1038/s41467-018-06050-2.

Hoegh-Guldberg, O. et al., 2008: Assisted colonization and rapid climate change. *Science*, 321 (5887), 345-346, doi:DOI: 10.1126/science.1157897.

Hoegh-Guldberg, O. et al., 2018: Impacts of 1.5ºC global warming on natural and human systems. In: IPCC special report on the impacts of global warming of 1.5 °C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change. http://www.ipcc.ch/report/sr15/.

Hof, C. et al., 2018: Bioenergy cropland expansion may offset positive effects of climate change mitigation for global vertebrate diversity. *Proceedings of the National Academy of Sciences*, 115 (52), 13294-13299, doi:10.1073/pnas.1807745115.

Hoffman, M. T., R. F. Rohde and L. Gillson, 2019: Rethinking catastrophe? Historical trajectories and modelled future vegetation change in southern Africa. *Anthropocene*, 25, 100189, doi:10.1016/j.ancene.2018.12.003.

Hoffmann, A. A. et al., 2019: Impacts of recent climate change on terrestrial flora and fauna: Some emerging Australian examples. *Austral Ecology*, 44 (1), 3-27, doi:10.1111/aec.12674.

Hoffmann, A. A. and C. M. Sgro, 2011: Climate change and evolutionary adaptation. *Nature*, 470 (7335), 479-485, doi:10.1038/nature09670.

Holbrook, N. J. et al., 2019: A global assessment of marine heatwaves and their drivers. *Nature Communications*, 10, doi:10.1038/s41467-019-10206-z.

Holden, Z. A. et al., 2018: Decreasing fire season precipitation increased recent western US forest wildfire activity. *Proceedings of the National Academy of Sciences*, 115 (36), E8349-E8357, doi:10.1073/pnas.1802316115.

Holz, A., S. W. Wood, T. T. Veblen and D. Bowman, 2015: Effects of high-severity fire drove the population collapse of the subalpine Tasmanian endemic conifer Athrotaxis cupressoides. *Global Change Biology*, 21 (1), 445-458, doi:10.1111/gcb.12674.

Hooper, D. U. et al., 2012: A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature*, 486, 105, doi:10.1038/nature11118 https://www.nature.com/articles/nature11118#supplementary-information.

Hoover, D. L., A. K. Knapp and M. D. Smith, 2014: Resistance and resilience of a grassland ecosystem to climate extremes. *Ecology*, 95 (9), 2646-2656.

Hopkins, J. et al., 2007: Conserving biodiversity in a changing climate: guidance on building capacity to adapt. *Defra, London*, 26.

Horn, K. J. and S. B. St. Clair, 2017: Wildfire and exotic grass invasion alter plant productivity in response to climate variability in the Mojave Desert. *Landscape Ecology*, 32 (3), 635-646, doi:10.1007/s10980-016-0466-7.

Hoscilo, A., S. E. Page, K. J. Tansey and J. O. Rieley, 2011: Effect of repeated fires on land-cover change on peatland in southern Central Kalimantan, Indonesia, from 1973 to 2005. *International Journal of Wildland Fire*, 20 (4), 578-588.

Houghton, R. A. et al., 2012: Carbon emissions from land use and land-cover change. *Biogeosciences*, 9 (12), 5125-5142, doi:10.5194/bg-9-5125-2012.

Houghton, R. A. and A. A. Nassikas, 2017: Global and regional fluxes of carbon from land use and land cover change 1850–2015. *Global Biogeochemical Cycles*, 31 (3), 456-472, doi:10.1002/2016GB005546.

Hovenden, M. J., P. C. D. Newton and K. E. Wills, 2014: Seasonal not annual rainfall determines grassland biomass response to carbon dioxide. *Nature*, 511 (7511), 583.

Høye, T. T. et al., 2013: Shorter flowering seasons and declining abundance of flower visitors in a warmer Arctic. *Nature Climate Change*, 3 (8), 759.

Hu, F. S. et al., 2015: Arctic tundra fires: natural variability and responses to climate change. *Frontiers in Ecology and the Environment*, 13, 369-377.

Huang, B. Y. et al., 2015: Extended Reconstructed Sea Surface Temperature Version 4 (ERSST.v4). Part I: Upgrades and Intercomparisons. *Journal of Climate*, 28 (3), 911-930, doi:10.1175/jcli-d-14-00006.1.

Huber, V., C. Wagner, D. Gerten and R. Adrian, 2012: To bloom or not to bloom: contrasting responses of cyanobacteria to recent heat waves explained by critical thresholds of abiotic drivers. *Oecologia*, 169 (1), 245-256, doi:10.1007/s00442-011-2186-7.

Hufft, R. A. and T. J. Zelikova, 2016: Ecological genetics, local adaptation, and phenotypic plasticity in Bromus tectorum in the context of a changing climate. In: Exotic Brome-Grasses in Arid and Semiarid Ecosystems of the Western US. Springer, 133–154.

Hugelius, G. et al., 2014: Estimated stocks of circumpolar permafrost carbon with quantified uncertainty ranges and identified data gaps. *Biogeosciences*, 11 (23), 6573-6593, doi:10.5194/bg-11-6573-2014.

Hugelius, G. et al., 2013: The Northern Circumpolar Soil Carbon Database: spatially distributed datasets of soil coverage and soil carbon storage in the northern permafrost regions. *Earth System Science Data*, 5 (1), 3-13.

Hughes, T. P. et al., 2017: Global warming and recurrent mass bleaching of corals. *Nature*, 543 (7645), 373-+, doi:10.1038/nature21707.

Hughes, T. P. et al., 2018: Global warming transforms coral reef assemblages. *Nature*, 556 (7702), 492, doi:10.1038/s41586-018-0041-2.

Humphrey, V. et al., 2018: Sensitivity of atmospheric CO2 growth rate to observed changes in terrestrial water storage. *Nature*, 560 (7720), 628-+, doi:10.1038/s41586-018-0424-4.

Huntingford, C. et al., 2013: Simulated resilience of tropical rainforests to CO2-induced climate change. *Nature Geoscience*, 6 (4), 268-273, doi:10.1038/ngeo1741.

Huntzinger, D. N. et al., 2017: Uncertainty in the response of terrestrial carbon sink to environmental drivers undermines carbon-climate feedback predictions. *Scientific Reports*, 7, doi:10.1038/s41598-017-03818-2.

Hurni, K., A. Heinimann, S. A.M. and J. Fox, 2017: Mapping the Expansion of Boom Crops in Mainland Southeast Asia Using Dense Time Stacks of Landsat Data. *Remote Sensing*, 9, 4, doi:doi:10.3390/rs9040320.

Hurteau, M. D., 2017: Quantifying the Carbon Balance of Forest Restoration and Wildfire under Projected Climate in the Fire-Prone Southwestern US. *PLOS ONE*, 12 (1), e0169275, doi:10.1371/journal.pone.0169275.

Iacob, O., J. S. Rowan, I. Brown and C. Ellis, 2014: Evaluating wider benefits of natural flood management strategies: an ecosystem-based adaptation perspective. *Hydrology Research*, 45 (6), 774-787.

Ibrahim, Y. Z., H. Balzter and J. Kaduk, 2018: Land degradation continues despite greening in the Nigeria-Niger border region. *Global Ecology and Conservation*, 16, doi:10.1016/j.gecco.2018.e00505.

IPBES, 2018a: *The IPBES regional assessment report on biodiversity and ecosystem services for Africa* [Archer, E. (ed.)]. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, 492 pp.

IPBES, 2018b: *The IPBES regional assessment report on biodiversity and ecosystem services for Asia and the Pacific* [Karki, M. (ed.)]. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, 612 pp.

IPBES, 2018c: *The IPBES regional assessment report on biodiversity and ecosystem services for Europe and Central Asia* [Rounsevell, M. (ed.)]. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, 892 pp.

IPBES, 2018d: *The IPBES regional assessment report on biodiversity and ecosystem services for the Americas* [Rice, J. (ed.)]. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, 656 pp.

IPBES, 2019: *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* [Brondizio, E. S., J. Settele, S. Díaz and H. T. Ngo (eds.)]. IPBES secretariat, Bonn, Germany.

IPCC, 2007: *Climate Change 2007 - Impacts, Adaptation and Vulnerability: Working Group II Contribution to the Fourth Assessment Report of the IPCC*. Cambridge University Press, 765 pp.

IPCC, 2012: *Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change (IPCC* [Field, C. B., V. Barros, T. F. Stocker and Q. Dahe (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC, 2018: Global Warming of 1.5 °C an IPCC special report on the impacts of global warming of 1.5 °C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change.

IPCC, 2019: *Summary for Policy Makers: IPCC Special Report on Climate Change and Land* [Ipcc (ed.)]. International Panel on Climate Change, Geneva, in press.

Isaak, D. et al., 2010: Effects of climate change and wildfire on stream temperatures and salmonid thermal habitat in a mountain river network. *Ecological Applications*, 20 (5), 1350-1371, doi:10.1890/09-0822.1.

Isaak, D. J. et al., 2016: Slow climate velocities of mountain streams portend their role as refugia for cold-water biodiversity. *Proceedings of the National Academy of Sciences*, 113 (16), 4374-4379, doi:10.1073/pnas.1522429113.

Isaak, D. J. et al., 2015: The cold-water climate shield: delineating refugia for preserving salmonid fishes through the 21st century. *Global Change Biology*, 21 (7), 2540-2553.

Ito, A., K. Nishina and H. M. Noda, 2016: Impacts of future climate change on the carbon budget of northern high-latitude terrestrial ecosystems: An analysis using ISI-MIP data. *Polar Science*, 10 (3), 346-355, doi:10.1016/j.polar.2015.11.002.

Iversen, C. M. et al., 2017: A global Fine-Root Ecology Database to address below-ground challenges in plant ecology. *New Phytologist*, 215 (1), 15-26.

Iversen, C. M. et al., 2015: The unseen iceberg: plant roots in arctic tundra. *New Phytologist*, 205 (1), 34-58.

Jackson, M. M. et al., 2016: Expansion of subalpine woody vegetation over 40 years on Vancouver Island, British Columbia, Canada. *Canadian Journal of Forest Research*, 46 (3), 437-443, doi:10.1139/cjfr-2015-0186.

Jackson, R. B. et al., 2005: Trading water for carbon with biological carbon sequestration. *Science*, 310 (5756), 1944-1947.

Jackson, R. B. et al., 2017: The Ecology of Soil Carbon: Pools, Vulnerabilities, and Biotic and Abiotic Controls. In: Annual Review of Ecology, Evolution, and Systematics, Vol 48 [Futuyma, D. J. (ed.)], 48, 419-445.

Jacobsen, A. L. and R. B. Pratt, 2018: Extensive drought-associated plant mortality as an agent of type-conversion in chaparral shrublands. *New Phytologist*, 219 (2), 498-504.

Jansen, 2019: Climate-sensitive controls on large spring emissions of Ch4 and CO2 from northern lakes. *Journal of Geophysical Research*.

Janssen, J. A. M. et al., 2016: *European Red List of Habitats. Part 2. Terrestrial and freshwater habitats*. Environment, Alterra Wageningen UR

IUCN

naturebureau

Susan Gubbay

John Rodwell Ecologist, Commission, E., 39 [Available at: https://ec.europa.eu/environment/nature/knowledge/pdf/terrestrial_EU_red_list_report.pdf].

Jaric, I. et al., 2019: Susceptibility of European freshwater fish to climate change: Species profiling based on life-history and environmental characteristics. *Global Change Biology*, 25 (2), 448-458, doi:10.1111/gcb.14518.

Jennings, E. et al., 2012: Effects of weather-related episodic events in lakes: an analysis based on high-frequency data. *Freshwater Biology*, 57 (3), 589-601, doi:10.1111/j.1365-2427.2011.02729.x.

Jeong, S. J. et al., 2011: Impact of vegetation feedback on the temperature and its diurnal range over the Northern Hemisphere during summer in a 2 x CO2 climate. *Climate Dynamics*, 37 (3-4), 821-833, doi:10.1007/s00382-010-0827-x.

Jeppesen, E. et al., 2012: Impacts of climate warming on the long-term dynamics of key fish species in 24 European lakes. *Hydrobiologia*, 694 (1), 1-39, doi:10.1007/s10750-012-1182-1.

Jewett, J. T. et al., 2011: Spatiotemporal Relationships between Climate and Whitebark Pine Mortality in the Greater Yellowstone Ecosystem. *Forest Science*, 57 (4), 320-335.

Jiang, Y. et al., 2017: Modeling long-term changes in tundra carbon balance following wildfire, climate change, and potential nutrient addition. *Ecological Applications*, 27 (1), 105-117, doi:10.1002/eap.1413.

Johnson, C. N. et al., 2018: Can trophic rewilding reduce the impact of fire in a more flammable world? *Philosophical Transactions of the Royal Society B: Biological Sciences*, 373 (1761), 20170443.

Jolly, W. M. et al., 2015: Climate-induced variations in global wildfire danger from 1979 to 2013. *Nature Communications*, 6, doi:10.1038/ncomms8537.

Jones, C., J. Lowe, S. Liddicoat and R. Betts, 2009: Committed terrestrial ecosystem changes due to climate change. *Nature Geoscience*, 2 (7), 484-487, doi:10.1038/ngeo555.

Jones, K. et al., 2018: One-third of global protected land is under intense human pressure. *Science*, 360 (6390), 788-+, doi:10.1126/science.aap9565.

Jones, K. R., J. E. M. Watson, H. P. Possingham and C. J. Klein, 2016: Incorporating climate change into spatial conservation prioritisation: A review. *Biological Conservation*, 194, 121-130, doi:10.1016/j.biocon.2015.12.008.

Jones, M. W., C. Santín, G. R. van der Werf and S. H. Doerr, 2019: Global fire emissions buffered by the production of pyrogenic carbon. *Nature Geoscience*, 12 (9), 742-747, doi:10.1038/s41561-019-0403-x.

Jung, M. et al., 2017: Compensatory water effects link yearly global land CO2 sink changes to temperature. *Nature*, 541 (7638), 516-520, doi:10.1038/nature20780.

Juroszek, P. and A. Von Tiedemann, 2013: Plant pathogens, insect pests and weeds in a changing global climate: a review of approaches, challenges, research gaps, key studies and concepts. *The Journal of Agricultural Science*, 151 (2), 163-188.

Kakouei, K. et al., 2018: Projected effects of Climate-change-induced flow alterations on stream macroinvertebrate abundances. *Ecology and Evolution*, 8 (6), 3393-3409, doi:10.1002/ece3.3907.

Kangur, K. et al., 2016: Changes in water temperature and chemistry preceding a massive kill of bottom-dwelling fish: an analysis of high-frequency buoy data of shallow Lake Vortsjarv (Estonia). *Inland Waters*, 6 (4), 535-542, doi:10.5268/IW-6.4.869.

Karell, P. et al., 2011: Climate change drives microevolution in a wild bird. *Nature Communications*, 2, doi:10.1038/ncomms1213.

Kasprzak, P. et al., 2017: Extreme weather event triggers cascade towards extreme turbidity in a clear-water lake. *Ecosystems*, 20 (8), 1407-1420.

Kaushal, S. et al., 2010: Rising stream and river temperatures in the United States. *Frontiers in Ecology and the Environment*, 8 (9), 461-466, doi:10.1890/090037.

Keeley, A. T. et al., 2018: New concepts, models, and assessments of climate-wise connectivity. *Environmental Research Letters*, 13 (7), 073002.

Keeley, J. E. et al., 2011: Fire as an evolutionary pressure shaping plant traits. *Trends in Plant Science*, 16 (8), 406-411.

Keenan, R. J. et al., 2015: Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management*, 352, 9–20.

Keenan, T. F. et al., 2014: Net carbon uptake has increased through warming-induced changes in temperate forest phenology. *Nature Climate Change*, 4, 598, doi:10.1038/nclimate2253 https://www.nature.com/articles/nclimate2253#supplementary-information.

Keenan, T. F. and W. J. Riley, 2018: Greening of the land surface in the world's cold regions consistent with recent warming. *Nature Climate Change*, 8 (9), 825-+, doi:10.1038/s41558-018-0258-y.

Keith, D. A. et al., 2014: Detecting Extinction Risk from Climate Change by IUCN Red List Criteria. *Conservation Biology*, 28 (3), 810-819, doi:10.1111/cobi.12234.

Keith, H., B. G. Mackey and D. B. Lindenmayer, 2009: Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. *Proceedings of the National Academy of Sciences of the United States of America*, 106 (28), 11635-11640, doi:10.1073/pnas.0901970106.

Kelley, D. I. et al., 2019: How contemporary bioclimatic and human controls change global fire regimes. *Nature Climate Change*, 9 (9), 690-+, doi:10.1038/s41558-019-0540-7.

Kelly, B., A. Whiteley and D. Tallmon, 2010: The Arctic melting pot. *Nature*, 468 (7326), 891-891, doi:10.1038/468891a.

Kelly, M., 2019: Adaptation to climate change through genetic accommodation and assimilation of plastic phenotypes. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 374 (1768), 10, doi:10.1098/rstb.2018.0176.

Kemp, K. B., P. E. Higuera, P. Morgan and J. T. Abatzoglou, 2019: Climate will increasingly determine post-fire tree regeneration success in low-elevation forests, Northern Rockies, USA. *Ecosphere*, 10 (1), doi:10.1002/ecs2.2568.

Kerby, J. and E. Post, 2013: Capital and income breeding traits differentiate trophic match–mismatch dynamics in large herbivores. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 368 (1624), 20120484.

Kettridge, N. et al., 2019: Severe wildfire exposes remnant peat carbon stocks to increased post-fire drying. *Scientific Reports*, 9 (1), 3727, doi:10.1038/s41598-019-40033-7.

Keuper, F. et al., 2017: Experimentally increased nutrient availability at the permafrost thaw front selectively enhances biomass production of deep-rooting subarctic peatland species. *Global Change Biology*, 23 (10), 4257-4266.

Keys, P. W., L. Wang-Erlandsson and L. J. Gordon, 2016: Revealing invisible water: moisture recycling as an ecosystem service. *PLOS ONE*, 11 (3), e0151993.

Kharouba, H. M. et al., 2018: Global shifts in the phenological synchrony of species interactions over recent decades. *Proceedings of the National Academy of Sciences of the United States of America*, 115 (20), 5211-5216, doi:10.1073/pnas.1714511115.

Kidane, Y. O., M. J. Steinbauer and C. Beierkuhnlein, 2019: Dead end for endemic plant species? A biodiversity hotspot under pressure. *Global Ecology and Conservation*, 19, doi:10.1016/j.gecco.2019.e00670.

Kim, H. et al., 2018: A protocol for an intercomparison of biodiversity and ecosystem services models using harmonized land-use and climate scenarios. *Geoscientific Model Development*, 11 (11), 4537-4562, doi:10.5194/gmd-11-4537-2018.

Kirchmeier-Young, M. C. et al., 2019: Attribution of the Influence of Human-Induced Climate Change on an Extreme Fire Season. *Earth's Future*, 7 (1), 2-10, doi:10.1029/2018EF001050.

Kirillin, G., 2010: Modeling the impact of global warming on water temperature and seasonal mixing regimes in small temperate lakes. *Boreal Environment Research*, 15 (2), 279-293.

Kirillin, G. et al., 2012: Physics of seasonally ice-covered lakes: a review. *Aquatic Sciences*, 74 (4), 659-682, doi:10.1007/s00027-012-0279-y.

Kirillin, G. and T. Shatwell, 2016: Generalized scaling of seasonal thermal stratification in lakes. *Earth-Science Reviews*, 161, 179-190, doi:10.1016/j.earscirev.2016.08.008.

Klinger, R. and M. Brooks, 2017: Alternative pathways to landscape transformation: invasive grasses, burn severity and fire frequency in arid ecosystems. *Journal of Ecology*, 105 (6), 1521-1533, doi:10.1111/1365-2745.12863.

Kløve, B. et al., 2014: Climate change impacts on groundwater and dependent ecosystems. *Journal of Hydrology*, 518, 250-266.

Knorr, K., L. Jiang and A. Arneth, 2016a: Climate, CO2, and demographic impacts on global wildfire emissions. *Biogeosciences*, 13, 267-282, doi:doi:10.5194/bg-13-267-2016.

Knorr, W., A. Arneth and L. Jiang, 2016b: Demographic controls of future fire risks. *Nature Climate Change*, doi:doi:10.1038/nclimate2999.

Koh, L. P., J. Miettinen, S. C. Liew and J. Ghazoul, 2011: Remotely sensed evidence of tropical peatland conversion to oil palm. *Proceedings of the National Academy of Sciences of the United States of America*, 108 (12), 5127-32, doi:10.1073/pnas.1018776108.

Kohl, M. et al., 2015: Changes in forest production, biomass and carbon: Results from the 2015 UN FAO Global Forest Resource Assessment. *Forest Ecology and Management*, 352, 21-34, doi:10.1016/j.foreco.2015.05.036.

Kolb, T. E. et al., 2016: Observed and anticipated impacts of drought on forest insects and diseases in the United States. *Forest Ecology and Management*, 380, 321-334, doi:10.1016/j.foreco.2016.04.051.

Koninck, R. d., S. p. Bernard and J.-F. o. Bissonnette, 2011: *Borneo transformed : agricultural expansion on the Southeast Asian frontier*. Challenges of the agrarian transition in Southeast Asia, NUS Press, Singapore, ix, 216 p. pp.

Koven, C. D. et al., 2015a: Controls on terrestrial carbon feedbacks by productivity versus turnover in the CMIP5 Earth System Models. *Biogeosciences*, 12 (17), 5211-5228, doi:10.5194/bg-12-5211-2015.

Koven, C. D. et al., 2015b: A simplified, data-constrained approach to estimate the permafrost carbon-climate feedback. *Philosophical Transactions of the Royal Society a-Mathematical Physical and Engineering Sciences*, 373 (2054), doi:10.1098/rsta.2014.0423.

Kraemer, B. et al., 2015: Morphometry and average temperature affect lake stratification responses to climate change. *Geophysical Research Letters*, 42 (12), 4981-4988, doi:10.1002/2015gl064097.

Kraemer, B., T. Mehner and R. Adrian, 2017: Reconciling the opposing effects of warming on phytoplankton biomass in 188 large lakes. *Scientific Reports*, 7, doi:10.1038/s41598-017-11167-3.

Krause, A. et al., 2018: Large uncertainty in carbon uptake potential of land-based climate-change mitigation efforts. *Global Change Biology*, doi:10.1111/gcb.14144.

Kreft, H. and W. Jetz, 2007: Global patterns and determinants of vascular plant diversity. *Proceedings of the National Academy of Sciences of the United States of America*, 104 (14), 5925-5930, doi:10.1073/pnas.0608361104.

Kristensen, T. N. et al., 2015: Low evolutionary potential for egg-to-adult viability in Drosophila melanogaster at high temperatures. *Evolution*, 69 (3), 803-814, doi:10.1111/evo.12617.

Krupnick, G. A., 2013: Conservation of Tropical Plant Biodiversity: What Have We Done, Where Are We Going? *Biotropica*, 45 (6), 693-708, doi:10.1111/btp.12064.

Krushelnycky, P. D. et al., 2013: Climate-associated population declines reverse recovery and threaten future of an iconic high-elevation plant. *Global Change Biology*, 19 (3), 911-922, doi:10.1111/gcb.12111.

Krysanova, V. and F. F. Hattermann, 2017: Intercomparison of climate change impacts in 12 large river basins: overview of methods and summary of results. *Climatic Change*, 141 (3), 363-379, doi:10.1007/s10584-017-1919-y.

Kuemmerlen, M. et al., 2015: An attack on two fronts: predicting how changes in land use and climate affect the distribution of stream macroinvertebrates. *Freshwater Biology*, 60 (7), 1443-1458, doi:10.1111/fwb.12580.

Kuha, J. et al., 2016: Response of boreal lakes to episodic weather-induced events. *Inland Waters*, 6 (4), 523-534, doi:10.5268/iw-6.4.886.

Kusserow, H., 2017: Desertification, resilience, and re-greening in the African Sahel - a matter of the observation period? *Earth System Dynamics*, 8 (4), 1141-1170, doi:10.5194/esd-8-1141-2017.

Laestadius, L. et al., 2011: Mapping opportunities for forest landscape restoration. *Unasylva (English ed.)*, 62 (238), 47-48.

Lagarde, F. et al., 2012: Bushes protect tortoises from lethal overheating in arid areas of Morocco. *Environmental Conservation*, 39 (2), 172-182.

Lal, P. N., T. Mitchell, R. Mechler and S. Hochrainer-Stigler, 2012: National systems for managing the risks from climate extremes and disasters.

Landry, J. S. et al., 2016: Modelling long-term impacts of mountain pine beetle outbreaks on merchantable biomass, ecosystem carbon, albedo, and radiative forcing. *Biogeosciences*, 13 (18), 5277-5295, doi:10.5194/bg-13-5277-2016.

Langdon, J. and J. Lawler, 2015: Assessing the impacts of projected climate change on biodiversity in the protected areas of western North America. *Ecosphere*, 6 (5), doi:10.1890/ES14-00400.1.

Langhans, S. D. et al., 2019: - Combining eight research areas to foster the uptake of ecosystem-based management in fresh waters. - 0 (- 0).

Lapola, D. M., M. D. Oyama and C. A. Nobre, 2009: Exploring the range of climate biome projections for tropical South America: The role of CO $_2$ fertilization and seasonality: FUTURE BIOME DISTRIBUTION IN SOUTH AMERICA. *Global Biogeochemical Cycles*, 23 (3), n/a-n/a, doi:10.1029/2008GB003357.

Lasslop, G. and S. Kloster, 2017: Human impact on wildfires varies between regions and with vegetation productivity. *Environmental Research Letters*, 12 (11), doi:10.1088/1748-9326/aa8c82.

Laurance, W. F. et al., 2012: Averting biodiversity collapse in tropical forest protected areas. *Nature*, 489 (7415), 290-294, doi:10.1038/nature11318.

Lavergne, S., N. Mouquet, W. Thuiller and O. Ronce, 2010: Biodiversity and Climate Change: Integrating Evolutionary and Ecological Responses of Species and Communities. In: Annual Review of Ecology, Evolution, and Systematics, Vol 41 [Futuyma, D. J., H. B. Shafer and D. Simberloff (eds.)]. Annual Reviews, Palo Alto, 41, 321-350.

Le Page, Y. et al., 2017: Synergy between land use and climate change increases future fire risk in Amazon forests. *Earth System Dynamics*, 8 (4), 1237-1246, doi:10.5194/esd-8-1237-2017.

le Polain de Waroux, Y. and E. F. Lambin, 2012: Monitoring degradation in arid and semi-arid forests and woodlands: The case of the argan woodlands (Morocco). *Applied Geography*, 32 (2), 777-786, doi:10.1016/j.apgeog.2011.08.005.

Le Quéré, C. et al., 2018: Global Carbon Budget 2018. *Earth Syst. Sci. Data*, 10 (4), 2141-2194, doi:10.5194/essd-10-2141-2018.

Lehikoinen, P. et al., 2019: Protected areas act as a buffer against detrimental effects of climate change-Evidence from large-scale, long-term abundance data. *Global Change Biology*, 25 (1), 304-313, doi:10.1111/gcb.14461.

Lehman, C. E. R. and C. L. Parr, 2016: Tropical grassy biomes: linking ecology, human use and conservation. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 371 (1703), doi:20160329 10.1098/rstb.2016.0329.

Lehmann, C. E. R. et al., 2014: Savanna vegetation-fire-climate relationships differ among continents. *Science*, 343 (6170), 548-552.

Lehner, B. and P. Doll, 2004: Development and validation of a global database of lakes, reservoirs and wetlands. *Journal of Hydrology*, 296 (1-4), 1-22, doi:10.1016/j.jhydrol.2004.03.028.

Lehner, B., K. Verdin and A. Jarvis, 2008: New global hydrography derived from spaceborne elevation data. *EOS, Transactions American Geophysical Union*, 89 (10), 93–94, doi:10.1029/2008EO100001.

Lehtonen, I. et al., 2016: Risk of large-scale fires in boreal forests of Finland under changing climate. *Natural Hazards and Earth System Sciences*, 16 (1), 239-253, doi:10.5194/nhess-16-239-2016.

Leifeld, J. and L. Menichetti, 2018: The underappreciated potential of peatlands in global climate change mitigation strategies. *Nature Communications*, 9 (1), 1071.

Lejeune, Q. et al., 2018: Historical deforestation locally increased the intensity of hot days in northern mid-latitudes. *Nature Climate Change*, 8 (5), 386-+, doi:10.1038/s41558-018-0131-z.

Lemordant, L. and P. Gentine, 2019: Vegetation Response to Rising CO2 Impacts Extreme Temperatures. *Geophysical Research Letters*, 46 (3), 1383-1392, doi:10.1029/2018gl080238.

Lenoir, J. and J. C. Svenning, 2015: Climate-related range shifts–a global multidimensional synthesis and new research directions. *Ecography*, 38 (1), 15-28, doi:https://doi.org/10.1111/ecog.00967.

Lenton, T. M. et al., 2008: Tipping elements in the Earth's climate system. *Proceedings of the National Academy of Sciences of the United States of America*, 105 (6), 1786-1793, doi:10.1073/pnas.0705414105.

Leonard, 2014: A compound event framework for understanding extreme impacts. *Climate Change*, 5, 113-128.

Lewis, S. L. et al., 2011: The 2010 Amazon Drought. *Science*, 331 (6017), 554-554, doi:10.1126/science.1200807.

Lewis, S. L., D. P. Edwards and D. Galbraith, 2015: Increasing human dominance of tropical forests. *Science*, 349 (6250), 827-832.

Lewis, S. L., C. E. Wheeler, E. T. A. Mitchard and A. Koch, 2019: *Restoring natural forests is the best way to remove atmospheric carbon*. Nature Publishing Group.

Li, D. et al., 2018a: Vulnerability of the global terrestrial ecosystems to climate change. *Global Change Biology*, 24 (9), 4095-4106, doi:10.1111/gcb.14327.

Li, Q., M. Ma, X. Wu and H. Yang, 2018b: Snow Cover and Vegetation-Induced Decrease in Global Albedo From 2002 to 2016. *Journal of Geophysical Research: Atmospheres*, 123 (1), 124–138.

Li, Z. et al., 2015: Potential impacts of climate change on vegetation dynamics in Central Asia. *Journal of Geophysical Research: Atmospheres*, 120 (24), 12345-12356.

Liang, J. J. et al., 2016: Positive biodiversity-productivity relationship predominant in global forests. *Science*, 354 (6309), doi:aaf8957 10.1126/science.aaf8957.

Lilleskov, E. et al., 2019: Is Indonesian peatland loss a cautionary tale for Peru? A two-country comparison of the magnitude and causes of tropical peatland degradation. *Mitigation and Adaptation Strategies for Global Change*, 24 (4), 591-623, doi:10.1007/s11027-018-9790-3.

Lippitt, C. L., D. A. Stow, J. F. O'Leary and J. Franklin, 2013: Influence of short-interval fire occurrence on post-fire recovery of fire-prone shrublands in California, USA. *International Journal of Wildland Fire*, 22 (2), 184-193, doi:10.1071/wf10099.

Lipsett-Moore, G. J., N. H. Wolff and E. T. Game, 2018: Emissions mitigation opportunities for savanna countries from early dry season fire management. *Nature Communications*, 9, doi:10.1038/s41467-018-04687-7.

Lisi, 2015: Watershed geomorphology and snowmelt control stream thermal sensitivity to air temperature. *Geophysical research lett*, 42, 33803388.

Littell, J. S., D. McKenzie, H. Y. Wan and S. A. Cushman, 2018: Climate Change and Future Wildfire in the Western United States: An Ecological Approach to Nonstationarity. *Earth's Future*, 6 (8), 1097-1111, doi:10.1029/2018EF000878.

Littlefield, C. E., M. Krosby, J. L. Michalak and J. J. Lawler, 2019: Connectivity for species on the move: supporting climate-driven range shifts. *Frontiers in Ecology and the Environment*, 17 (5), 270-278.

Liu, B. Y. et al., 2019a: Using Information Theory to Evaluate Directional Precipitation Interactions Over the West Sahel Region in Observations and Models. *Journal of Geophysical Research: Atmospheres*, 124 (3), 1463-1473, doi:10.1029/2018JD029160.

Liu, D., 2016: Water supply: China's sponge cities to soak up rainwater. *Nature*, 537 (7620), 307.

Liu, H. et al., 2012: Overcoming extreme weather challenges: Successful but variable assisted colonization of wild orchids in southwestern China. *Biological Conservation*, 150 (1), 68-75, doi:10.1016/j.biocon.2012.02.018.

Liu, Y. et al., 2019b: Field-experiment constraints on the enhancement of the terrestrial carbon sink by CO2 fertilization. *Nature Geoscience*, doi:10.1038/s41561-019-0436-1.

Liu, Y. J. et al., 2017: Do invasive alien plants benefit more from global environmental change than native plants? *Global Change Biology*, 23 (8), 3363-3370, doi:10.1111/gcb.13579.

Liu, Y. Y. et al., 2015: Recent reversal in loss of global terrestrial biomass. *Nature Climate Change*, 5 (5), 470.

Liu, Z. H., A. P. Ballantyne and L. A. Cooper, 2018: Increases in Land Surface Temperature in Response to Fire in Siberian Boreal Forests and Their Attribution to Biophysical Processes. *Geophysical Research Letters*, 45 (13), 6485-6494, doi:10.1029/2018gl078283.

Liu, Z. H., A. P. Ballantyne and L. A. Cooper, 2019c: Biophysical feedback of global forest fires on surface temperature. *Nature Communications*, 10, doi:10.1038/s41467-018-08237-z.

Lloret, F. and T. Kitzberger, 2018: Historical and event-based bioclimatic suitability predicts regional forest vulnerability to compound effects of severe drought and bark beetle infestation. *Global Change Biology*, 24 (5), 1952-1964, doi:10.1111/gcb.14039.

Locatelli, B. et al., 2015: Tropical reforestation and climate change: beyond carbon. *Restoration Ecology*, 23 (4), 337-343.

Longo, M. et al., 2016: Aboveground biomass variability across intact and degraded forests in the Brazilian Amazon. *Global Biogeochemical Cycles*, 30 (11), 1639-1660, doi:10.1002/2016GB005465.

López-i-Gelats, F. et al., 2015: Adaptation Strategies of Andean Pastoralist Households to Both Climate and Non-Climate Changes. *Human Ecology*, 43 (2), 267-282.

López-i-Gelats, F., E. D. G. Fraser, J. F. Morton and M. G. Rivera-Ferre, 2016: What drives the vulnerability of pastoralists to global environmental change? A qualitative meta-analysis. *Global Environmental Change*, 39, 258-274, doi:https://doi.org/10.1016/j.gloenvcha.2016.05.011.

Loranty, M. M. et al., 2014: Vegetation controls on northern high latitude snow-albedo feedback: observations and CMIP5 model simulations. *Global Change Biology*, 20 (2), 594-606, doi:10.1111/gcb.12391.

Loranty, M. M. and S. J. Goetz, 2012: Shrub expansion and climate feedbacks in Arctic tundra. *Environmental Research Letters*, 7 (1), 011005.

Lu, F. et al., 2018: Effects of national ecological restoration projects on carbon sequestration in China from 2001 to 2010. *Proceedings of the National Academy of Sciences of the United States of America*, 115 (16), 4039-4044, doi:10.1073/pnas.1700294115.

Lubetkin, K., A. Westerling and L. Kueppers, 2017: Climate and landscape drive the pace and pattern of conifer encroachment into subalpine meadows. *Ecological Applications*, 27 (6), 1876-1887, doi:10.1002/eap.1574.

Lunt, I. D. et al., 2013: Using assisted colonisation to conserve biodiversity and restore ecosystem function under climate change. *Biological Conservation*, 157, 172-177, doi:10.1016/j.biocon.2012.08.034.

Lyra, A. D., S. C. Chou and G. D. Sampaio, 2016: Sensitivity of the Amazon biome to high resolution climate change projections. *Acta Amazonica*, 46 (2), 175-187, doi:10.1590/1809-4392201502225.

Macfarlane, W. W., J. A. Logan and W. R. Kern, 2013: An innovative aerial assessment of Greater Yellowstone Ecosystem mountain pine beetle-caused whitebark pine mortality. *Ecological Applications*, 23 (2), 421-437, doi:10.1890/11-1982.1.

Macgregor, N. A. and N. van Dijk, 2014: Adaptation in practice: how managers of nature conservation areas in eastern England are responding to climate change. *Environmental Management*, 54 (4), 700-719.

Mack, M. C. et al., 2011: Carbon loss from an unprecedented Arctic tundra wildfire. *Nature*, 475 (7357), 489.

Mackey, B. et al., 2013: Untangling the confusion around land carbon science and climate change mitigation policy. *Nature Climate Change*, 3 (6), 552.

MacLean, H. J. et al., 2019: Evolution and plasticity of thermal performance: an analysis of variation in thermal tolerance and fitness in 22 Drosophila species. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 374 (1778), 10, doi:10.1098/rstb.2018.0548.

Maes, D. et al., 2019: The potential of species distribution modelling for reintroduction projects: the case study of the Chequered Skipper in England. *Journal of Insect Conservation*, 23 (2), 419-431, doi:10.1007/s10841-019-00154-w.

Maino, J. L. et al., 2016: Mechanistic models for predicting insect responses to climate change. *Current Opinion in Insect Science*, 17, 81-86, doi:10.1016/j.cois.2016.07.006.

Malhi, Y. et al., 2009: Exploring the likelihood and mechanism of a climate-change-induced dieback of the Amazon rainforest. *Proceedings of the National Academy of Sciences*, 106 (49), 20610-20615, doi:10.1073/pnas.0804619106.

Malhi, Y. et al., 2016: Megafauna and ecosystem function from the Pleistocene to the Anthropocene. *Proceedings of the National Academy of Sciences*, 113 (4), 838-846.

Malhi, Y. et al., 2014: Tropical forests in the Anthropocene. *Annual Review of Environment and Resources*, 39.

Manea, A. and M. R. Leishman, 2019: The resprouting response of co-occurring temperate woody plant and grass species to elevated CO2 : An insight into woody plant encroachment of grasslands. *Austral Ecology*, 44 (5), 917-926, doi:10.1111/aec.12760.

Mann, M. L. et al., 2016: Incorporating Anthropogenic Influences into Fire Probability Models: Effects of Human Activity and Climate Change on Fire Activity in California. *PLOS ONE*, 11 (4), e0153589, doi:10.1371/journal.pone.0153589.

Mansuy, N. et al., 2019: Contrasting human influences and macro-environmental factors on fire activity inside and outside protected areas of North America. *Environmental Research Letters*, 14 (6), doi:10.1088/1748-9326/ab1bc5.

Mao, J. F. et al., 2016: Human-induced greening of the northern extratropical land surface. *Nature Climate Change*, 6 (10), 959-+, doi:10.1038/nclimate3056.

Marcé, R. et al., 2019: Emissions from dry inland waters are a blind spot in the global carbon cycle. *Earth-Science Reviews*, 188, 240-248, doi:10.1016/j.earscirev.2018.11.012.

Marengo, J. A. et al., 2018: Changes in Climate and Land Use Over the Amazon Region: Current and Future Variability and Trends. *Frontiers in Earth Science*, 6, doi:10.3389/feart.2018.00228.

Mariani, M. et al., 2019: Climate change reduces resilience to fire in subalpine rainforests. *Global Change Biology*, 25 (6), 2030-2042, doi:10.1111/gcb.14609.

Mariani, M. et al., 2018: Climate Change Amplifications of Climate-Fire Teleconnections in the Southern Hemisphere. *Geophysical Research Letters*, 45 (10), 5071-5081, doi:10.1029/2018gl078294.

Marimon, B. S. et al., 2014: Disequilibrium and hyperdynamic tree turnover at the forest–cerrado transition zone in southern Amazonia. *Plant Ecology & Diversity*, 7 (1-2), 281-292.

Marini, L. et al., 2017: Climate drivers of bark beetle outbreak dynamics in Norway spruce forests. *Ecography*, 40 (12), 1426-1435, doi:10.1111/ecog.02769.

Mark, B. G. et al., 2017: Glacier loss and hydro-social risks in the Peruvian Andes. *Global and Planetary Change*, 159, 61-76, doi:10.1016/j.gloplacha.2017.10.003.

Markovic, D. et al., 2014: Europe's freshwater biodiversity under climate change: distribution shifts and conservation needs. *Diversity and Distributions*, 20 (9), 1097-1107, doi:10.1111/ddi.12232.

Martinez-Vilalta, J. and F. Lloret, 2016: Drought-induced vegetation shifts in terrestrial ecosystems: The key role of regeneration dynamics. *Global and Planetary Change*, 144, 94-108, doi:10.1016/j.gloplacha.2016.07.009.

Mason, S. C. et al., 2015: Geographical range margins of many taxonomic groups continue to shift polewards. *Biological Journal of the Linnean Society*, 115 (3), 586-597.

Masubelele, M. L., M. T. Hoffman and W. J. Bond, 2015a: Biome stability and long-term vegetation change in the semi-arid, south-eastern interior of South Africa: a synthesis of repeat photo-monitoring studies. *South African Journal of Botany*, 101, 139-147.

Masubelele, M. L., M. T. Hoffman and W. J. Bond, 2015b: A repeat photograph analysis of long-term vegetation change in semi-arid South Africa in response to land use and climate. *Journal of Vegetation Science*, 26 (5), 1013-1023.

Maurin, O. et al., 2014: Savanna fire and the origins of the ‘underground forests’ of Africa. *New Phytologist*, 204 (1), 201-214.

Mawdsley, J. R., R. O’malley and D. S. Ojima, 2009: A review of climate-change adaptation strategies for wildlife management and biodiversity conservation. *Conservation Biology*, 23 (5), 1080-1089.

Maxwell, S. L. et al., 2019: Conservation implications of ecological responses to extreme weather and climate events. *Diversity and Distributions*, 25 (4), 613-625, doi:10.1111/ddi.12878.

McDowell, N. et al., 2018: Drivers and mechanisms of tree mortality in moist tropical forests. *New Phytologist*, 219 (3), 851-869.

McDowell, N. G. et al., 2016: Multi-scale predictions of massive conifer mortality due to chronic temperature rise. *Nature Climate Change*, 6 (3), 295-300, doi:10.1038/nclimate2873.

McGuire, A. D. et al., 2018: Assessing historical and projected carbon balance of Alaska: A synthesis of results and policy/management implications. *Ecological Applications*, 28 (6), 1396-1412, doi:10.1002/eap.1768.

McIntyre, P. J. et al., 2015: Twentieth-century shifts in forest structure in California: Denser forests, smaller trees, and increased dominance of oaks. *Proceedings of the National Academy of Sciences*, 112 (5), 1458-1463, doi:10.1073/pnas.1410186112.

McNicol, I. M., C. M. Ryan and E. T. Mitchard, 2018: Carbon losses from deforestation and widespread degradation offset by extensive growth in African woodlands. *Nature Communications*, 9 (1), 3045.

Meakem, V. et al., 2018: Role of tree size in moist tropical forest carbon cycling and water deficit responses. *New Phytologist*, 219 (3), 947-958.

Mechler, R. and L. M. Bouwer, 2015: Understanding trends and projections of disaster losses and climate change: is vulnerability the missing link? *Climatic Change*, 133 (1), 23-35.

Mehran, A. et al., 2017: Compounding Impacts of Human-Induced Water Stress and Climate Change on Water Availability. *Scientific Reports*, 7, doi:10.1038/s41598-017-06765-0.

Mekonnen, M. M. and A. Y. Hoekstra, 2016: Four billion people facing severe water scarcity. *Science Advances*, 2 (2), doi:10.1126/sciadv.1500323.

Mekonnen, Z. A., W. J. Riley and R. F. Grant, 2018: 21st century tundra shrubification could enhance net carbon uptake of North America Arctic tundra under an RCP8.5 climate trajectory. *Environmental Research Letters*, 13 (5), doi:10.1088/1748-9326/aabf28.

Melillo, J. M. et al., 2016: Protected areas' role in climate-change mitigation. *AMBIO*, 45 (2), 133-145, doi:10.1007/s13280-015-0693-1.

Meller, L. et al., 2015: Balance between climate change mitigation benefits and land use impacts of bioenergy: conservation implications for European birds. *GCB Bioenergy*, 7 (4), 741-751, doi:10.1111/gcbb.12178.

Merriam, E., R. Fernandez, J. Petty and N. Zegre, 2017: Can brook trout survive climate change in large rivers? If it rains. *Science of the Total Environment*, 607, 1225-1236, doi:10.1016/j.scitotenv.2017.07.049.

Messager, M. L. et al., 2016: Estimating the volume and age of water stored in global lakes using a geo-statistical approach. *Nature Communications*, 7, 13603, doi:10.1038/ncomms13603 https://www.nature.com/articles/ncomms13603#supplementary-information.

Mezei, P. et al., 2017: Storms, temperature maxima and the Eurasian spruce bark beetle Ips typographus-An infernal trio in Norway spruce forests of the Central European High Tatra Mountains. *Agricultural and Forest Meteorology*, 242, 85-95, doi:10.1016/j.agrformet.2017.04.004.

Micale, V., B. Tonkonogy and F. Mazza, 2018: *Understanding and Increasing Finance for Climate Adaptation in Developing Countries*. [Available at: https://climatepolicyinitiative.org/wp-content/uploads/2018/12/Understanding-and-Increasing-Finance-for-Climate-Adaptation-in-Developing-Countries-1.pdf].

Michalak, J., J. Lawler, D. Roberts and C. Carroll, 2018: Distribution and protection of climatic refugia in North America. *Conservation Biology*, 32 (6), 1414-1425, doi:10.1111/cobi.13130.

Midgley, G., S. Marais, M. Barnett and K. Wågsæther, 2012: Biodiversity, climate change and sustainable development–harnessing synergies and celebrating successes. *South African National Biodiversity Institute (SANBI), Conservation South Africa (CSA), and Indigo Development and Change*.

Midgley, G. F. and W. J. Bond, 2015: Future of African terrestrial biodiversity and ecosystems under anthropogenic climate change. *Nature Climate Change*, 5 (9), 823-829, doi:10.1038/nclimate2753.

Miettinen, J. et al., 2017: From carbon sink to carbon source: Extensive peat oxidation in insular Southeast Asia since 1990. *Environmental Research Letters*, 12 (2), doi:10.1088/1748-9326/aa5b6f.

Miles, M. W., V. V. Miles and I. Esau, 2019: Varying climate response across the tundra, forest-tundra and boreal forest biomes in northern West Siberia. *Environmental Research Letters*, 14 (7), doi:10.1088/1748-9326/ab2364.

Milly, P. C., K. A. Dunne and A. V. Vecchia, 2005: Global pattern of trends in streamflow and water availability in a changing climate. *Nature*, 438 (7066), 347-50, doi:10.1038/nature04312.

Minteer, B. A. and J. P. Collins, 2010: Move it or lose it? The ecological ethics of relocating species under climate change. *Ecological Applications*, 20 (7), 1801-1804, doi:10.1890/10-0318.1.

Miralles, D. G., P. Gentine, S. I. Seneviratne and A. J. Teuling, 2019: Land-atmospheric feedbacks during droughts and heatwaves: state of the science and current challenges. *Annals of the New York Academy of Sciences*, 1436 (1), 19-35, doi:10.1111/nyas.13912.

Mitchard, E. T. A., 2018: The tropical forest carbon cycle and climate change. *Nature*, 559 (7715), 527-534, doi:10.1038/s41586-018-0300-2.

Mitchard, E. T. A. et al., 2014: Markedly divergent estimates of Amazon forest carbon density from ground plots and satellites. *Global Ecology and Biogeography*, 23 (8), 935-946, doi:10.1111/geb.12168.

Mod, H. K. and M. Luoto, 2016: Arctic shrubification mediates the impacts of warming climate on changes to tundra vegetation. *Environmental Research Letters*, 11 (12), doi:124028 10.1088/1748-9326/11/12/124028.

Moftakhari, H. R. et al., 2017: Compounding effects of sea level rise and fluvial flooding. *Proceedings of the National Academy of Sciences of the United States of America*, 114 (37), 9785-9790, doi:10.1073/pnas.1620325114.

Mokany, K. et al., 2016: Integrating modelling of biodiversity composition and ecosystem function. *Oikos*, 125 (1), 10-19, doi:10.1111/oik.02792.

Moloney, K. A. et al., 2019: Increased fire risk in Mojave and Sonoran shrublands due to exotic species and extreme rainfall events. *Ecosphere*, 10 (2), doi:10.1002/ecs2.2592.

Moncrieff, G. R., S. Chamaille-Jammes and W. J. Bond, 2014: Modelling direct and indirect impacts of browser consumption on woody plant growth: moving beyond biomass. *Oikos*, 123 (3), 315-322, doi:10.1111/j.1600-0706.2013.00904.x.

Moncrieff, G. R. et al., 2016: The future distribution of the savannah biome: model-based and biogeographic contingency. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 371 (1703), doi:20150311 10.1098/rstb.2015.0311.

Moncrieff, G. R., S. Scheiter, J. A. Slingsby and S. I. Higgins, 2015: Understanding global change impacts on South African biomes using Dynamic Vegetation Models. *South African Journal of Botany*, 101, 16-23, doi:10.1016/j.sajb.2015.02.004.

Moomaw, W. R. et al., 2018: Wetlands In a Changing Climate: Science, Policy and Management. *Wetlands*, 38 (2), 183-205, doi:10.1007/s13157-018-1023-8.

Morelli, T. L. et al., 2016: Managing climate change refugia for climate adaptation. *PLOS ONE*, 11 (8), e0159909.

Morgan, J. A. et al., 2004: Water relations in grassland and desert ecosystems exposed to elevated atmospheric CO 2. *Oecologia*, 140 (1), 11-25.

Morin, X. and W. Thuiller, 2009: Comparing niche- and process-based models to reduce prediction uncertainty in species range shifts under climate change. *Ecology*, 90 (5), 1301-1313, doi:10.1890/08-0134.1.

Moritz, M. A. et al., 2012: Climate change and disruptions to global fire activity. *Ecosphere*, 3 (6), doi:10.1890/es11-00345.1.

Morueta-Holme, N. et al., 2015: Strong upslope shifts in Chimborazo's vegetation over two centuries since Humboldt. *Proceedings of the National Academy of Sciences of the United States of America*, 112 (41), 12741-12745, doi:10.1073/pnas.1509938112.

Mote, P. W. et al., 2018: Dramatic declines in snowpack in the western US. *Npj Climate and Atmospheric Science*, 1 (1), 2.

Mouillot, F. and C. B. Field, 2005: Fire history and the global carbon budget: a 1 degrees x 1 degrees fire history reconstruction for the 20th century. *Global Change Biology*, 11 (3), 398-420, doi:10.1111/j.1365-2486.2005.00920.x.

Muhlfeld, C. et al., 2014: Invasive hybridization in a threatened species is accelerated by climate change. *Nature Climate Change*, 4 (7), 620-624, doi:10.1038/nclimate2252.

Munson, S. M., A. L. Long, C. S. A. Wallace and R. H. Webb, 2016: Cumulative drought and land-use impacts on perennial vegetation across a North American dryland region. *Applied Vegetation Science*, 19 (3), 430-441, doi:10.1111/avsc.12228.

Munson, S. M. et al., 2012: Forecasting climate change impacts to plant community composition in the Sonoran Desert region. *Global Change Biology*, 18 (3), 1083-1095, doi:10.1111/j.1365-2486.2011.02598.x.

Murdiyarso, D. et al., 2015: The potential of Indonesian mangrove forests for global climate change mitigation. *Nature Climate Change*, 5 (12), 1089-1092, doi:10.1038/nclimate2734.

Murphy, B. P., A. N. Andersen and C. L. Parr, 2016: The underestimated biodiversity of tropical grassy biomes. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371 (1703), 20150319.

Murphy, B. P. et al., 2019: Biomass consumption by surface fires across Earth's most fire prone continent. *Global Change Biology*, 25 (1), 254-268, doi:10.1111/gcb.14460.

Myers-Smith, I. H. et al., 2015: Climate sensitivity of shrub growth across the tundra biome. *Nature Climate Change*, 5 (9), 887.

Myers-Smith, I. H. et al., 2011: Shrub expansion in tundra ecosystems: dynamics, impacts and research priorities. *Environmental Research Letters*, 6 (4), 045509, doi:10.1088/1748-9326/6/4/045509.

Myers-Smith, I. H. et al., 2019a: Eighteen years of ecological monitoring reveals multiple lines of evidence for tundra vegetation change. *Ecological Monographs*, 89 (2), e01351.

Myers-Smith, I. H., H. J. D. Thomas and A. D. Bjorkman, 2019b: Plant traits inform predictions of tundra responses to global change. *New Phytologist*, 221 (4), 1742-1748.

Nackley, L. L. et al., 2018: CO2 enrichment does not entirely ameliorate Vachellia karroo drought inhibition: A missing mechanism explaining savanna bush encroachment. *Environmental and Experimental Botany*, 155, 98–106.

Nalau, J., S. Becken and B. Mackey, 2018: Ecosystem-based Adaptation: A review of the constraints. *Environmental Science & Policy*, 89, 357-364.

Natali, S. M., E. A. G. Schuur and R. L. Rubin, 2012: Increased plant productivity in Alaskan tundra as a result of experimental warming of soil and permafrost. *Journal of Ecology*, 100 (2), 488-498, doi:10.1111/j.1365-2745.2011.01925.x.

Nath, T. K., 2016: *Monoculture farming : global practices, ecological impact and benefits/drawbacks*. Global agriculture developments, Nova Science Publishers, Hauppauge, New York, xi, 184 pages pp.

Naudts, K. et al., 2016: Europe's forest management did not mitigate climate warming. *Science*, 351 (6273), 597-600, doi:10.1126/science.aad7270.

Nemens, D. G., J. M. Varner, K. R. Kidd and B. Wing, 2018: Do repeated wildfires promote restoration of oak woodlands in mixed-conifer landscapes? *Forest Ecology and Management*, 427, 143-151, doi:10.1016/j.foreco.2018.05.023.

Nepstad, D. C., C. M. Stickler, B. Soares and F. Merry, 2008: Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 363 (1498), 1737-1746, doi:10.1098/rstb.2007.0036.

Nesper, M. et al., 2019: Simplification of shade tree diversity reduces nutrient cycling resilience in coffee agroforestry. *Journal of Applied Ecology*, 56 (1), 119-131.

Newbold, T. et al., 2014: A global model of the response of tropical and sub-tropical forest biodiversity to anthropogenic pressures. *Proceedings of the Royal Society B-Biological Sciences*, 281 (1792), doi:20141371 10.1098/rspb.2014.1371.

Newson, S. E. et al., 2014: Can site and landscape-scale environmental attributes buffer bird populations against weather events? *Ecography*, 37 (9), 872-882, doi:10.1111/ecog.00575.

Ngai, R. et al., 2017: Working with natural processes - evidence directory. Appendix 2: literature review. Environment Agency, Bristol, UK.

Nishina, K. et al., 2014: Quantifying uncertainties in soil carbon responses to changes in global mean temperature and precipitation. *Earth System Dynamics*, 5 (1), 197-209, doi:10.5194/esd-5-197-2014.

Nishina, K. et al., 2015: Decomposing uncertainties in the future terrestrial carbon budget associated with emission scenarios, climate projections, and ecosystem simulations using the ISI-MIP results. *Earth System Dynamics*, 6 (2), 435-445, doi:10.5194/esd-6-435-2015.

Nobre, C. A. et al., 2016: Land-use and climate change risks in the Amazon and the need of a novel sustainable development paradigm. *Proceedings of the National Academy of Sciences of the United States of America*, 113 (39), 10759-10768, doi:10.1073/pnas.1605516113.

Nolan, C. et al., 2018: Past and future global transformation of terrestrial ecosystems under climate change. *Science*, 361 (6405), 920-923, doi:10.1126/science.aan5360.

Norton, B. A. et al., 2015: Planning for cooler cities: A framework to prioritise green infrastructure to mitigate high temperatures in urban landscapes. *Landscape and Urban Planning*, 134, 127-138.

O'Connell, C. S., L. Ruan and W. L. Silver, 2018: Drought drives rapid shifts in tropical rainforest soil biogeochemistry and greenhouse gas emissions. *Nat Commun*, 9 (1), 1348, doi:10.1038/s41467-018-03352-3.

O'Connor, T. G., J. R. Puttick and M. T. Hoffman, 2014: Bush encroachment in southern Africa: changes and causes. *African Journal of Range & Forage Science*, 31 (2), 67–88.

O'Donnell, F. C., W. T. Flatley, A. E. Springer and P. Z. Fule, 2018: Forest restoration as a strategy to mitigate climate impacts on wildfire, vegetation, and water in semiarid forests. *Ecological Applications*, 28 (6), 1459-1472, doi:10.1002/eap.1746.

O'Leary, D. S. et al., 2016: A NEW METHOD COMPARING SNOWMELT TIMING WITH ANNUAL AREA BURNED. *Fire Ecology*, 12 (1), 41-51, doi:10.4996/fireecology.1201041.

O'Reilly, C. et al., 2015: Rapid and highly variable warming of lake surface waters around the globe. *Geophysical Research Letters*, 42 (24), 10773-10781, doi:10.1002/2015gl066235.

Okland, B. et al., 2019: Range expansion of the small spruce bark beetle Ips amitinus: a newcomer in northern Europe. *Agricultural and Forest Entomology*, 21 (3), 286-298, doi:10.1111/afe.12331.

Olivares, E. A. O. et al., 2019: Climate Change, Land Use/Land Cover Change, and Population Growth as Drivers of Groundwater Depletion in the Central Valleys, Oaxaca, Mexico. *Remote Sensing*, 11 (11), doi:10.3390/rs11111290.

Oliver, E. C. J., 2019: Mean warming not variability drives marine heatwave trends. *Climate Dynamics*, 53 (3-4), 1653-1659, doi:10.1007/s00382-019-04707-2.

Oliver, T. H. et al., 2017: Large extents of intensive land use limit community reorganization during climate warming. *Global Change Biology*, 23 (6), 2272-2283.

Oliver, T. H. et al., 2015a: Biodiversity and resilience of ecosystem functions. *Trends in Ecology & Evolution*, 30 (11), 673-684.

Oliver, T. H. et al., 2015b: Interacting effects of climate change and habitat fragmentation on drought-sensitive butterflies. *Nature Climate Change*, 5 (10), 941.

Olofsson, J. and E. Post, 2018: Effects of large herbivores on tundra vegetation in a changing climate, and implications for rewilding. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 373 (1761), 20170437.

Olson, D. M. et al., 2001: Terrestrial ecoregions of the worlds: A new map of life on Earth. *BioScience*, 51 (11), 933-938, doi:10.1641/0006-3568(2001)051[0933:teotwa]2.0.co;2.

Ondei, S., L. D. Prior, T. Vigilante and D. M. J. S. Bowman, 2017: Fire and cattle disturbance affects vegetation structure and rain forest expansion into savanna in the Australian monsoon tropics. *Journal of Biogeography*, 44 (10), 2331-2342.

Ordonez, A., J. W. Williams and J. C. Svenning, 2016: Mapping climatic mechanisms likely to favour the emergence of novel communities. *Nature Climate Change*, 6 (12), 1104-1109, doi:10.1038/nclimate3127.

Orlove, B. et al., 2019: Framing climate change in frontline communities: anthropological insights on how mountain dwellers in the USA, Peru, and Italy adapt to glacier retreat. *Regional Environmental Change*, 19 (5), 1295-1309.

Ortega, H., 2019: Methane emissions from contrasting urban freshwaters: Rates, drivers, and a whole-city footprint. *Global Change Biology*.

Ostberg, S. et al., 2018: The Biosphere Under Potential Paris Outcomes. *Earth's Future*, 6 (1), 23-39, doi:10.1002/2017EF000628.

Ostberg, S., W. Lucht, S. Schaphoff and D. Gerten, 2013: Critical impacts of global warming on land ecosystems. *Earth System Dynamics*, 4 (2), 347-357, doi:10.5194/esd-4-347-2013.

Osuri, A. M. et al., 2016: Contrasting effects of defaunation on aboveground carbon storage across the global tropics. *Nature Communications*, 7, 11351.

Otto, F. E. L. et al., 2018: Anthropogenic influence on the drivers of the Western Cape drought 2015–2017. *Environmental Research Letters*, 13 (12), 124010, doi:10.1088/1748-9326/aae9f9.

Oyama, M. D. and C. A. Nobre, 2003: A new climate-vegetation equilibrium state for tropical South America. *Geophysical Research Letters*, 30 (23), doi:10.1029/2003gl018600.

Ozkundakci, D. et al., 2016: Winter severity determines functional trait composition of phytoplankton in seasonally ice-covered lakes. *Global Change Biology*, 22 (1), 284-298, doi:10.1111/gcb.13085.

Pacifici, M. et al., 2015: Assessing species vulnerability to climate change. *Nature Climate Change*, 5 (3), 215-225, doi:10.1038/NCLIMATE2448.

Pacifici, M. et al., 2017: Species' traits influenced their response to recent climate change. *Nature Climate Change*, 7 (3), 205-208, doi:10.1038/nclimate3223.

Page, S. E. and A. Hooijer, 2016: In the line of fire: the peatlands of Southeast Asia. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, 371 (1696), 20150176, doi:10.1098/rstb.2015.0176.

Page, S. E., J. O. Rieley and C. J. Banks, 2011: Global and regional importance of the tropical peatland carbon pool. *Global Change Biology*, 17 (2), 798-818, doi:10.1111/j.1365-2486.2010.02279.x.

PaiMazumder, D. and J. M. Done, 2016: Potential predictability sources of the 2012 US drought in observations and a regional model ensemble. *Journal of Geophysical Research-Atmospheres*, 121 (21), 12581-12592, doi:10.1002/2016jd025322.

Pan, Y. et al., 2011: A large and persistent carbon sink in the world's forests. *Science*, 333 (6045), 988-993.

Papanikolaou, A. D., I. Kühn, M. Frenzel and O. Schweiger, 2017: Semi-natural habitats mitigate the effects of temperature rise on wild bees. *Journal of Applied Ecology*, 54 (2), 527-536.

Park, C.-E. et al., 2018: Keeping global warming within 1.5 °C constrains emergence of aridification. *Nature Climate Change*, 8 (1), 70, doi:10.1038/s41558-017-0034-4.

Park, C., S. Jeong, C. Ho and J. Kim, 2015: Regional Variations in Potential Plant Habitat Changes in Response to Multiple Global Warming Scenarios. *Journal of Climate*, 28 (7), 2884-2899, doi:10.1175/JCLI-D-13-00753.1.

Park, I. W. and G. D. Jenerette, 2019: Causes and feedbacks to widespread grass invasion into chaparral shrub dominated landscapes. *Landscape Ecology*, 34 (3), 459-471.

Parks, S., L. Holsinger, C. Miller and M. Parisien, 2018: Analog-based fire regime and vegetation shifts in mountainous regions of the western US. *Ecography*, 41 (6), 910-921, doi:10.1111/ecog.03378.

Parks, S. A., S. Z. Dobrowski, J. D. Shaw and C. Miller, 2019: Living on the edge: trailing edge forests at risk of fire-facilitated conversion to non-forest. *Ecosphere*, 10 (3), doi:10.1002/ecs2.2651.

Parmesan, C., 2006: Ecological and Evolutionary Responses to Recent Climate Change. *Annual Review of Ecology, Evolution, and Systematics*, 37 (1), 637-669, doi:10.1146/annurev.ecolsys.37.091305.110100.

Parmesan, C., 2019: Chapter 3: Range and Abundance Changes. In: Biodiversity and Climate Change: Transforming the Biosphere. Yale University Press.

Parmesan, C. et al., 2013: Beyond climate change attribution in conservation and ecological research. *Ecology Letters*, 16 (s1), 58-71, doi:10.1111/ele.12098.

Parmesan, C. and M. E. Hanley, 2015: Plants and climate change: complexities and surprises. *Annals of Botany*, 116 (6), 849-864, doi:10.1093/aob/mcv169.

Parmesan, C., T. L. Root and M. R. Willig, 2000: Impacts of extreme weather and climate on terrestrial biota. *Bulletin of the American Meteorological Society*, 81 (3), 443-450.

Parmesan, C. and G. Yohe, 2003: A globally coherent fingerprint of climate change impacts across natural systems. *Nature*, 421 (6918), 37-42, doi:10.1038/nature01286.

Parr, C. L. et al., 2014: Tropical grassy biomes: misunderstood, neglected, and under threat. *Trends in Ecology & Evolution*, 29 (4), 205-213.

Pastick, N. J. et al., 2017: Historical and projected trends in landscape drivers affecting carbon dynamics in Alaska. *Ecological Applications*, 27 (5), 1383-1402, doi:10.1002/eap.1538/full.

Patino-Martinez, J., A. Marco, L. Quiñones and L. Hawkes, 2012: A potential tool to mitigate the impacts of climate change to the Caribbean leatherback sea turtle. *Global Change Biology*, 18 (2), 401-411.

Patrut, A. et al., 2018: The demise of the largest and oldest African baobabs. *Nature Plants*, 4 (7), 423-426, doi:10.1038/s41477-018-0170-5.

Pausas, J. G., 2015: Alternative fire-driven vegetation states. *Journal of Vegetation Science*, 26 (1), 4-6.

Paz-Kagan, T. et al., 2017: What mediates tree mortality during drought in the southern Sierra Nevada? *Ecological Applications*, 27 (8), 2443-2457, doi:10.1002/eap.1620.

Pearce-Higgins, J. W., S. M. Eglington, B. Martay and D. E. Chamberlain, 2015: Drivers of climate change impacts on bird communities. *Journal of Animal Ecology*, 84 (4), 943-954, doi:10.1111/1365-2656.12364.

Pearson, R. G. et al., 2013: Shifts in Arctic vegetation and associated feedbacks under climate change. *Nature Climate Change*, 3 (7), 673-677, doi:10.1038/nclimate1858.

Pechlivanidis, I. G. et al., 2017: Analysis of hydrological extremes at different hydro-climatic regimes under present and future conditions. *Climatic Change*, 141 (3), 467-481, doi:10.1007/s10584-016-1723-0.

Pecl, G. T. et al., 2017: Biodiversity redistribution under climate change: Impacts on ecosystems and human well-being. *Science*, 355 (6332), doi:10.1126/science.aai9214.

Pekel, J., A. Cottam, N. Gorelick and A. Belward, 2016: High-resolution mapping of global surface water and its long-term changes. *Nature*, 540 (7633), 418-+, doi:10.1038/nature20584.

Pellegrini, A. F. A. et al., 2018: Fire frequency drives decadal changes in soil carbon and nitrogen and ecosystem productivity. *Nature*, 553 (7687), 194-198, doi:10.1038/nature24668.

Pelletier, J. et al., 2018: Carbon sink despite large deforestation in African tropical dry forests (miombo woodlands). *Environmental Research Letters*, 13 (9), doi:10.1088/1748-9326/aadc9a.

Penuelas, J. et al., 2017: Shifting from a fertilization-dominated to a warming-dominated period. *Nature Ecology & Evolution*, 1 (10), 1438-1445, doi:10.1038/s41559-017-0274-8.

Peñuelas, J. et al., 2013: Human-induced nitrogen–phosphorus imbalances alter natural and managed ecosystems across the globe. *Nature Communications*, 4, 2934, doi:10.1038/ncomms3934.

Peñuelas, J., T. Rutishauser and I. Filella, 2009: Phenology Feedbacks on Climate Change. *Science*, 324 (5929), 887-888, doi:10.1126/science.1173004.

Pereira, H. M. et al., 2010: Scenarios for Global Biodiversity in the 21st Century. *Science*, 330 (6010), 1496-1501, doi:10.1126/science.1196624.

Peres, C. A. et al., 2016: Dispersal limitation induces long-term biomass collapse in overhunted Amazonian forests. *Proceedings of the National Academy of Sciences of the United States of America*, 113 (4), 892-897, doi:10.1073/pnas.1516525113.

Perga, M. et al., 2018: Storm impacts on alpine lakes: Antecedent weather conditions matter more than the event intensity. *Global Change Biology*, 24 (10), 5004-5016, doi:10.1111/gcb.14384.

Perugini, L. et al., 2017: Biophysical effects on temperature and precipitation due to land cover change. *Environmental Research Letters*, 12 (5), doi:10.1088/1748-9326/aa6b3f.

Peterson, E. E. et al., 2013: Modelling dendritic ecological networks in space: an integrated network perspective. *Ecology Letters*, 16 (5), 707-719, doi:10.1111/ele.12084.

Peylin, P. et al., 2013: Global atmospheric carbon budget: results from an ensemble of atmospheric CO2 inversions. *Biogeosciences*, 10 (10), 6699-6720, doi:10.5194/bg-10-6699-2013.

Phoenix, G. K. and J. W. Bjerke, 2016: Arctic browning: extreme events and trends reversing arctic greening. *Global Change Biology*, 22 (9), 2960-2962.

Piao, S. et al., 2014: Evidence for a weakening relationship between interannual temperature variability and northern vegetation activity. *Nature Communications*, 5, 5018.

Piao, S. et al., 2013: Evaluation of terrestrial carbon cycle models for their response to climate variability and to CO2 trends. *Global Change Biology*, 19 (7), 2117-2132, doi:10.1111/gcb.12187.

Piao, S. L. et al., 2015: Detection and attribution of vegetation greening trend in China over the last 30 years. *Global Change Biology*, 21 (4), 1601-1609, doi:10.1111/gcb.12795.

Piccolroaz, S. et al., 2016: Prediction of river water temperature: a comparison between a new family of hybrid models and statistical approaches. *Hydrological Processes*, 30 (21), 3901-3917, doi:10.1002/hyp.10913.

Piccolroaz, S., M. Toffolon, C. Robinson and A. Siviglia, 2018: Exploring and Quantifying River Thermal Response to Heatwaves. *Water*, 10 (8), doi:10.3390/w10081098.

Platts, P. et al., 2014: Conservation implications of omitting narrow-ranging taxa from species distribution models, now and in the future. *Diversity and Distributions*, 20 (11), 1307-1320, doi:10.1111/ddi.12244.

Plaza, C. et al., 2019: Direct observation of permafrost degradation and rapid soil carbon loss in tundra. *Nature Geoscience*, 12 (8), 627-+, doi:10.1038/s41561-019-0387-6.

Polk, M. H., 2016: "They are drying out": Social-ecological consequences of glacier recession on mountain peatlands in Husacaran National Park, Peru. Dissertation. The University of Texas at Austin

Polk, M. H. et al., 2017: Exploring hydrologic connections between tropical mountain wetlands and glacier recession in Peru's Cordillera Blanca. *Applied Geography*, 78, 94-103.

Polley, H. W., H. S. Mayeux, H. B. Johnson and C. R. Tischler, 1997: Atmospheric CO2, soil water, and shrub/grass ratios on rangelands. *Rangeland Ecology & Management/Journal of Range Management Archives*, 50 (3), 278-284.

Poloczanska, E. S. et al., 2013: Global imprint of climate change on marine life. *Nature Climate Change*, 3 (10), 919-925, doi:10.1038/nclimate1958.

Poorter, L. et al., 2015: Diversity enhances carbon storage in tropical forests. *Global Ecology and Biogeography*, 24 (11), 1314-1328, doi:10.1111/geb.12364.

Popp, A. et al., 2014: Land-use protection for climate change mitigation. *Nature Climate Change*, doi:10.1038/nclimate2444.

Post, E. et al., 2013: Ecological consequences of sea-ice decline. *Science*, 341 (6145), 519-524.

Post, E. et al., 2009: Ecological dynamics across the Arctic associated with recent climate change. *Science*, 325 (5946), 1355-1358.

Postigo, J. C., 2013: Adaptation of Andean Herders to Political and Climatic Changes. In: Continuity and Change in Cultural Adaptation to Mountain Environments [Lozny, L. R. (ed.)]. Springer New York, 7, 229-258.

Postigo, J. C., 2019: Multi-temporal adatations to change in the Cetral Andes. In: Climate and Culture [Feola, G., H. Geoghegan and A. Arnall (eds.)]. Cambridge University Press, Cambridge, UK, 117-140.

Potts, A. J., G. R. Moncrieff, W. J. Bond and R. M. Cowling, 2015: An operational framework for biome boundary research with examples from South Africa. *South African Journal of Botany*, 101, 5-15, doi:10.1016/j.sajb.2015.07.002.

Poulter, B. et al., 2010: Net biome production of the Amazon Basin in the 21st century. *Global Change Biology*, 16 (7), 2062-2075, doi:10.1111/j.1365-2486.2009.02064.x.

Pounds, J. et al., 2006: Widespread amphibian extinctions from epidemic disease driven by global warming. *Nature*, 439 (7073), 161-167, doi:10.1038/nature04246.

Pouteau, R., T. W. Giambelluca, C. Ah-Peng and J. Y. Meyer, 2018: Will climate change shift the lower ecotone of tropical montane cloud forests upwards on islands? *Journal of Biogeography*, 45 (6), 1326-1333, doi:10.1111/jbi.13228.

Powell, E. J. et al., 2019: A review of coastal management approaches to support the integration of ecological and human community planning for climate change. *Journal of Coastal Conservation*, 23 (1), 1-18.

Prăvălie, R., 2018: Major perturbations in the Earth's forest ecosystems. Possible implications for global warming. *Earth-Science Reviews*, 185, 544-571, doi:https://doi.org/10.1016/j.earscirev.2018.06.010.

Pritchard, S., 2011: Soil organisms and global climate change. *Plant Pathology*, 60 (1), 82-99.

Prober, S. M. et al., 2019: Shifting the conservation paradigm: a synthesis of options for renovating nature under climate change. *Ecological Monographs*, 89 (1), e01333.

Probert, J. R. et al., 2019: Anthropogenic modifications to fire regimes in the wider Serengeti-Mara ecosystem. *Global Change Biology*, 25, 3406-3423, doi:10.1111/gcb.14711.

Prudhomme, C. et al., 2014: Hydrological droughts in the 21st century, hotspots and uncertainties from a global multimodel ensemble experiment. *Proceedings of the National Academy of Sciences of the United States of America*, 111 (9), 3262-3267, doi:10.1073/pnas.1222473110.

Pugh, T. A. M. et al., 2019: Role of forest regrowth in global carbon sink dynamics. *Proceedings of the National Academy of Sciences of the United States of America*, 116 (10), 4382-4387, doi:10.1073/pnas.1810512116.

Pugh, T. A. M. et al., 2016: Key knowledge and data gaps in modelling the influence of CO2 concentration on the terrestrial carbon sink. *Journal of Plant Physiology*, 203, 3-15, doi:10.1016/j.jplph.2016.05.001.

Pureswaran, D. S., A. Roques and A. Battisti, 2018: Forest Insects and Climate Change. *Current Forestry Reports*, 4 (2), 35-50, doi:10.1007/s40725-018-0075-6.

Puschendorf, R. et al., 2011: Environmental Refuge from Disease-Driven Amphibian Extinction. *Conservation Biology*, 25 (5), 956-964, doi:10.1111/j.1523-1739.2011.01728.x.

Pütz, S. et al., 2014: Long-term carbon loss in fragmented Neotropical forests. *Nature Communications*, 5, 5037, doi:10.1038/ncomms6037 https://www.nature.com/articles/ncomms6037#supplementary-information.

Pyne, M. I. and N. L. Poff, 2017: Vulnerability of stream community composition and function to projected thermal warming and hydrologic change across ecoregions in the western United States. *Global Change Biology*, 23 (1), 77-93, doi:10.1111/gcb.13437.

Qie, L. et al., 2017: Long-term carbon sink in Borneo’s forests halted by drought and vulnerable to edge effects. *Nature Communications*, 8 (1), 1966.

Quesada, B., A. Arneth, E. Robertson and N. de Noblet-Ducoudre, 2018: Potential strong contribution of future anthropogenic land-use and land-cover change to the terrestrial carbon cycle. *Environmental Research Letters*, 13 (6), doi:10.1088/1748-9326/aac4c3.

Racault, M. F. et al., 2017a: Impact of El Nino Variability on Oceanic Phytoplankton. *Frontiers in Marine Science*, 4, doi:10.3389/fmars.2017.00133.

Racault, M. F., S. Sathyendranath, N. Menon and T. Platt, 2017b: Phenological Responses to ENSO in the Global Oceans. *Surveys in Geophysics*, 38 (1), 277-293, doi:10.1007/s10712-016-9391-1.

Radeloff, V. C. et al., 2015: The rise of novelty in ecosystems. *Ecological Applications*, 25 (8), 2051-2068.

Radville, L., M. L. McCormack, E. Post and D. M. Eissenstat, 2016: Root phenology in a changing climate. *Journal of Experimental Botany*, 67 (12), 3617-3628.

Raffa, K. F. et al., 2008: Cross-scale drivers of natural disturbances prone to anthropogenic amplification: The dynamics of bark beetle eruptions. *BioScience*, 58 (6), 501-517, doi:10.1641/b580607.

Raffa, K. F., E. N. Powell and P. A. Townsend, 2013: Temperature-driven range expansion of an irruptive insect heightened by weakly coevolved plant defenses. *Proceedings of the National Academy of Sciences of the United States of America*, 110 (6), 2193-2198, doi:10.1073/pnas.1216666110.

Rahbek, C. et al., 2019: Humboldt’s enigma: What causes global patterns of mountain biodiversity? *Science*, 365 (6458), 1108, doi:10.1126/science.aax0149.

Rammig, A. et al., 2010: Estimating the risk of Amazonian forest dieback. *New Phytologist*, 187 (3), 694-706, doi:10.1111/j.1469-8137.2010.03318.x.

Randerson, J. T. et al., 2006: The impact of boreal forest fire on climate warming. *Science*, 314 (5802), 1130-1132, doi:10.1126/science.1132075.

Rannow, S. et al., 2014: Managing Protected Areas Under Climate Change: Challenges and Priorities. *Environmental Management*, 54 (4), 732-743, doi:10.1007/s00267-014-0271-5.

Rappaport, D. I. et al., 2018: Quantifying long-term changes in carbon stocks and forest structure from Amazon forest degradation. *Environmental Research Letters*, 13 (6), doi:10.1088/1748-9326/aac331.

Rasquinha, D. N. and M. Sankaran, 2016: Modelling biome shifts in the Indian subcontinent under scenarios of future climate change. *Current Science*, 111, 147-156.

Ratnam, J. et al., 2011: When is a ‘forest’ a savanna, and why does it matter? *Global Ecology and Biogeography*, 20 (5), 653-660, doi:10.1111/j.1466-8238.2010.00634.x.

Redmond, M. D., P. J. Weisberg, N. S. Cobb and M. J. Clifford, 2018: Woodland resilience to regional drought: Dominant controls on tree regeneration following overstorey mortality. *Journal of Ecology*, 106 (2), 625-639, doi:10.1111/1365-2745.12880.

Regehr, E. V. et al., 2016: Conservation status of polar bears (Ursus maritimus) in relation to projected sea-ice declines. *Biology Letters*, 12 (12), doi:10.1098/rsbl.2016.0556.

Reich, P. B., S. E. Hobbie and T. D. Lee, 2014: Plant growth enhancement by elevated CO2 eliminated by joint water and nitrogen limitation. *Nature Geoscience*, 7 (12), 920-924, doi:10.1038/ngeo2284.

Reid, A. et al., 2019: Emerging threats and persistent conservation challenges for freshwater biodiversity. *Biological Reviews*, 94 (3), 849-873, doi:10.1111/brv.12480.

Restaino, C. et al., 2019: Forest structure and climate mediate drought-induced tree mortality in forests of the Sierra Nevada, USA. *Ecological Applications*, 29 (4), doi:10.1002/eap.1902.

Retallick, R. W. R., H. McCallum and R. Speare, 2004: Endemic infection of the amphibian chytrid fungus in a frog community post-decline. *Plos Biology*, 2 (11), 1965-1971, doi:10.1371/journal.pbio.0020351.

Reu, B. et al., 2014: Future no-analogue vegetation produced by no-analogue combinations of temperature and insolation. *Global Ecology and Biogeography*, 23 (2), 156-167, doi:10.1111/geb.12110.

Reyes-Fox, M. et al., 2014: Elevated CO 2 further lengthens growing season under warming conditions. *Nature*, 510 (7504), 259.

Richardson, D. M. et al., 2009: Multidimensional evaluation of managed relocation. *Proceedings of the National Academy of Sciences of the United States of America*, 106 (24), 9721-9724, doi:10.1073/pnas.0902327106.

Rigling, A. et al., 2013: Driving factors of a vegetation shift from Scots pine to pubescent oak in dry Alpine forests. *Global Change Biology*, 19 (1), 229-240, doi:10.1111/gcb.12038.

Rinkevich, B., 2019: The Active Reef Restoration Toolbox is a Vehicle for Coral Resilience and Adaptation in a Changing World. *Journal of Marine Science and Engineering*, 7 (7), doi:10.3390/jmse7070201.

Roberts, D. et al., 2012: Exploring ecosystem-based adaptation in Durban, South Africa:“learning-by-doing” at the local government coal face. *Environment and Urbanization*, 24 (1), 167-195.

Roberts, D. and S. O’Donoghue, 2013: Urban environmental challenges and climate change action in Durban, South Africa. *Environment and Urbanization*, 25 (2), 299-319.

Robinne, F. N. et al., 2018: A spatial evaluation of global wildfire-water risks to human and natural systems. *Science of the Total Environment*, 610, 1193-1206, doi:10.1016/j.scitotenv.2017.08.112.

Rogelj, J. et al., 2018: Mitigation pathways compatible with 1.5°C in the context of sustainable development.

Rogers, B. M., A. J. Soja, M. L. Goulden and J. T. Randerson, 2015: Influence of tree species on continental differences in boreal fires and climate feedbacks. *Nature Geoscience*, 8, 228, doi:10.1038/ngeo2352 https://www.nature.com/articles/ngeo2352#supplementary-information.

Rohde, R. F. et al., 2019: Vegetation and climate change in the Pro-Namib and Namib Desert based on repeat photography: Insights into climate trends. *Journal of Arid Environments*.

Rollinson, C. R. et al., 2017: Emergent climate and CO2 sensitivities of net primary productivity in ecosystem models do not agree with empirical data in temperate forests of eastern North America. *Global Change Biology*, 23 (7), 2755-2767, doi:10.1111/gcb.13626.

Rolls, R., J. Heino and B. Chessman, 2016: Unravelling the joint effects of flow regime, climatic variability and dispersal mode on beta diversity of riverine communities. *Freshwater Biology*, 61 (8), 1350-1364, doi:10.1111/fwb.12793.

Romanovsky, V. E. et al., 2019: Terrestrial permafrost. *Bulletin of the American Meteorological Society*, 100 (S153-S156).

Root, T. L., D. P. MacMynowski, M. D. Mastrandrea and S. H. Schneider, 2005: Human-modified temperatures induce species changes: Joint attribution. *Proceedings of the National Academy of Sciences of the United States of America*, 102 (21), 7465-7469, doi:10.1073/pnas.0502286102.

Root, T. L. et al., 2003: Fingerprints of global warming on wild animals and plants. *Nature*, 421 (6918), 57, doi:10.1038/nature01333.

Rosan, T. M. et al., 2019: Extensive twenty-first century woody encroachment in South America's savanna. *Geophysical Research Letters*.

Rose, 2016: Climate-induced warming of lakes can be either amplified or suppressed by trends in water clarity. *Limnology and Oceanography Letters*, 1, 44-53.

Rosenzweig, C. et al., 2008: Attributing physical and biological impacts to anthropogenic climate change. *Nature*, 453 (7193), 353-357, doi:10.1038/nature06937.

Rowland, E. L., N. Fresco, D. Reid and H. A. Cooke, 2016: Examining climate-biome (“cliome”) shifts for Yukon and its protected areas. *Global Ecology and Conservation*, 8, 1-17, doi:10.1016/j.gecco.2016.07.006.

Roy, J. et al., 2016: Elevated CO2 maintains grassland net carbon uptake under a future heat and drought extreme. *Proceedings of the National Academy of Sciences*, 113 (22), 6224-6229.

Rudgers, J. A. et al., 2018: Climate sensitivity functions and net primary production: A framework for incorporating climate mean and variability. *Ecology*, 99 (3), 576-582.

Ruesch, A. S. et al., 2012: Projected climate-induced habitat loss for salmonids in the John Day River network, Oregon, USA. *Conservation Biology*, 26 (5), 873-882.

Ruffault, J. and F. Mouillot, 2015: How a new fire-suppression policy can abruptly reshape the fire-weather relationship. *Ecosphere*, 6 (10), doi:10.1890/es15-00182.1.

Runting, R. K. et al., 2017: Incorporating climate change into ecosystem service assessments and decisions: a review. *Global Change Biology*, 23 (1), 28-41, doi:10.1111/gcb.13457.

Rusak, 2018: Wind and trophic status explain within and among lake variability of algal biomass. *Limnology and Oceanography Letters*.

Ryan, C. M. et al., 2016: Ecosystem services from southern African woodlands and their future under global change. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371 (1703), 20150312.

Saatchi, S. S. et al., 2011: Benchmark map of forest carbon stocks in tropical regions across three continents. *Proceedings of the National Academy of Sciences of the United States of America*, 108 (24), 9899-9904, doi:10.1073/pnas.1019576108.

Sadegh, M. et al., 2018: Multihazard Scenarios for Analysis of Compound Extreme Events. *Geophysical Research Letters*, 45 (11), 5470-5480, doi:10.1029/2018gl077317.

Saintilan, N. and K. Rogers, 2015: Woody plant encroachment of grasslands: a comparison of terrestrial and wetland settings. *New Phytologist*, 205 (3), 1062-1070.

Salazar, L. F. and C. A. Nobre, 2010: Climate change and thresholds of biome shifts in Amazonia. *Geophysical Research Letters*, 37, doi:10.1029/2010gl043538.

Salmon, V. G. et al., 2016: Nitrogen availability increases in a tundra ecosystem during five years of experimental permafrost thaw. *Global Change Biology*, 22 (5), 1927-1941.

Sampaio, G. et al., 2007: Regional climate change over eastern Amazonia caused by pasture and soybean cropland expansion. *Geophysical Research Letters*, 34 (17), doi:10.1029/2007gl030612.

Sanches, L. et al., 2019: Global regulation of methane emission from natural lakes. *Scientific Reports*, 9, doi:10.1038/s41598-018-36519-5.

Sankey, J. B. et al., 2017: Climate, wildfire, and erosion ensemble foretells more sediment in western USA watersheds. *Geophysical Research Letters*, 44 (17), 8884-8892, doi:10.1002/2017gl073979.

Santanello, J. A., J. Roundy and P. A. Dirmeyer, 2015: Quantifying the Land-Atmosphere Coupling Behavior in Modern Reanalysis Products over the U.S. Southern Great Plains. *Journal of Climate*, 28 (14), 5813-5829, doi:10.1175/jcli-d-14-00680.1.

Santiago, J. et al., 2016: Brown trout thermal niche and climate change: expected changes in the distribution of cold-water fish in central Spain. *Ecohydrology*, 9 (3), 514-528, doi:10.1002/eco.1653.

Santín, C. et al., 2016: Towards a global assessment of pyrogenic carbon from vegetation fires. *Global Change Biology*, 22 (1), 76-91, doi:10.1111/gcb.12985.

Schaefer, K. et al., 2014: The impact of the permafrost carbon feedback on global climate. *Environmental Research Letters*, 9 (8), doi:10.1088/1748-9326/9/8/085003.

Scharfenberger, U. et al., 2019: Effects of trophic status, water level, and temperature on shallow lake metabolism and metabolic balance: A standardized pan-European mesocosm experiment. *Limnology and Oceanography*, 64 (2), 616-631, doi:10.1002/lno.11064.

Scheffer, M. et al., 2012: Thresholds for boreal biome transitions. *Proceedings of the National Academy of Sciences of the United States of America*, 109 (52), 21384-21389, doi:10.1073/pnas.1219844110.

Scheffers, B. R. et al., 2016: The broad footprint of climate change from genes to biomes to people. *Science*, 354 (6313), aaf7671, doi:10.1126/science.aaf7671.

Scheiter, S. and S. I. Higgins, 2009: Impacts of climate change on the vegetation of Africa: an adaptive dynamic vegetation modelling approach. *Global Change Biology*, 15 (9), 2224-2246, doi:10.1111/j.1365-2486.2008.01838.x.

Scheiter, S., S. I. Higgins, J. Beringer and L. B. Hutley, 2015: Climate change and long-term fire management impacts on Australian savannas. *New Phytologist*, 205 (3), 1211-1226, doi:10.1111/nph.13130.

Schimel, D. et al., 2015a: Observing terrestrial ecosystems and the carbon cycle from space. *Global Change Biology*, 21 (5), 1762-1776, doi:10.1111/gcb.12822.

Schimel, D., B. B. Stephens and J. B. Fisher, 2015b: Effect of increasing CO2 on the terrestrial carbon cycle. *Proceedings of the National Academy of Sciences of the United States of America*, 112 (2), 436-441, doi:10.1073/pnas.1407302112/-/DCSupplemental.

Schleuning, M. et al., 2016: Ecological networks are more sensitive to plant than to animal extinction under climate change. *Nature Communications*, 7, doi:13965 10.1038/ncomms13965.

Schmidt, S., G. Lischeid, T. Hintze and R. Adrian, 2019: Disentangling limnological processes in the time-frequency domain. *Limnology and Oceanography*, 64 (2), 423-440, doi:10.1002/lno.11049.

Schmitz, O. J. et al., 2018: Animals and the zoogeochemistry of the carbon cycle. *Science*, 362 (6419), doi:10.1126/science.aar3213.

Schneider, P. and S. Hook, 2010: Space observations of inland water bodies show rapid surface warming since 1985. *Geophysical Research Letters*, 37, doi:10.1029/2010gl045059.

Scholze, M., W. Knorr, N. W. Arnell and I. C. Prentice, 2006: A climate-change risk analysis for world ecosystems. *Proceedings of the National Academy of Sciences*, 103 (35), 13116-13120, doi:10.1073/pnas.0601816103.

Schulte-Uebbing, L. and W. de Vries, 2018: Global-scale impacts of nitrogen deposition on tree carbon sequestration in tropical, temperate, and boreal forests: A meta-analysis. *Global Change Biology*, 24 (2), E416-E431, doi:10.1111/gcb.13862.

Schuur, E. A. G. et al., 2015: Climate change and the permafrost carbon feedback. *Nature*, 520 (7546), 171-179, doi:10.1038/nature14338.

Schuur, E. A. G. et al., 2009: The effect of permafrost thaw on old carbon release and net carbon exchange from tundra. *Nature*, 459 (7246), 556.

Schwartz, M. W. et al., 2015: Increasing elevation of fire in the Sierra Nevada and implications for forest change. *Ecosphere*, 6 (7), doi:10.1890/es15-00003.1.

Schwartz, M. W. et al., 2012: Managed Relocation: Integrating the Scientific, Regulatory, and Ethical Challenges. *BioScience*, 62 (8), 732-743, doi:10.1525/bio.2012.62.8.6.

Searchinger, T. D. et al., 2015: High carbon and biodiversity costs from converting Africa's wet savannahs to cropland. *Nature Climate Change*, 5 (5), 481-486, doi:10.1038/nclimate2584.

Seddon, N. et al., 2016: Ecosystem based adaptation: a win-win formula for sustainability in a warming world? IIED, London.

Seddon, N. et al., 2019: Grounding nature-based climate solutions in sound biodiversity science. *Nature Climate Change*, 9 (2), 84-87, doi:10.1038/s41558-019-0405-0.

Seekell, D., P. Bystrom and J. Karlsson, 2018: Lake morphometry moderates the relationship between water color and fish biomass in small boreal lakes. *Limnology and Oceanography*, 63 (5), 2171-2178, doi:10.1002/lno.10931.

Seekell, D. et al., 2015: The influence of dissolved organic carbon on primary production in northern lakes. *Limnology and Oceanography*, 60 (4), 1276-1285, doi:10.1002/lno.10096.

Seimon, T. A. et al., 2017: Long-term monitoring of tropical alpine habitat change, Andean anurans, and chytrid fungus in the Cordillera Vilcanota, Peru: Results from a decade of study. *Ecology and Evolution*, 7 (5), 1527-1540, doi:10.1002/ece3.2779.

Seneviratne, S. I. et al., 2012: Changes in climate extremes and their impacts on the natural physical environment. Cambridge University Press, Cambridge, UK, and New York, NY, USA, pp. 109-230.

Settele, J. et al., 2014: Terrestrial and inland water systems. In: Climate Change 2014: Impacts, Adaptation, Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the IPCC [Field, C. B., V. R. Barros, D. J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea and L. L. White (eds.)]. Cambridge University Press, Cambridge, 271-359.

Sgro, C. M., A. J. Lowe and A. A. Hoffmann, 2011: Building evolutionary resilience for conserving biodiversity under climate change. *Evolutionary Applications*, 4 (2), 326-337, doi:10.1111/j.1752-4571.2010.00157.x.

Sgro, C. M., J. S. Terblanche and A. A. Hoffmann, 2016: What Can Plasticity Contribute to Insect Responses to Climate Change? In: Annual Review of Entomology, Vol 61 [Berenbaum, M. R. (ed.)], 61, 433-451.

Shah, D. N. et al., 2014: Current and future latitudinal gradients in stream macroinvertebrates richness across North America. *Freshwater Science*, 33 (4), 1136-1147.

Shah, R. D. T., D. N. Shah and S. Domisch, 2012: Range shifts of a relict Himalayan dragonfly in the Hindu Kush Himalayan region under climate change scenarios. *International Journal of Odonatology*, 15 (3), 209-222, doi:Doi 10.1080/13887890.2012.697399.

Shanahan, T. M. et al., 2009: Atlantic Forcing of Persistent Drought in West Africa. *Science*, 324 (5925), 377-380, doi:10.1126/science.1166352.

Sharma, S. et al., 2019: Widespread loss of lake ice around the Northern Hemisphere in a warming world. *Nature Climate Change*, 9 (3), 227-+, doi:10.1038/s41558-018-0393-5.

Shatwell, T., W. Thiery and G. Kirillin, 2019: Future projections of temperature and mixing regime of European temperate lakes. *Hydrology and Earth System Sciences*, 23 (3), 1533-1551, doi:10.5194/hess-23-1533-2019.

Shin, Y.-J. et al., 2019: *IPBES Global Assessment on Biodiversity and Ecosystem Services, Chapter 4: Plausible futures of nature, its contributions to people and their good quality of life*. Bonn, GErmany, in press.

Shuman, J. K., H. H. Shugart and T. L. O'halloran, 2011: Sensitivity of Siberian larch forests to climate change. *Global Change Biology*, 17 (7), 2370-2384, doi:10.1111/j.1365-2486.2011.02417.x.

Siepielski, A. M. et al., 2019: No evidence that warmer temperatures are associated with selection for smaller body sizes. *Proc. R. Soc. B*, 286: 20191332, doi:http://dx.doi.org/10.1098/rspb.2019.1332.

Sierra-Correa, P. C. and J. R. C. Kintz, 2015: Ecosystem-based adaptation for improving coastal planning for sea-level rise: A systematic review for mangrove coasts. *Marine Policy*, 51, 385-393.

Sigdel, S. R. et al., 2018: Moisture-mediated responsiveness of treeline shifts to global warming in the Himalayas. *Global Change Biology*, 24 (11), 5549-5559, doi:10.1111/gcb.14428.

Siipi, H. and M. Ahteensuu, 2016: Moral Relevance of Range and Naturalness in Assisted Migration. *Environmental Values*, 25 (4), 465-483, doi:10.3197/096327116x14661540759278.

Sillett, S. C. et al., 2015: How do tree structure and old age affect growth potential of California redwoods? *Ecological Monographs*, 85 (2), 181-212, doi:10.1890/14-1016.1.

Sillmann, J. et al., 2017: Understanding, modeling and predicting weather and climate extremes: Challenges and opportunities. *Weather and Climate Extremes*, 18, 65-74.

Silva, C. V. J. et al., 2018: Drought-induced Amazonian wildfires instigate a decadal-scale disruption of forest carbon dynamics. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 373 (1760), doi:10.1098/rstb.2018.0043.

Silva, L. C. R. et al., 2016: Tree growth acceleration and expansion of alpine forests: The synergistic effect of atmospheric and edaphic change. *Science Advances*, 2 (8), e1501302.

Silva, P. S. et al., 2019: Impacts of the 1.5 degrees C global warming target on future burned area in the Brazilian Cerrado. *Forest Ecology and Management*, 446, 193-203, doi:10.1016/j.foreco.2019.05.047.

Singer, M. C., 2017: Shifts in time and space interact as climate warms. *Proceedings of the National Academy of Sciences of the United States of America*, 114 (49), 12848-12850, doi:10.1073/pnas.1718334114.

Singer, M. C. and C. Parmesan, 2010: Phenological asynchrony between herbivorous insects and their hosts: signal of climate change or pre-existing adaptive strategy? *Philosophical Transactions of the Royal Society B-Biological Sciences*, 365 (1555), 3161-3176, doi:10.1098/rstb.2010.0144.

Singleton, M. P., A. E. Thode, A. J. S. Meador and J. M. Iniguez, 2019: Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. *Forest Ecology and Management*, 433, 709-719, doi:10.1016/j.foreco.2018.11.039.

Sistla, S. A. et al., 2013: Long-term warming restructures Arctic tundra without changing net soil carbon storage. *Nature*, 497 (7451), 615.

Sitch, S., P. M. Cox, W. J. Collins and C. Huntingford, 2007: Indirect radiative forcing of climate change through ozone effects on the land-carbon sink. *Nature*, 448 (7155), 791-U4, doi:10.1038/nature06059.

Sitch, S. et al., 2015: Recent trends and drivers of regional sources and sinks of carbon dioxide. *Biogeosciences*, 12 (3), 653-679, doi:10.5194/bg-12-653-2015.

Sitch, S. et al., 2008: Evaluation of the terrestrial carbon cycle, future plant geography and climate-carbon cycle feedbacks using five Dynamic Global Vegetation Models (DGVMs). *Global Change Biology*, 14 (9), 2015-2039, doi:10.1111/j.1365-2486.2008.01626.x.

Sittaro, F., A. Paquette, C. Messier and C. A. Nock, 2017: Tree range expansion in eastern North America fails to keep pace with climate warming at northern range limits. *Global Change Biology*, 23 (8), 3292-3301, doi:10.1111/gcb.13622.

Skowno, A. L. et al., 2017: Woodland expansion in South African grassy biomes based on satellite observations (1990–2013): general patterns and potential drivers. *Global Change Biology*, 23 (6), 2358–2369.

Slingsby, J. et al., 2017: Intensifying postfire weather and biological invasion drive species loss in a Mediterranean-type biodiversity hotspot. *Proceedings of the National Academy of Sciences of the United States of America*, 114 (18), 4697-4702, doi:10.1073/pnas.1619014114.

Smale, D. A. et al., 2019: Marine heatwaves threaten global biodiversity and the provision of ecosystem services. *Nature Climate Change*, 9 (4), 306-+, doi:10.1038/s41558-019-0412-1.

Smit, I. P. et al., 2016: An examination of the potential efficacy of high-intensity fires for reversing woody encroachment in savannas. *Journal of Applied Ecology*, 53 (5), 1623-1633.

Smith, P. et al., 2019: *Chapter 6: Interlinkages between Desertification, Land Degradation, Food Security and GHG fluxes: synergies, trade-offs and Integrated Response Options* [press, i. (ed.)]. Special Report Climate and Land, IPCC, Geneva, Switzerland.

Smith, P. et al., 2018: Impacts on terrestrial biodiversity of moving from a 2°C to a 1.5°C target. *Philosophical Transactions of the Royal Society A*, 376.

Smithers, B. V., M. P. North, C. I. Millar and A. M. Latimer, 2018: Leap frog in slow motion: Divergent responses of tree species and life stages to climatic warming in Great Basin subalpine forests. *Global Change Biology*, 24 (2), E442-E457, doi:10.1111/gcb.13881.

Snyder, K. A. et al., 2019: Effects of changing climate on the hydrological cycle in cold desert ecosystems of the Great Basin and Columbia Plateau. *Rangeland Ecology & Management*, 72 (1), 1–12.

Socolar, J. B., P. N. Epanchin, S. R. Beissinger and M. W. Tingley, 2017: Phenological shifts conserve thermal niches in North American birds and reshape expectations for climate-driven range shifts. *Proceedings of the National Academy of Sciences of the United States of America*, 114 (49), 12976-12981, doi:10.1073/pnas.1705897114.

Solomon, C. et al., 2015: Ecosystem Consequences of Changing Inputs of Terrestrial Dissolved Organic Matter to Lakes: Current Knowledge and Future Challenges. *Ecosystems*, 18 (3), 376-389, doi:10.1007/s10021-015-9848-y.

Somers, L. D. et al., under review: Groundwater buffers decreasing glacier melt in the Andes - but not forever. *Geophysical Research Letters*.

Song, X.-P. et al., 2018: Global land change from 1982 to 2016. *Nature*, 560 (7720), 639.

Sousa, P. M. et al., 2018: The 'Day Zero'Cape Town drought and the poleward migration of moisture corridors. *Environmental Research Letters*, 13 (12), 124025.

Spalding, M. D. et al., 2014: The role of ecosystems in coastal protection: Adapting to climate change and coastal hazards. *Ocean & Coastal Management*, 90, 50-57, doi:10.1016/j.ocecoaman.2013.09.007.

Stålhandske, S., K. Gotthard and O. Leimar, 2017: Winter chilling speeds spring development of temperate butterflies. *Journal of Animal Ecology*, 86 (4), 718-729, doi:10.1111/1365-2656.12673.

Stan, K. and A. Sanchez-Azofeifa, 2019: Tropical dry forest diversity, climatic response, and resilience in a changing climate. *Forests*, 10 (5), 1-19, doi:10.3390/f10050443.

Staver, A. C., S. Archibald and S. A. Levin, 2011: The global extent and determinants of savanna and forest as alternative biome states. *Science*, 334 (6053), 230-232.

Stegehuis, A. I. et al., 2015: An observation-constrained multi-physics WRF ensemble for simulating European mega heat waves. *Geoscientific Model Development*, 8 (7), 2285-2298, doi:10.5194/gmd-8-2285-2015.

Stephens, S. L. et al., 2014: Temperate and boreal forest mega-fires: characteristics and challenges. *Frontiers in Ecology and the Environment*, 12 (2), 115-122, doi:10.1890/120332.

Stephens, S. L. et al., 2018: Drought, Tree Mortality, and Wildfire in Forests Adapted to Frequent Fire. *BioScience*, 68 (2), 77-88, doi:10.1093/biosci/bix146.

Stephenson, N. L. et al., 2019: Which trees die during drought? The key role of insect host-tree selection. *Journal of Ecology*, 107 (5), 2383-2401, doi:10.1111/1365-2745.13176.

Stevens-Rumann, C. S. et al., 2018: Evidence for declining forest resilience to wildfires under climate change. *Ecology Letters*, 21 (2), 243-252, doi:10.1111/ele.12889.

Stevens, N., B. Erasmus, S. Archibald and W. Bond, 2016: Woody encroachment over 70 years in South African savannahs: overgrazing, global change or extinction aftershock? *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371 (1703), 20150437.

Stevens, N., C. E. Lehmann, B. P. Murphy and G. Durigan, 2017: Savanna woody encroachment is widespread across three continents. *Global Change Biology*, 23 (1), 235–244.

Straile, D., R. Adrian and D. Schindler, 2012: Uniform Temperature Dependency in the Phenology of a Keystone Herbivore in Lakes of the Northern Hemisphere. *PLOS ONE*, 7 (10), doi:10.1371/journal.pone.0045497.

Strauss, J. et al., 2017: Deep Yedoma permafrost: A synthesis of depositional characteristics and carbon vulnerability. *Earth-Science Reviews*, 172, 75-86, doi:10.1016/j.earscirev.2017.07.007.

Strömberg, C. A. E., 2011: Evolution of grasses and grassland ecosystems. *Annual Review of Earth and Planetary Sciences*, 39, 517-544.

Sturm, M., T. Douglas, C. Racine and G. E. Liston, 2005: Changing snow and shrub conditions affect albedo with global implications. *Journal of Geophysical Research-Biogeosciences*, 110 (G1), doi:10.1029/2005jg000013.

Suarez-Moreno, R., B. Rodriguez-Fonseca, J. A. Barroso and A. H. Fink, 2018: Interdecadal Changes in the Leading Ocean Forcing of Sahelian Rainfall Interannual Variability: Atmospheric Dynamics and Role of Multidecadal SST Background. *Journal of Climate*, 31 (17), 6687-6710, doi:10.1175/jcli-d-17-0367.1.

Suggitt, A. et al., 2017: Conducting robust ecological analyses with climate data. *Oikos*, 126 (11), 1533-1541, doi:10.1111/oik.04203.

Suggitt, A. et al., 2015: Microclimate affects landscape level persistence in the British Lepidoptera. *Journal of Insect Conservation*, 19 (2), 237-253, doi:10.1007/s10841-014-9749-y.

Suggitt, A. J. et al., 2018: Extinction risk from climate change is reduced by microclimatic buffering. *Nature Climate Change*, 8 (8), 713-717, doi:10.1038/s41558-018-0231-9.

Sullivan, M. J. P. et al., 2017: Diversity and carbon storage across the tropical forest biome. *Scientific Reports*, 7, doi:10.1038/srep39102.

Swann, A. L. S. et al., 2018: Continental-scale consequences of tree die-offs in North America: identifying where forest loss matters most. *Environmental Research Letters*, 13 (5), 055014, doi:10.1088/1748-9326/aaba0f.

Sweet, L. C. et al., 2019: Congruence between future distribution models and empirical data for an iconic species at Joshua Tree National Park. *Ecosphere*, 10 (6), doi:10.1002/ecs2.2763.

Syphard, A. D., T. J. Brennan and J. E. Keeley, 2019: Drivers of chaparral type conversion to herbaceous vegetation in coastal Southern California. *Diversity and Distributions*, 25 (1), 90-101.

Syphard, A. D., J. E. Keeley and J. T. Abatzoglou, 2017: Trends and drivers of fire activity vary across California aridland ecosystems. *Journal of Arid Environments*, 144, 110-122, doi:10.1016/j.jaridenv.2017.03.017.

Syphard, A. D., T. Sheehan, H. Rustigian-Romsos and K. Ferschweiler, 2018: Mapping future fire probability under climate change: Does vegetation matter? *PLOS ONE*, 13 (8), doi:10.1371/journal.pone.0201680.

Tainio, A. et al., 2016: Conservation of grassland butterflies in Finland under a changing climate. *Regional Environmental Change*, 16 (1), 71-84, doi:10.1007/s10113-014-0684-y.

Tape, K. D., K. Christie, G. Carroll and J. A. O'Donnell, 2016a: Novel wildlife in the A rctic: the influence of changing riparian ecosystems and shrub habitat expansion on snowshoe hares. *Global Change Biology*, 22 (1), 208-219.

Tape, K. D. et al., 2016b: Range expansion of moose in Arctic Alaska linked to warming and increased shrub habitat. *PLOS ONE*, 11 (4), e0152636.

Tape, K. D. et al., 2018: Tundra be dammed: Beaver colonization of the Arctic. *Global Change Biology*, 24 (10), 4478-4488.

Tarnocai, C. et al., 2009: Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biogeochemical Cycles*, 23, doi:10.1029/2008gb003327.

Taylor, S., M. Knight and A. Harfoot, 2014: *National biodiversity climate change vulnerability model. Natural England Research Report NERR054*. Natural England, York, UK.

te Beest, M., J. Sitters, C. B. Ménard and J. Olofsson, 2016: Reindeer grazing increases summer albedo by reducing shrub abundance in Arctic tundra. *Environmental Research Letters*, 11 (12), 125013.

Tebakari, T., S. Wongsa and Y. Hayashi, 2018: Floods in Southern Thailand in December 2016 and January 2017. *Journal of Disaster Research*, 13 (4), 793-803, doi:10.20965/jdr.2018.p0793.

Teplitsky, C. and V. Millien, 2014: Climate warming and Bergmann's rule through time: is there any evidence? *Evolutionary Applications*, 7 (1), 156-168, doi:10.1111/eva.12129.

Terrer, C. et al., 2019: Nitrogen and phosphorus constrain the CO2 fertilization of global plant biomass. *Nature Climate Change*, 9 (9), 684-+, doi:10.1038/s41558-019-0545-2.

Tesi, T. et al., 2016: Massive remobilization of permafrost carbon during post-glacial warming. *Nature Communications*, 7, doi:10.1038/ncomms13653.

Thackeray, S. et al., 2010: Trophic level asynchrony in rates of phenological change for marine, freshwater and terrestrial environments. *Global Change Biology*, 16 (12), 3304-3313, doi:10.1111/j.1365-2486.2010.02165.x.

Thomas, C. D., 2011: Translocation of species, climate change, and the end of trying to recreate past ecological communities. *Trends in Ecology & Evolution*, 26 (5), 216-221, doi:10.1016/j.tree.2011.02.006.

Thomas, C. D. et al., 2001: Ecological and evolutionary processes at expanding range margins. *Nature*, 411 (6837), 577-581, doi:10.1038/35079066.

Thomas, C. D. et al., 2012: Protected areas facilitate species' range expansions. *Proceedings of the National Academy of Sciences*, 109 (35), 14063-14068.

Thomas, S. M., S. W. Griffiths and S. J. Ormerod, 2016: Beyond cool: adapting upland streams for climate change using riparian woodlands. *Global Change Biology*, 22 (1), 310-324.

Thompson, L. G. et al., 2017: Impacts of Recent Warming and the 2015/2016 El Nino on Tropical Peruvian Ice Fields. *Journal of Geophysical Research-Atmospheres*, 122 (23), 12688-12701, doi:10.1002/2017jd026592.

Thompson, S. J., C. M. Handel, R. M. Richardson and L. B. McNew, 2016: When Winners Become Losers: Predicted Nonlinear Responses of Arctic Birds to Increasing Woody Vegetation. *PLOS ONE*, 11 (11), doi:e0164755 10.1371/journal.pone.0164755.

Thomson, J. et al., 2012: The influences of climatic variation and vegetation on stream biota: lessons from the Big Dry in southeastern Australia. *Global Change Biology*, 18 (5), 1582-1596, doi:10.1111/j.1365-2486.2011.02609.x.

Thornton, P. K., P. J. Ericksen, M. Herrero and A. J. Challinor, 2014: Climate variability and vulnerability to climate change: a review. *Global Change Biology*, 20 (11), 3313-3328, doi:10.1111/gcb.12581.

Thrane, J., D. Hessen and T. Andersen, 2014: The Absorption of Light in Lakes: Negative Impact of Dissolved Organic Carbon on Primary Productivity. *Ecosystems*, 17 (6), 1040-1052, doi:10.1007/s10021-014-9776-2.

Timpane-Padgham, B. L., T. Beechie and T. Klinger, 2017: A systematic review of ecological attributes that confer resilience to climate change in environmental restoration. *PLOS ONE*, 12 (3), e0173812.

Tingley, M. W. et al., 2012: The push and pull of climate change causes heterogeneous shifts in avian elevational ranges. *Global Change Biology*, 18 (11), 3279-3290, doi:10.1111/j.1365-2486.2012.02784.x.

Tootchi, A., A. Jost and A. Ducharne, 2019: Multi-source global wetland maps combining surface water imagery and groundwater constraints. *Earth Syst. Sci. Data*, 11 (1), 189-220, doi:10.5194/essd-11-189-2019.

Tranvik, L. et al., 2009: Lakes and reservoirs as regulators of carbon cycling and climate. *Limnology and Oceanography*, 54 (6), 2298-2314, doi:10.4319/lo.2009.54.6_part_2.2298.

Trichon, V., P. Hiernaux, R. Walcker and E. Mougin, 2018: The persistent decline of patterned woody vegetation: The tiger bush in the context of the regional Sahel greening trend. *Global Change Biology*, 24 (6), 2633-2648, doi:10.1111/gcb.14059.

Trisurat, Y., 2018: Planning Thailand's Protected Areas in Response to Future Land Use and Climate Change. *International Journal of Conservation Science*, 9 (4), 805-820.

Trisurat, Y., J. Alkemade and E. Arets, 2009: Projecting forest tree distributions and adaptation to climate change in northern Thailand. *Journal of ecology and the natural environment*, 1 (3), 55-63.

Trisurat, Y., P. Eawpanich and R. Kalliola, 2016: Integrating land use and climate change scenarios and models into assessment of forested watershed services in Southern Thailand. *Environmental Research*, 147, 611-620.

Trisurat, Y., R. P. Shrestha and R. Kjelgren, 2011: Plant species vulnerability to climate change in Peninsular Thailand. *Applied Geography*, 31 (3), 1106-1114.

Tucker, C. J., 1979: RED AND PHOTOGRAPHIC INFRARED LINEAR COMBINATIONS FOR MONITORING VEGETATION. *Remote Sensing of Environment*, 8 (2), 127-150, doi:10.1016/0034-4257(79)90013-0.

Turco, M. et al., 2018: Exacerbated fires in Mediterranean Europe due to anthropogenic warming projected with non-stationary climate-fire models. *Nature Communications*, 9 (1), 3821, doi:10.1038/s41467-018-06358-z.

Turetsky, M. R. et al., 2019: *Permafrost collapse is accelerating carbon release*. Nature Publishing Group.

Tydecks, L. et al., 2018: Spatial and topical imbalances in biodiversity research. *PLOS ONE*, 13 (7), doi:10.1371/journal.pone.0199327.

Tyukavina, A. et al., 2015: Aboveground carbon loss in natural and managed tropical forests from 2000 to 2012. *Environmental Research Letters*, 10 (7), doi:10.1088/1748-9326/10/7/074002.

Uboni, A. et al., 2016: Long-Term Trends and Role of Climate in the Population Dynamics of Eurasian Reindeer. *PLOS ONE*, 11 (6), doi:e0158359 10.1371/journal.pone.0158359.

Uda, S. K., L. Hein and D. Atmoko, 2019: Assessing the health impacts of peatland fires: a case study for Central Kalimantan, Indonesia. *Environ Sci Pollut Res Int*, doi:10.1007/s11356-019-06264-x.

Ummenhofer, C. C. and G. A. Meehl, 2017: Extreme weather and climate events with ecological relevance: a review. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 372 (1723), doi:10.1098/rstb.2016.0135.

UNEP-WCMC, IUCN and NGS, 2018: *Protected Planet Report 2018*. UNEP-WCMC, IUCN and NGS, Cambridge, UK; Gland, Switzerland; Washington D.C., USA.

UNEP, 2019: *Global Environment Outlook 6*. United Nations Environment Programme, Nairobi, Kenya [Available at: https://www.unenvironment.org/resources/global-environment-outlook-6].

Urban, M., 2015: Accelerating extinction risk from climate change. *Science*, 348 (6234), 571-573, doi:10.1126/science.aaa4984.

Valdivia, C. et al., 2010: Adapting to Climate Change in Andean Ecosystems: Landscapes, Capitals, and Perceptions Shaping Rural Livelihood Strategies and Linking Knowledge Systems. *Annals of the Association of American Geographers*, 100 (4), 818-834.

Valkó, O., P. Török, B. Deák and B. Tóthmérész, 2014: Prospects and limitations of prescribed burning as a management tool in European grasslands. *Basic and Applied Ecology*, 15 (1), 26-33.

Valladares, F. et al., 2014: The effects of phenotypic plasticity and local adaptation on forecasts of species range shifts under climate change. *Ecology Letters*, 17 (11), 1351-1364.

van Asselen, S. and P. H. Verburg, 2013: Land cover change or land use intensification: simulating land system change with a global-scale land change model. *Global Change Biology*, n/a-n/a, doi:10.1111/gcb.12331.

van der Kolk, H.-J. et al., 2016: Potential Arctic tundra vegetation shifts in response to changing temperature, precipitation and permafrost thaw. *Biogeosciences*, 13 (22), 6229-6245, doi:10.5194/bg-13-6229-2016.

Van der Sande, M. T. et al., 2016: Old-growth Neotropical forests are shifting in species and trait composition. *Ecological Monographs*, 86 (2), 228-243.

Van Der Sleen, P. et al., 2015: No growth stimulation of tropical trees by 150 years of CO 2 fertilization but water-use efficiency increased. *Nature Geoscience*, 8 (1), 24.

van der Werf, G. R. et al., 2017: Global fire emissions estimates during 1997–2016. *Earth System Science Data*, 9 (2), 697-720, doi:10.5194/essd-9-697-2017.

Van Gestel, N. et al., 2018: Predicting soil carbon loss with warming. *Nature*, 554 (7693), E4.

van Mantgem, P. et al., 2009: Widespread Increase of Tree Mortality Rates in the Western United States. *Science*, 323 (5913), 521-524, doi:10.1126/science.1165000.

van Marle, M. J. E. et al., 2017: Fire and deforestation dynamics in Amazonia (1973-2014): Fire Dynamics in Amazonia (1973-2014). *Global Biogeochemical Cycles*, 31 (1), 24-38, doi:10.1002/2016GB005445.

Van Pelt, R. et al., 2016: Emergent crowns and light-use complementarity lead to global maximum biomass and leaf area in Sequoia sempervirens forests. *Forest Ecology and Management*, 375, 279-308, doi:10.1016/j.foreco.2016.05.018.

van Vliet, J., D. A. Eitelberg and P. H. Verburg, 2017: A global analysis of land take in cropland areas and production displacement from urbanization. *Global Environmental Change*, 43, 107-115, doi:https://doi.org/10.1016/j.gloenvcha.2017.02.001.

van Vuuren, D. P. and T. R. Carter, 2014: Climate and socio-economic scenarios for climate change research and assessment: reconciling the new with the old. *Climatic Change*, 122 (3), 415-429, doi:10.1007/s10584-013-0974-2.

van Wilgen, B. W. and A. Wannenburgh, 2016: Co-facilitating invasive species control, water conservation and poverty relief: achievements and challenges in South Africa's Working for Water programme. *Current Opinion in Environmental Sustainability*, 19, 7-17.

Varkkey, H., A. Tyson, S. Al and B. Choiruzzad, 2018: Forest Policy and Economics Palm oil intensification and expansion in Indonesia and Malaysia : Environmental and socio-political factors influencing policy. 92 (September 2017), 148-159, doi:10.1016/j.forpol.2018.05.002.

Veldman, J. W. et al., 2015a: Toward an old-growth concept for grasslands, savannas, and woodlands. *Frontiers in Ecology and the Environment*, 13 (3), 154-162.

Veldman, J. W. et al., 2015b: Tyranny of trees in grassy biomes. *Science*, 347 (6221), 484-485.

Veldman, J. W. et al., 2015c: Where tree planting and forest expansion are bad for biodiversity and ecosystem services. *BioScience*, 65 (10), 1011-1018.

Veldman, J. W. et al., 2017: Grassy biomes: An inconvenient reality for large-scale forest restoration? A comment on the essay by Chazdon and Laestadius. *American Journal of Botany*, 104 (5), 649-651.

Venter, Z. S., M. D. Cramer and H.-J. Hawkins, 2018: Drivers of woody plant encroachment over Africa. *Nature Communications*, 9 (1), 2272.

Veraverbeke, S. et al., 2017: Lightning as a major driver of recent large fire years in North American boreal forests. *Nature Climate Change*, 7 (7), 529-+, doi:10.1038/nclimate3329.

Verburg, R. et al., 2019: An innovation perspective to climate change adaptation in coffee systems. *Environmental Science & Policy*, 97, 16-24.

Vetter, T., 2017: Evaluation of sources of uncertainty in projected hydrological changes under climate change in 12 large-scale river basins. *Climatic Change*, v. 141 (no. 3), pp. 419-433-2017 v.141 no.3, doi:10.1007/s10584-016-1794-y.

Vicca, S. et al., 2016: Remotely-sensed detection of effects of extreme droughts on gross primary production. *Scientific Reports*, 6, 28269, doi:10.1038/srep28269.

Viedma, O., I. R. Urbieta and J. M. Moreno, 2018: Wildfires and the role of their drivers are changing over time in a large rural area of west-central Spain. *Scientific Reports*, 8 (1), 17797, doi:10.1038/s41598-018-36134-4.

Vieira, K. S., P. F. G. Montenegro, G. G. Santana and W. L. d. S. Vieira, 2018: Effect of climate change on distribution of species of common horned frogs in South America. *PLOS ONE*, 13 (9), e0202813, doi:10.1371/journal.pone.0202813.

Vignola, R., B. Locatelli, C. Martinez and P. Imbach, 2009: Ecosystem-based adaptation to climate change: what role for policy-makers, society and scientists? *Mitigation and Adaptation Strategies for Global Change*, 14 (8), 691, doi:10.1007/s11027-009-9193-6.

Villamayor, J. et al., 2018: Atlantic Control of the Late Nineteenth-Century Sahel Humid Period. *Journal of Climate*, 31 (20), 8225-8240, doi:10.1175/jcli-d-18-0148.1.

Virkkala, R. et al., 2014: Protected areas alleviate climate change effects on northern bird species of conservation concern. *Ecology and Evolution*, 4 (15), 2991-3003.

Vitali, A. et al., 2017: Deconstructing human-shaped treelines: Microsite topography and distance to seed source control Pinus nigra colonization of treeless areas in the Italian Apennines. *Forest Ecology and Management*, 406, 37-45, doi:10.1016/j.foreco.2017.10.004.

Vogel, M. M. et al., 2017: Regional amplification of projected changes in extreme temperatures strongly controlled by soil moisture-temperature feedbacks. *Geophysical Research Letters*, 44 (3), 1511-1519, doi:10.1002/2016gl071235.

Volkova, L. et al., 2018: Importance of disturbance history on net primary productivity in the world's most productive forests and implications for the global carbon cycle. *Global Change Biology*, 24 (9), 4293-4303, doi:10.1111/gcb.14309.

von Deimling, T. S. et al., 2015: Observation-based modelling of permafrost carbon fluxes with accounting for deep carbon deposits and thermokarst activity. *Biogeosciences*, 12 (11), 3469-3488, doi:10.5194/bg-12-3469-2015.

Vonlanthen, P. et al., 2012: Eutrophication causes speciation reversal in whitefish adaptive radiations. *Nature*, 482 (7385), 357-U1500, doi:10.1038/nature10824.

Vorosmarty, C. et al., 2010: Global threats to human water security and river biodiversity. *Nature*, 467 (7315), 555-561, doi:10.1038/nature09440.

Vuille, M. et al., 2018: Rapid decline of snow and ice in the tropical Andes – Impacts, uncertainties and challenges ahead. *Earth-Science Reviews*, 176, 195-213, doi:https://doi.org/10.1016/j.earscirev.2017.09.019.

Wadgymar, S. M., M. N. Cumming and A. E. Weis, 2015: The success of assisted colonization and assisted gene flow depends on phenology. *Global Change Biology*, 21 (10), 3786-3799, doi:10.1111/gcb.12988.

Waldram, M. S., W. J. Bond and W. D. Stock, 2008: Ecological engineering by a mega-grazer: white rhino impacts on a South African savanna. *Ecosystems*, 11 (1), 101-112.

Walker, X. J. et al., 2019: Increasing wildfires threaten historic carbon sink of boreal forest soils. *Nature*, 572 (7770), 520-+, doi:10.1038/s41586-019-1474-y.

Wan, W. H. et al., 2017: Hydrological Drought in the Anthropocene: Impacts of Local Water Extraction and Reservoir Regulation in the US. *Journal of Geophysical Research-Atmospheres*, 122 (21), 11313-11328, doi:10.1002/2017jd026899.

Wan, Z. et al., 2018: Ecological responses of Stipa steppe in Inner Mongolia to experimentally increased temperature and precipitation. 2: Plant species diversity and sward characteristics. *The Rangeland Journal*, 40 (2), 147-152.

Wang, J. J. et al., 2015: Wildfire Altering Terrestrial Precursors of Disinfection Byproducts in Forest Detritus. *Environmental Science & Technology*, 49 (10), 5921-5929, doi:10.1021/es505836m.

Wang, S. R. et al., 2018: Potential shift from a carbon sink to a source in Amazonian peatlands under a changing climate. *Proceedings of the National Academy of Sciences of the United States of America*, 115 (49), 12407-12412, doi:10.1073/pnas.1801317115.

Ward, D., M. T. Hoffman and S. J. Collocott, 2014: A century of woody plant encroachment in the dry Kimberley savanna of South Africa. *African Journal of Range & Forage Science*, 31 (2), 107-121.

Wårlind, D., B. Smith, T. Hickler and A. Arneth, 2014: Nitrogen feedbacks increase future terrestrial ecosystem carbon uptake in an individual-based dynamic vegetation model. *Biogeosciences*, 11 (21), 6131-6146, doi:10.5194/bg-11-6131-2014.

Warren, R., 2011: The role of interactions in a world implementing adaptation and mitigation solutions to climate change. *Philosophical Transactions of the Royal Society a-Mathematical Physical and Engineering Sciences*, 369 (1934), 217-241, doi:10.1098/rsta.2010.0271.

Warren, R. et al., 2018: The projected effect on insects, vertebrates, and plants of limiting global warming to 1.5 degrees C rather than 2 degrees C. *Science*, 360 (6390), 791-+, doi:10.1126/science.aar3646.

Warren, R. et al., 2013: Quantifying the benefit of early climate change mitigation in avoiding biodiversity loss. *Nature Climate Change*, 3 (7), 678-682, doi:10.1038/nclimate1887.

Warszawski, L. et al., 2014: The Inter-Sectoral Impact Model Intercomparison Project (ISI–MIP): Project framework. *Proceedings of the National Academy of Sciences*, 111 (9), 3228-3232, doi:10.1073/pnas.1312330110.

Warszawski, L. et al., 2013: A multi-model analysis of risk of ecosystem shifts under climate change. *Environmental Research Letters*, 8 (4), doi:10.1088/1748-9326/8/4/044018.

Wauchope, H. S. et al., 2017: Rapid climate-driven loss of breeding habitat for Arctic migratory birds. *Global Change Biology*, 23 (3), 1085-1094, doi:10.1111/gcb.13404.

Welbergen, J. A., S. M. Klose, N. Markus and P. Eby, 2007: Climate change and the effects of temperature extremes on Australian flying-foxes. *Proceedings of the Royal Society B: Biological Sciences*, 275 (1633), doi:https://doi.org/10.1098/rspb.2007.1385.

Westerling, A. L., 2016: Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371 (1696), 20150178, doi:10.1098/rstb.2015.0178.

Westerling, A. L. et al., 2011: Climate change and growth scenarios for California wildfire. *Climatic Change*, 109, 445-463, doi:10.1007/s10584-011-0329-9.

Weydmann, A., W. Walczowski, J. Carstensen and S. Kwaśniewski, 2018: Warming of Subarctic waters accelerates development of a key marine zooplankton *Calanus finmarchicus*. *Global Change Biology*, 24 (1), 172-183, doi:10.1111/gcb.13864.

Weyhenmeyer, G. and J. Karlsson, 2009: Nonlinear response of dissolved organic carbon concentrations in boreal lakes to increasing temperatures. *Limnology and Oceanography*, 54 (6), 2513-2519, doi:10.4319/lo.2009.54.6_part_2.2513.

Weyhenmeyer, G. et al., 2011: Large geographical differences in the sensitivity of ice-covered lakes and rivers in the Northern Hemisphere to temperature changes. *Global Change Biology*, 17 (1), 268-275, doi:10.1111/j.1365-2486.2010.02249.x.

Whittaker, R. H., 1975: *Community and Ecosystems*. McMillan, New York, NY.

Wieder, W. R., C. C. Cleveland, D. M. Lawrence and G. B. Bonan, 2015a: Effects of model structural uncertainty on carbon cycle projections: biological nitrogen fixation as a case study. *Environmental Research Letters*, 10 (4), doi:044016 10.1088/1748-9326/10/4/044016.

Wieder, W. R., C. C. Cleveland, W. K. Smith and K. Todd-Brown, 2015b: Future productivity and carbon storage limited by terrestrial nutrient availability. *Nature Geoscience*, 8, 441, doi:10.1038/ngeo2413 https://www.nature.com/articles/ngeo2413#supplementary-information.

Wiedner, C., J. Rucker, R. Bruggemann and B. Nixdorf, 2007: Climate change affects timing and size of populations of an invasive cyanobacterium in temperate regions. *Oecologia*, 152 (3), 473-484, doi:10.1007/s00442-007-0683-5.

Wiens, J., 2016: Climate-Related Local Extinctions Are Already Widespread among Plant and Animal Species. *Plos Biology*, 14 (12), doi:10.1371/journal.pbio.2001104.

Wijedasa, L. S. et al., 2018: Carbon emissions from South-East Asian peatlands will increase despite emission-reduction schemes. *Glob Chang Biol*, 24 (10), 4598-4613, doi:10.1111/gcb.14340.

Wik, M. et al., 2016: Climate-sensitive northern lakes and ponds are critical components of methane release. *Nature Geoscience*, 9 (2), 99.

Wilcox, K. R. et al., 2017: Asymmetric responses of primary productivity to precipitation extremes: a synthesis of grassland precipitation manipulation experiments. *Global Change Biology*, 23 (10), 4376-4385.

Wilkin, K. M., D. D. Ackerly and S. L. Stephens, 2016: Climate Change Refugia, Fire Ecology and Management. *Forests*, 7 (4), doi:10.3390/f7040077.

Williams, A. P. et al., 2019: Observed Impacts of Anthropogenic Climate Change on Wildfire in California. *Earth's Future*, 7 (8), 892-910, doi:10.1029/2019ef001210.

Williams, A. P. et al., 2015: Contribution of anthropogenic warming to California drought during 2012-2014. *Geophysical Research Letters*, 42 (16), 6819-6828, doi:10.1002/2015gl064924.

Williamson, C. et al., 2015: Ecological consequences of long-term browning in lakes. *Scientific Reports*, 5, doi:10.1038/srep18666.

Willis, S. G. et al., 2015: Integrating climate change vulnerability assessments from species distribution models and trait-based approaches. *Biological Conservation*, 190, 167-178, doi:10.1016/j.biocon.2015.05.001.

Willis, S. G. et al., 2009: Assisted colonization in a changing climate: a test-study using two UK butterflies. *Conservation Letters*, 2 (1), 45-51, doi:10.1111/j.1755-263X.2008.00043.x.

Wilson, A. M. et al., 2010: A hierarchical Bayesian model of wildfire in a Mediterranean biodiversity hotspot: implications of weather variability and global circulation. *Ecological Modelling*, 221 (1), 106-112.

Winckler, J., Q. Lejeune, C. H. Reick and J. Pongratz, 2019: Nonlocal Effects Dominate the Global Mean Surface Temperature Response to the Biogeophysical Effects of Deforestation. *Geophysical Research Letters*, 46 (2), 745-755, doi:10.1029/2018gl080211.

Winder, M., J. Reuter and S. Schladow, 2009a: Lake warming favours small-sized planktonic diatom species. *Proceedings of the Royal Society B-Biological Sciences*, 276 (1656), 427-435, doi:10.1098/rspb.2008.1200.

Winder, M., D. Schindler, T. Essington and A. Litt, 2009b: Disrupted seasonal clockwork in the population dynamics of a freshwater copepod by climate warming. *Limnology and Oceanography*, 54 (6), 2493-2505, doi:10.4319/lo.2009.54.6_part_2.2493.

Winslow, L., J. Read, G. Hansen and P. Hanson, 2015: Small lakes show muted climate change signal in deepwater temperatures. *Geophysical Research Letters*, 42 (2), 355-361, doi:10.1002/2014gl062325.

Woodward, F. I., M. R. Lomas and C. K. Kelly, 2004: Global climate and the distribution of plant biomes. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences*, 359 (1450), 1465-1476, doi:10.1098/rstb.2004.1525.

Woodward, G., D. M. Perkins and L. E. Brown, 2010: Climate change and freshwater ecosystems: impacts across multiple levels of organization. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 365 (1549), 2093-2106, doi:DOI 10.1098/rstb.2010.0055.

Worth, J. R. P. et al., 2016: Gondwanan conifer clones imperilled by bushfire. *Scientific Reports*, 6, doi:10.1038/srep33930.

Wu, M. et al., 2016: Vegetation-climate feedbacks modulate rainfall patterns in Africa under future climate change. *Earth System Dynamics*, 7 (3), 627-647, doi:10.5194/esd-7-627-2016.

Wuebbles, D. J., D. W. Fahey and K. A. Hibbard, 2017: Climate science special report: fourth national climate assessment, volume I.

Xenopoulos, M. A. et al., 2005: Scenarios of freshwater fish extinctions from climate change and water withdrawal. *Global Change Biology*, 11 (10), 1557-1564, doi:DOI 10.1111/j.1365-2486.2005.01008.x.

Xu, L. et al., 2017: Spatial Distribution of Carbon Stored in Forests of the Democratic Republic of Congo. *Scientific Reports*, 7, doi:10.1038/s41598-017-15050-z.

Xu, X. et al., 2018: Tree cover shows strong sensitivity to precipitation variability across the global tropics. *Global Ecology and Biogeography*, 27 (4), 450–460.

Xue, K. et al., 2016: Tundra soil carbon is vulnerable to rapid microbial decomposition under climate warming. *Nature Climate Change*, 6 (6), 595.

Xue, Q., M. Z. Majeed, W. Zhang and C. S. Ma, 2019: Adaptation of Drosophila species to climate change - A literature review since 2003. *Journal of Integrative Agriculture*, 18 (4), 805-814, doi:10.1016/s2095-3119(18)62042-8.

Yang, Y. et al., 2016: Long-term CO2 fertilization increases vegetation productivity and has little effect on hydrological partitioning in tropical rainforests. *Journal of Geophysical Research: Biogeosciences*, 121 (8), 2125-2140.

Yang, Y. et al., 2018: Post-drought decline of the Amazon carbon sink. *Nature Communications*, 9, doi:10.1038/s41467-018-05668-6.

Yom-Tov, Y., 2001: Global warming and body mass decline in Israeli passerine birds. *Proceedings of the Royal Society of London Series B-Biological Sciences*, 268 (1470), 947-952.

Yom-Tov, Y. and S. Yom-Tov, 2004: Climatic Change and Body Size in Two Species of Japanese Rodents. *Biological Journal of the Linnean Society*, 82 (2), 263-267.

Yosef, G. et al., 2018: Large-scale semi-arid afforestation can enhance precipitation and carbon sequestration potential. *Scientific Reports*, 8, doi:10.1038/s41598-018-19265-6.

Young, A. M., P. E. Higuera, P. A. Duffy and F. S. Hu, 2017a: Climatic thresholds shape northern high-latitude fire regimes and imply vulnerability to future climate change. *Ecography*, 40 (5), 606-617, doi:10.1111/ecog.02205.

Young, D. J. N. et al., 2017b: Long-term climate and competition explain forest mortality patterns under extreme drought. *Ecology Letters*, 20 (1), 78-86, doi:10.1111/ele.12711.

Young, K. R., A. G. Ponette-Gonzalez, M. H. Polk and J. K. Lipton, 2017c: Snowlines and Treelines in the Tropical Andes. *Annals of the American Association of Geographers*, 107 (2), 429-440, doi:10.1080/24694452.2016.1235479.

Yu, Y. et al., 2017a: Observed positive vegetation-rainfall feedbacks in the Sahel dominated by a moisture recycling mechanism. *Nature Communications*, 8, doi:10.1038/s41467-017-02021-1.

Yu, Z. et al., 2017b: Global gross primary productivity and water use efficiency changes under drought stress. *Environmental Research Letters*, 12 (1), doi:10.1088/1748-9326/aa5258.

Yuan, W. P. et al., 2019: Increased atmospheric vapor pressure deficit reduces global vegetation growth. *Science Advances*, 5 (8), doi:10.1126/sciadv.aax1396.

Zaehle, S., 2013: Terrestrial nitrogen - carbon cycle interactions at the global scale. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 368 (1621), doi:10.1098/rstb.2013.0125.

Zaloumis, N. P. and W. J. Bond, 2016: Reforestation or conservation? The attributes of old growth grasslands in South Africa. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371 (1703), 20150310.

Zelazowski, P. et al., 2011: Changes in the potential distribution of humid tropical forests on a warmer planet. *Philosophical Transactions. Series A, Mathematical, Physical, and Engineering Sciences*, 369 (1934), 137-160, doi:10.1098/rsta.2010.0238.

Zemp, D. C. et al., 2017: Self-amplified Amazon forest loss due to vegetation-atmosphere feedbacks. *Nature Communications*, 8, doi:10.1038/ncomms14681.

Zeng, N., J. D. Neelin, K. M. Lau and C. J. Tucker, 1999: Enhancement of interdecadal climate variability in the Sahel by vegetation interaction. *Science*, 286 (5444), 1537-1540, doi:10.1126/science.286.5444.1537.

Zeng, Z. et al., 2017: Climate mitigation from vegetation biophysical feedbacks during the past three decades. *Nature Climate Change*, 7, 432, doi:10.1038/nclimate3299 https://www.nature.com/articles/nclimate3299#supplementary-information.

Zeng, Z. Z., L. Q. Peng and S. L. Piao, 2018a: Response of terrestrial evapotranspiration to Earth's greening. *Current Opinion in Environmental Sustainability*, 33, 9-25, doi:10.1016/j.cosust.2018.03.001.

Zeng, Z. Z. et al., 2018b: Impact of Earth Greening on the Terrestrial Water Cycle. *Journal of Climate*, 31 (7), 2633-2650, doi:10.1175/jcli-d-17-0236.1.

Zhang, C., M. Jansen, L. De Meester and R. Stoks, 2016: Energy storage and fecundity explain deviations from ecological stoichiometry predictions under global warming and size-selective predation. *Journal of Animal Ecology*, 85 (6), 1431-1441, doi:10.1111/1365-2656.12531.

Zhang, C. and W. Ren, 2017: Complex climatic and CO2 controls on net primary productivity of temperate dryland ecosystems over central Asia during 1980–2014. *Journal of Geophysical Research: Biogeosciences*, 122 (9), 2356–2374.

Zhang, W. et al., 2019: Ecosystem structural changes controlled by altered rainfall climatology in tropical savannas. *Nature Communications*, 10 (1), 671.

Zheng, H. X., F. H. S. Chiew, S. Charles and G. Podger, 2018: Future climate and runoff projections across South Asia from CMIP5 global climate models and hydrological modelling. *Journal of Hydrology-Regional Studies*, 18, 92-109, doi:10.1016/j.ejrh.2018.06.004.

Zhou, L. et al., 2014: Widespread decline of Congo rainforest greenness in the past decade. *Nature*, 509 (7498), 86.

Zhou, S., Y. Zhang, A. P. Williams and P. Gentine, 2019: Projected increases in intensity, frequency, and terrestrial carbon costs of compound drought and aridity events. *Science Advances*, 5 (1), doi:10.1126/sciadv.aau5740.

Zhou, Z. et al., 2015: Responses of alpine grassland to climate warming and permafrost thawing in two basins with different precipitation regimes on the Qinghai-Tibetan Plateaus. *Arctic, Antarctic, and Alpine Research*, 47 (1), 125-131.

Zhu, Z. et al., 2016: Greening of the Earth and its drivers. *Nature Climate Change*, 6 (8), 791.

Zimmer, A. et al., 2018: Time lag between glacial retreat and upward migration alters tropical alpine communities. *Perspectives in Plant Ecology, Evolution and Systematics*, 30, 89-102, doi:https://doi.org/10.1016/j.ppees.2017.05.003.

Zimov, N. S. et al., 2009: Carbon storage in permafrost and soils of the mammoth tundra-steppe biome: Role in the global carbon budget. *Geophysical Research Letters*, 36 (2).

Zomer, R. J. et al., 2014: Environmental stratification to model climate change impacts on biodiversity and rubber production in Xishuangbanna, Yunnan, China. *Biological Conservation*, 170, 264-273.

Zscheischler, J. and S. I. Seneviratne, 2017: Dependence of drivers affects risks associated with compound events. *Science Advances*, 3 (6), doi:10.1126/sciadv.1700263.

Zscheischler, J. et al., 2018: Future climate risk from compound events. *Nature Climate Change*, 8 (6), 469-477, doi:10.1038/s41558-018-0156-3.

Zubkova, M., L. Boschetti, J. T. Abatzoglou and L. Giglio, 2019: Changes in Fire Activity in Africa from 2002 to 2016 and Their Potential Drivers. *Geophysical Research Letters*, 46 (13), 7643-7653, doi:10.1029/2019gl083469.

Zumbado-Ulate, H., F. Bolanos, G. Gutierrez-Espeleta and R. Puschendorf, 2014: Extremely Low Prevalence of Batrachochytrium dendrobatidis in Frog Populations from Neotropical Dry Forest of Costa Rica Supports the Existence of a Climatic Refuge from Disease. *EcoHealth*, 11 (4), 593-602, doi:10.1007/s10393-014-0967-2.

Large Tables

Table 2.2: Global Fingerprints of Climate Change Impacts across Wild Species. Updated from Parmesan and Hanley 2015 [PLACEHOLDER FOR SECOND ORDER DRAFT: update still in progress]. For each dataset, a response for an individual species or functional group was classified as (1) no response (no significant change in the measured trait over time), (2) if a significant change was found, the response was classified as either consistent or not consistent with expectations from local or regional climate trends. Percentages are approximate and estimated for the studies as a whole. Individual analyses within the studies may differ. The specific metrics of climate change analysed for associations with biological change vary somewhat across studies, but most use changes in local or regional temperatures (e.g., mean monthly T or mean annual T), with some using precipitation metrics (e.g. total annual rainfall). E.g. a consistent responses would be poleward range shifts in areas that are warming. Probability (P) of getting the observed ratio of consistent:not consistent responses by chance was $<10^{-13}$ for Parmesan & Yohe 2003, Root 2003, Root 2005 and Poloczanska 2013, and was <0.001 for Rosenzweig 2008 (source=publication) (Parmesan and Yohe, 2003; Root et al., 2003; Root et al., 2005; Rosenzweig et al., 2008; Poloczanska et al., 2013). Test were all binomial tests against p=0.5, performed by Parmesan

Study	N: total numbers of species and functional groups (n of studies)	Species in given system (%) Terrestrial (T) Marine (M) Freshwater (F)	Breakdown of types of change (% of species or studies in each category)	In each category, % (n) of species showing significant change	Changes consistent with local or regional climate change (% of all species that showed change regardless of type of change, *e.g.* includes phenological changes)	Geographical region	Study designed for attribution to climate change
(Parmesan and Yohe, 2003)	1598	T: 85.2% M: 13.5% F: 1.3%	58% distribution/abundance changes (n=920); 42% phenology (n=678)	Distribution/abundance: 50% (n=460/920); Phenology 87% (n=423)	[2]84%	global	Yes
(Root et al., 2003)	1468	T: 94% M: 5.4% F: 0.6%	58% on dist/abun (n=926)	52 % of distribution (n=483/926)	[2]82.3%	global	yes
(Rosenzweig et al., 2008)	55 studies (~100-200 species)	[1]T: 65% [1]M: 13% [1]F: 22%	33% (n=18 studies)		[3]90%	global	
(Poloczanska et al., 2013)	857 = 1,735 sp x trait combinations[3]	T: 0% M: 100% F: 0%	80% (n=1060/1323 total distributions + abundances)[3]	63% (279/446 total for distributions)	[2]83%	global	yes
(Bowler et al., 2017)	1,167	T 680 M 216 F 271	abundance; population trends	T: 48% M: 61% F: 35%	n/a	Europe	

(Cohen et al., 2018)	127 studies	T 100% (animals)	phenology	not given	81% of 127 studies of animals show phenological change in direction of earlier spring. Some studies were multi-species	Europe North America Australia Japan	n/a
(Thackeray et al., 2010)	726 taxa: vertebrates invertebratesplants	T not given M not given F not given	phenology	not given	83.8% of species 92.5% T plants 62.2% F plants	UK	n/a
(Chen et al., 2011)	24 taxonomic group x region combinations for latitude, 31 for elevation	T >264 M>10 F >34	range shifts: elevation and latitude	not given	not given	global	no
(Ficetola and Maiorano, 2016)	n= 66 studies temperature effects; 15 precipitation	T/F 100% (amphibians)	phenology and abundances	not given	not given	global	n/a
(Mason et al., 2015)	16 taxonomic groups	T inverts & herptiles F (caddisflies, aquatic bugs)	range shifts/ 2 time periods	not given (can't access supplemental data tables)	not given	UK	yes
(Halupka and Halupka, 2017)	54 species	T 100% (birds)	phenology: Length of breeding season		shows differences in sign of response between single & multi-broods & migrant vs resident	northern hemisphere	yes
(Kharouba et al., 2018)	108 species in 54 pair-wise interactions	Not given	phenology of predator/prey interactions		86% of 108 species with phenological advance, 1% no change, 14% retarding	global	partial: assumes asynchrony should increase

(Pacifici et al., 2017)	70 studies of 120 mammals; 66 studies of 569 birds	T 100% (birds and mammals)	multiple: range change, abundance, reproductive rate, survival, body mass	not given	aim is to ask whether impacts are negative in any sense, including range contraction and/or reduced reproductive rates, or other measures of fitness estimates past change for IUCN threatened sp based on actual obs change in more common sp;	global for bird; mammals N America	unclear - complex methods

Table 2.3: Confidence in detecting and attributing observed changes in terrestrial and freshwater species and systems to climate change. [[PLACEHOLDER FOR SECOND ORDER DRAFT: table to be expanded with continued literature search]. Summary table is fully detailed in Table 2.A.1. Lines of evidence for attribution of observed changes to climate change and increased CO_2 are used to support stated confidence in attribution of key statement on observed biological changes to climate change and increased atmospheric CO_2. Icons represent lines of evidence.

Lines of evidence: Paleo data: 🦴; Experiments: 🧪; Long-term observations: 🔭 ; Fingerprint of climate change response: 🫆; Models 🖧 ; Complex statistical analysis:

Key statement	**Region**	**Period**	**Lines of evidence**	**Climate change attribution**
About half of all species where land use change has been minimal have shifted their ranges, with 80-90% of movements being in the direction expected from regional warming trends - i.e. poleward and upward.	Global	Range 20 - 260 years	🦴 🧪 🔭 🫆 🖧 📊	*high evidence high agreement* *very high confidence*
Downward shifts and east-west shifts (shown for trees and birds) have been associated with regional increases in precipitation where precipitation has been shown to be the principle driver of a range boundary	USA	~ 40-60 years	🔭 📊 🖧	*low evidence,* *high agreement,* *medium confidence*
About 2/3 of all species with long-term (>20 years) records have shifted the timing of spring events in directions expected from regional winter and spring warming.	Global	Varies by study. Range: 20 - 400 years	🧪 🔭 🫆 🖧 📊	*high evidence* *high agreement* *very high confidence*
Winter chilling-depending species have delayed or not changed spring events despite spring warming countered by winter warming. When these species are taken into account, it is estimated that 92% of species in these studies have responded to regional warming trends	Northern Europe and USA	Varies by study. Range = 26-46 years	🧪 🔭 🖧 📊 🫆	*medium evidence* *high agreement* *high confidence*

Wildfire expansion due to climate-driven increased aridity	Western North America	1979-2017	🔭 🖧 📈	*high evidence* *high agreement* *high confidence*
Tree mortality caused by climate-driven (increases in temperature and reduced precipitation) increasing aridity	North America and Africa	ca. 1945-2007	🔭 📈	*medium evidence* *high agreement* *medium confidence*
Biomes have shifted upwards in latitude and elevation at 19 sites in boreal, temperate, and tropical ecosystems	Global	1500-2008	⛏ 🔭 📈 🖧	*high evidence* *high agreement* *high confidence*
Beetles & moths shifting poleward and upward has brought new pest species into some forests; warming winters and longer growing season has increased destructive outbreaks of beetles and moths in temperate and boreal forests	North America, Europe	Varies by study	⛏ 🔭 🧪	[PLACEHOLDER FOR SECOND ORDER DRAFT]
Exotic species are responding differently from native species in both abundance changes and phenological changes, but not in a consistent fashion	North America		🔭	*low/medium evidence* *low agreement*
The polar bear has shown range contractions, population declines and increased hybridization with the brown bear driven by declines in sea ice and related prey species [PLACEHOLDER FOR SECOND ORDER DRAFT: in process of gathering literature for other species that are most cold-adapted - e.g. sea-ice dependent, mountain-top restricted, upper headwaters, coldest lakes]	Arctic		⛏ 🔭 🧪	[PLACEHOLDER FOR SECOND ORDER DRAFT]
chytrid fungus has been impacted by climate change and has locally increased or decreased depending upon local climatic conditions and recent climate trends [PLACEHOLDER FOR SECOND ORDER DRAFT: to be expanded to other diseases of wildlife, with potential for human disease risk where appropriate]	Global		🔭 🧪 🖧 📈	[PLACEHOLDER FOR SECOND ORDER DRAFT]

Warming has amplified the trophic state lakes are already in. Eutrophic lakes have become more productive while oligotrophic lakes have become more nutrient limited	Global	Varies by study. Range 20-50 years	🧪 🔭 🫆 🖧	*High evidence* *high/mediun confidence*
Forest biome shifts. Boreal forests have shifted into arctic tundra, treeline has shifted upward into alpine tundra, temperate deciduous shrubs and forests moved upwards into conifer forest	Global	year 1500 to 2008	🟫 🔭 〰 🖧	*high evidence,* *high agreement,* *high confidence*
Woody encroachment into open (grassland, desert) systems has occurred, with climate change as one of the primary drivers -	Global		🟫 🔭 🧪 〰 🖧	[PLACEHOLDER FOR SECOND ORDER DRAFT]
Browning of boreal lakes has been driven by rising temperatures and changes in precipitation patterns that have increased dissolved organic matter (DOM) concentrations in freshwater systems	Boreal	past decades	🔭 🧪 🖧	*high evidence* *high confidence*

Table 2.4: Terrestrial and freshwater ecosystem feedbacks which affect the Earth's climate system dynamics; following (Prăvălie, 2018).

Perturbation	**Implications for Warming/Feedback Mechanism the Earth's Climate System Dynamics**
Phenological change	Increased primary productivity and plant growth with CO_2 fertilization (Mao et al., 2016; Wang et al., 2018) Increasing growing season length (Peñuelas et al., 2009; Barichivich et al., 2013) vegetation greening reduces diurnal temperature range through evapotranspiration (mid-latitudes) and albedo (high latitudes) (Jeong et al., 2011); CO_2 storage in biomass (cooling) (Keenan et al., 2014); Reduced albedo in snow-covered regions as canopies become taller and darker; increased evapotranspiration, a key component of the global water cycle and energy balance which influences global rainfall, temperature, and atmospheric motion (Zeng et al., 2017).
Insect outbreaks	Reduced carbon uptake and storage; Increased surface albedo (cooling) (Landry et al., 2016); increased CO2 emissions leading to net warming; decreased leaf area index and gross primary productivity (Ghimire et al., 2015) leading to reduced evapotranspiration and increased land surface temperature (Bright et al., 2013).
Range shifts	Reduced albedo in snow-covered regions as trees expand poleward (Chae et al., 2015); enhanced permafrost thawing; expansion of insect outbreak range increases forest impact (Pureswaran et al., 2018); biome dependent changes in albedo and evapotranspiration regimes (Naudts et al., 2016).
Die-off and large-scale mortality events	Decreased Gross primary productivity (GPP); decline in carbon storage; increased CO_2 emissions; increased solar radiation, reduced soil moisture, higher surface runoff; albedo effects (Lewis et al., 2011; Prăvălie, 2018)
Deforestation	Reduced carbon storage (Pugh et al., 2019); increase in (regional) surface air temperature due to reduced evaporation (less cooling); increased albedo in high-latitude systems (regional radiative cooling) (Loranty et al., 2014); increased air temperature and diurnal temperature variation (Alkama and Cescatti, 2016), locally and globally (Winckler et al., 2019); reduced precipitation (Perugini et al., 2017)
Forest degradation	Reduced carbon storage (de Paula et al., 2015; Bustamante et al., 2016; de Andrade et al., 2017; Mitchard, 2018)
Fragmentation	Carbon losses because biomass is less developed in forest edges (Pütz et al., 2014; Chaplin-Kramer et al., 2015; Haddad et al., 2015)
Air pollution	Decreased plant productivity, transpiration and carbon sequestration in forest with lower biomass due to ozone toxicity (Sitch et al., 2007; Ainsworth et al., 2012); increased productivity due to increase in diffuse solar radiation caused by terrestrial aerosols
Declining populations of megafauna	Changes to physical and chemical properties of organic matter, soils and sediments influence carbon uptake and storage (Schmitz et al., 2018); increased or decreased carbon storage biomass and carbon storage, with differences across biomes determined by floristic structure and animal size (Bello et al., 2015; Osuri et al., 2016; Peres et al., 2016; Berzaghi et al., 2018; Schmitz et al., 2018; He et al., 2019)
Fire	Increased carbon and aerosol emissions(van der Werf et al., 2017); surface warming (Liu et al., 2019c); albedo effect dependent on ecosystem and species-level traits (Rogers et al., 2015; Chen et al., 2018) (initial albedo decrease post-fire; increased albedo where snow exposure is increased by canopy removal and species composition change during recovery); black carbon deposition on snow and sea ice (short-term) (Randerson et al., 2006); indirect increases in carbon emissions due to soil erosion (Caon et al., 2014)
Change in forest composition	Reduced carbon storage due to decline in biomass (McIntyre et al., 2015)
Woody encroachment in non-forested ecosystems	Reduced production, enhanced water use, reduced albedo and altered land-atmosphere feedbacks. Uncertain feedbacks to C cycle (some suggest it is an increase, others suggest a decrease)
Net Primary Productivity (NPP) shifts	Reduced albedo following high-latitude expansion of trees caused by photosynthetic enhancement of growth; increase photosynthesis and net ecosystem production (NEP) (Fernandez-Martinez et al., 2019); increased NPP in N-limited ecosystems due to increased nitrogen deposition from agriculture and combustion (Du and de Vries, 2018; Schulte-Uebbing and de Vries, 2018); Nutrient limited lakes are likely to become less productive,

	while nutrient rich lakes are likely to become more productive due to warming induced prolongation of stable stratification (Adrian and Hessen, 2016; Kraemer et al., 2017).
Biogeochemical shifts	Decline in carbon storage due to nitrogen limitation (Reich et al., 2014; Wieder et al., 2015b); Increased carbon storage (Peñuelas et al., 2013); Increase CO2 emissions from dried river beds or lake shores caused by extended droughts (Marcé et al., 2019).

Table 2.5: Projected vulnerabilities and risks of ecosystems to biome shifts from spatial analyses of vegetation biogeography, in order by type of analysis, analysis area, and projected change in temperature. This table updates the assessment in Betts (Betts et al., 2015). Gonzalez (Gonzalez et al., 2010a) and Eigenbrod (Eigenbrod et al., 2015) conducted analyses on equal-area spatial data; others did not analyze pixels of equal area. Authors generally reported biome change estimates directly as fraction of land area; for authors that did not directly report the biome change, changes are estimates from authors' maps and are indicated by a tilde (~). Authors generally analyzed changes from ~1990 to 2100, except for Heubes (2011) and Li (Li et al., 2018b), who projected changes from 2000 to 2050, and Aleman (Aleman et al., 2017) and Rasquinha and Sankaran (Rasquinha and Sankaran, 2016), who projected changes from 2000 to 2070.

Area	ΔT (ºC)	Emissions scenario	GCMs	Vegetation model	biomes	Spatial resolution (km)	Biome change, fraction of area (%)	Criterion	Ref.
Dynamic global vegetation models									
World	1	RCP2.6	3	Hybrid, JeDi, JULES, LPJmL, ORCHIDEE, SGVM, VISIT	5-14	~50	~4	Gamma >0.3	(Warszawski et al., 2013)
World	1.5	1.5° C	16	LPJ	2	~150	~5	P >0.80	(Scholze et al., 2006)
World	≤2	B1	12	LPJ	8	~50	7	|change| >30%	(Park et al., 2015)
World	2.4	B1	3	MC1	13	50	10	confidence >0.8	(Gonzalez et al., 2010a)
World	2.5	+2-3° C	16	LPJ	2	~150	~5	P >0.80	(Scholze et al., 2006)
World	2	RCP4.5	3	Hybrid, JeDi, JULES, LPJmL, ORCHIDEE, SGVM, VISIT	5-14	~50	13	Gamma >0.3	(Warszawski et al., 2013)
World	3	RCP6.0	3	Hybrid, JeDi, JULES, LPJmL, ORCHIDEE, SGVM, VISIT	5-14	~50	28	Gamma >0.3	(Warszawski et al., 2013)

World	2.5-3.5	A1B	18	LPJ	8	~50	10	|change| >30%	(Park et al., 2015)
World	3.4	A1B	3	MC1	13	50	13	confidence >0.8	(Gonzalez et al., 2010a)
World	3.5	3.5° C	16	LPJ	2	~150	~5	P >0.80	(Scholze et al., 2006)
World	≥3.5	A2	18	LPJ	8	~50	13	|change| >30%	(Park et al., 2015)
World	4	A2	3	MC1	13	50	16	confidence >0.8	(Gonzalez et al., 2010a)
World	3.1-4.7	historical climate and B1, A1B, A2	3	MC1	13	50	12	confidence >0.8	(Gonzalez et al., 2010a)
World	~3.5-5.5	A1B	8	CLM	5	~280	~10-30	range of GCMs	(Alo and Wang, 2008)
World	4	RCP8.5	3	Hybrid, JeDi, JULES, LPJmL, ORCHIDEE, SGVM, VISIT	5-14	~50	35	Gamma >0.3	(Warszawski et al., 2013)
World	4.6	A1FI	1	HyLand	2	~250-375	~10	|change| >50%	(Sitch et al., 2008)
World	4.6	A1FI	1	LPJ	2	~250-375	~20	|change| >50%	(Sitch et al., 2008)
World	4.6	A1FI	1	ORCHIDEE	2	~250 x 375	~10	|change| >50%	(Sitch et al., 2008)
World	4.6	A1FI	1	TRIFFID	2	~250 x 375	~15	|change| >50%	(Sitch et al., 2008)
Africa	-	A1B	1	aDGVM	5	~30	~26	change in one GCM	(Scheiter and Higgins, 2009)
Asia - Qinghai-Tibetan Plateau	1.5	RCP4.5	1	LPJ	19	~50	55	change in one GCM	(Gao et al., 2016a)
Asia - Qinghai-Tibetan Plateau	4.2	RCP8.5	1	LPJ	19	~50	70	change in one GCM	(Gao et al., 2016a)

Asia - Siberia	2	+2.6° C after 130 y	-	FAREAST	2	372 sites	~5	|change| >50% of area	(Shuman et al., 2011)
Europe	2.9-4.9	A2	2	LPJ-GUESS	13	~12 x 18	~30-40	change in one GCM	(Hickler et al., 2012)
South America - Amazon	2	A2	1	HadCM3LC	2	~250 x 375	~30	change in one GCM	(Jones et al., 2009)
South America - Amazon	~3	RCP4.5	1	Inland	15	~190 x 125	~50	change in one GCM	(Lyra et al., 2016)
South America - Amazon	~6	RCP8.5	1	Inland	15	~190 x 125	~80	change in one GCM	(Lyra et al., 2016)
Equilibrium models									
World	1	RCP2.6	10	vulnerability index	14	~10	10	vulnerability index >0.7	(Li et al., 2018a)
World	1.8	RCP4.5	10	vulnerability index	14	~10	12	vulnerability index >0.7	(Li et al., 2018a)
World	3.7	RCP8.5	10	vulnerability index	14	~10	15	vulnerability index >0.7	(Li et al., 2018a)
World	2–4	A1B	10	EVE	5	~100	37	average of GCMs	(Bergengren et al., 2011)
Africa - South		A1B	1	aDGVM	7	~20	50	change in one GCM	(Moncrieff et al., 2015)
Africa - West		A2	17	GAM	5	~10	~50	weighted average of GCMs	(Heubes et al., 2011)
Asia - India	3	+3ºC, +15% precipitation	1	Minimum distance supervised classification	7	1	~25	change in one scenario	(Chakraborty et al., 2013)

Asia - India		RCP4.5	19	RF	11	~10	14	agreement >0.75	(Rasquinha and Sankaran, 2016)
Asia - India		RCP8.5	17	RF	11	~10	18	agreement >0.75	(Rasquinha and Sankaran, 2016)
North America - Northwest		Historical climate and A2	2	Rehfeldt	33	~1	50-57	change in one GCM	(Langdon and Lawler, 2015)
North America - Yukon	3.9-6.9	A2	5	SNAP-EWHALE	25	~18	50	two projected changes in biome	(Rowland et al., 2016)
South America		A2	14	CPTEC-PVM2	13	~170	~5-40	confidence >0.75	(Lapola et al., 2009)
Tropical forests	2	+2° C	16	MWCD	2	~100	<5	P >0.80	(Zelazowski et al., 2011)
Tropical forests	4	+4° C	16	MWCD	2	~100	~5	P >0.80	(Zelazowski et al., 2011)
Combined climate change and land use change									
World	1	RCP2.6		LPJmL	9	~50	22	Gamma >0.3	(Ostberg et al., 2018)
World	1.8	RCP4.5		LPJmL	9	~50	34	Gamma >0.3	(Ostberg et al., 2018)
World	2.2	RCP6.0		LPJmL	9	~50	41	Gamma >0.3	(Ostberg et al., 2018)
World	3.7	RCP8.5		LPJmL	9	~50	54	Gamma >0.3	(Ostberg et al., 2018)
World	3.1-4.7	historical climate and B1, A1B, A2		MC1	13	48	48	confidence >0.8	(Eigenbrod et al., 2015)
Latin America	1	RCP2.6	5	LPJmL	9	~50	8-14	average of GCMs	(Boit et al., 2016)

Latin America	3.7	RCP8.5	5	LPJmL	9	~50	10-15	average of GCMs	(Boit et al., 2016)

Table 2.7: Evidence to support proposed climate change adaptation measures for biodiversity.

Proposed Adaptation Measures for Biodiversity	Confidence Assessment (tentative - to be reviewed for SOD)	Comment	Selected References
Increase connectivity in terrestrial habitats – corridors, stepping stones	*Medium evidence Medium agreement*	Good evidence that some species move more quickly in more connected landscapes. However, not all species do and some species that benefit are invasive / pest / disease species.	(Keeley et al., 2018)
Increase connectivity in river networks	[PLACEHOLDER FOR SECOND ORDER DRAFT]		(Abell et al., 2017; Brooks et al., 2018)
Increase habitat patch size site and expand protected area	*Limited evidence High agreement*	Generally increase resilience because of functioning natural processes, large species populations and refugial areas	(Eigenbrod et al., 2015; Oliver et al., 2015a)
Increase replication and representation of protected areas	*limited evidence, high agreement*	Various benefits inferred, including, wider range of climatic and other conditions, less risk of extreme events affecting many rather than few areas. More sites available for colonisation by range expanding species and better conditions to maintain species in situ under range contraction.	(Mawdsley et al., 2009; Thomas et al., 2012; Virkkala et al., 2014; Gillingham et al., 2015)
Protect microclimatic refugia	*Medium evidence High agreement*	Locally cool areas can be identified and there is evidence species can survive better in such areas.	(Haslem et al., 2015; Suggitt et al., 2015; Morelli et al., 2016; Bramer et al., 2018; Suggitt et al., 2018)
creating shade to lower temperatures for vulnerable species	*limited evidence, high agreement*		(Broadmeadow et al., 2011; Lagarde et al., 2012; Patino-Martinez et al., 2012; Thomas et al., 2016)

Restoring hydrological processes of wetlands and rivers, including by raising water tables and restoring original channels of watercourses,	[PLACEHOLDER FOR SECOND ORDER DRAFT]	Wetland restoration is well established as a conservation measure in some countries. Can reduce vulnerability to drought with climate change but getting evidence to demonstrate effectiveness as an adaptation measure evidence requires long term monitoring of a range of sites. Little restoration of degraded tropical peatlands to date	(Timpane-Padgham et al., 2017; Moomaw et al., 2018)
Reinstating natural fire regimes	[PLACEHOLDER FOR SECOND ORDER DRAFT]		
Trophic rewilding	[[PLACEHOLDER FOR SECOND ORDER DRAFT]	Reduce large scale woody encroachment of savannahs	(Stevens et al., 2016; Bakker and Svenning, 2018; Olofsson and Post, 2018)
Reduce non-climatic stressors to increase resilience of ecosystems	*limited evidence Medium agreement*	As a general principal climate change is recognised as a 'threat multiplier' but specific details are often unclear	(Oliver et al., 2017)
Increase water supply for ecosystems by improved irrigation and abstraction regimes	[PLACEHOLDER FOR SECOND ORDER DRAFT]		
Woody encroachment control in savannah areas	[PLACEHOLDER FOR SECOND ORDER DRAFT]	e.g., using goats to control woody encroachment`	(Coffman et al., 2014; Valkó et al., 2014; Batáry et al., 2015; Smit et al., 2016; Fulbright et al., 2018)
Replace livestock with native ungulates (particularly non-ruminants) to promote biodiversity	[[PLACEHOLDER FOR SECOND ORDER DRAFT]		(Hempson et al., 2017; Cromsigt et al., 2018)
Assisted translocation and migration of species	*Limited evidence*		(Brooker et al., 2018)
Intensive management for specific species	*Limited evidence*	A variety of approaches including manipulating microclimate and competition between species.	(Greenwood et al., 2016)
Ex-situ conservation (seedbanks/genetic stores, etc.)	Not possible to assess effectiveness at the present time	Seed banks have been established but long term effectiveness could only be evaluate at a later point.	(Christmas et al., 2016)

adjusting conservation objectives of protected areas to reflect changing species distributions and habitat characteristics	*Limited evidence*	(Rannow et al., 2014)

Table 2.8: Examples of widely advocated Ecosystem Based Adaptation measures with assessments of confidence.

Ecosystem Based Adaptation Measures	[PLACEHOLDER FOR SECOND ORDER DRAFT: **Confidence Assessment]**	**Ecosystem Service**	**Climate Change Impact**	**Social Benefits**	**Comment / Applicable Biome / Ecosystem**	**Selected References**
Natural Flood risk management in river systems – restoring wetlands, restoring natural river courses (removing canalisation) planting of trees – to slow flow and store water out of urban areas	*Medium evidence medium agreement*	Flood regulation through restoring riparian/wetland integrity	Increase rainfall intensity	Reduction of damage from flood		(Iacob et al., 2014)
Planting shade trees and restoration to maintain cool areas of rivers to maintain fisheries	[PLACEHOLDER FOR SECOND ORDER DRAFT]	food provision	warmer water temperatures	food supply, rural economy		(Broadmeadow et al., 2011) (Isaak et al., 2015) (Thomas et al., 2016)
Creation of sponge cities	[PLACEHOLDER FOR SECOND ORDER DRAFT: Still needs testing]	Flood regulation through implementation of sponge infrastructure to soak up excessive rain water	Increased rainfall intensity or reduced rainfall intensity	Runoff retention, flood control, groundwater recharge, stormwater reuse		(Liu, 2016)
Managed realignment of coastlines, re-establishing coastal habitats including mangroves, salt marsh.	*Robust evidence High agreement*	coastal flood protection	rising sea level and increasing storm energy	protection of life, property and livelihoods		(Spalding et al., 2014) (Høye et al., 2013) (Powell et al., 2019)
Use of shade trees in agriculture	*medium evidence, medium agreement,*	local climate regulation for agriculture	high temperature	food supply		(Verburg et al., 2019) (Nesper et al., 2019) (Blaser et al., 2018)

Urban green space – trees, parks, local nature reserves - to provide urban cooling	*robust evidence, high agreement*	local climate regulation	higher temperatures and heatwaves	cooler microclimates for people		(Aram et al., 2019) (Norton et al., 2015)
Rewilding to reduce intense fires	*High evidence, medium confidence*	Using biomass removal by herbivores to reduce extreme fire events, whilst restoring natural fire regimes	Mega-fires from increases in drought/ heat	Reduce infristructire damage from fires/ prevent loss of life	Flammable ecosystems with historic herbivore presence	(Johnson et al., 2018) (Waldram et al., 2008)
Alien plant control to improve water security	*High evidence, high agreement*	Water provision	Increasing droughts	Job creation through clearing, improved water security	Water scarce regions prone to an increase in droughts	(van Wilgen and Wannenburgh, 2016)
Woody plant control (of encroaching biomass) in open grassy ecosystems	*Medium evidence, medium agreements*	Grass biomass/ Grazing creation through removal of encroaching trees which causes a loss of grazing (outshading of grass)	Elevated CO_2 increasing tree growth/ increases in rainfall promoting tree growth	Improved income through bush clearing , fuelwood supplies, restore grazing	Rangelands	(Haussmann et al., 2016)

Table 2.9: Proposed research priorities for terrestrial ecosystems in the face of global change (Doblas-Miranda et al., 2015).

	Effects of the interactions between drivers of global change on ecosystem functioning
1	To establish the role of the landscape mosaic on fire spread
2	To further research the combined effect of different drivers on biological invasions and pest expansion
3	To address the interaction between drivers of global change and recent forest management practices
	Monitoring and data assessment of ecosystem response to global change
4	To obtain more realistic information, at larger temporal and spatial scales, of the impacts of global change and ecosystem services to be used in models
5	To assess forest mortality events associated with climatic extremes (particularly drought)
	Managing ecosystems to enhance resilience
6	To focus global change research on identifying and managing vulnerable areas

7	To use the functional and life-history traits concepts to study resilience and community assembly after disturbance
8	To promote cross-disciplinary research to study the relationship between genotypic and phenotypic diversity as a source of forest resilience
	Embracing the link between ecosystem functions and services
9	To understand how forest management affects the balance between C storage and water resources at large spatial and temporal scales
10	To analyse the interplay between landscape-scale processes and biodiversity conservation along wide gradients of landscape complexity
	Scaling ecosystem dynamics in space and time under different scenarios
11	To refine predictive models by including interactions between global change drivers and socio-economic contexts
12	To use manipulative, interdisciplinary and multiscale experiments to understand forest–atmosphere feedbacks
13	To improve the representation of key mechanisms linking plant hydraulics with landscape hydrology

Appendix 2.A: Supplementary Material

Table 2.A.1: Attribution and assessment of uncertainties associated with key statements on observed impacts. [[PLACEHOLDER FOR SECOND ORDER DRAFT: Lines of evidence to be added with new literature]

			Evidence for non-climate/CO2 drivers	-	**LINES OF EVIDENCE for climate change (including increased atmospheric CO2) as primary driver of observed change**								**Agreement for climate change attribution**	**Confidence level**	**References**
Key statement	**Geographic region**	**Period**	**Non-climate/CO2 Driver Land Use Change: Evidence for changes in land use as driver of observed change**	**Non-climate/CO2 Driver: Other: Evidence for changes in other drivers as driver of observed change**	**TYPE = Paleo data**	**TYPE = Experiments**	**TYPE = Long-term Observations**	**TYPE = a Fingerprint of climate change response**	**TYPE = Outputs of models of expected current climate change response match observed trends; OR models with and without CC - which match observed best? ; outputs of models project future impacts that are consistent with observed changes**	**TYPE = Change in climate variable at relevant scale has been linked to GHG forcing**	**TYPE = multivariate statistical analysis**	**TYPE = Meta-analyses**			
About half of all species where land use change has been a minimal driver and with longterm (>20 years) of records have shifted their ranges, with 80-90% of movements being in the direction expected from regional warming trends -	Global	Varies by study. Range = 20 - 250 years, mean = xx years	Minimized by study designs (1)		Polewards and upward ranges shifts have been common responses to past major climatic shifts (2)	Translocation of temperature-limited species outside the historic range boundaries has been unsuccessful in the absence of warming and successful during	Yearly variability in polewards range boundaries for mobile birds and butterflies highly significantly correlated with annual temperature variability (4)	Very long-term records (>50 years) demonstrate "sign-switching" (5) in which a species poleward boundary shifts polewards during warming periods and towards the equator during	species distribution models, Phenological models, and other process-based models driven by climate parameters have high predictive power in back-casting observed	Yes. Warming seasonal and annual temperatures have been linked to GHG forcing at both regional and global scales (8)		multiple global meta-analyses of xx- xx species show from 40% to 60% of species in a given region or taxonomic group having shifted their poleward range boundary further poleward over	high level of evidence, high agreement,	*very high confidence*	(1) {Parmesn, 2003}; (2) {Coope, xx}; (3){refs); (4) {refs}; (5){ }, (6) {Parmesan, 2003}, (7) {Hill, xx}, {Chuine, xx}, (8) {WGI); (9) {Parmesan, 2003}, {Root, 2003}, {Rosenzweig, 2008}, + refs in section 2.2.4

i.e. poleward and upward. Conclusions from prior ARs are further supported with new literature for butterflies, birds, xxx. New studies document that other taxa are also exhibiting consistent reponses, including for freshwater fish, xxxxx						warming periods (3)		cooling periods (6)	distributional changes (7)			the past 20-120 years (9)			
Where precipitation has been shown to be the principle driver of a range boundary, regional increases in precipitation have been associated with downward shifts and east-west shifts (shown for trees and birds)	USA, other?														refs in section 2.4.2.1
About 2/3 of all species with long-term (>20 years) of records have shifted the timing of spring events in directions expected from regional winter and spring warming.	Global	Varies by study. Range = 20 - 400 years, mean = xx years	NA	Photoperiod is an important cue for some species, which would show up as either no change in phenology over time, or where both photoperiod and temperature are drivers, photoperiod cues may tend to counter temperature cues (1)	NA	Controlled experiments demonstrate that temperature has large effects on timing of spring events for many species (2)	Yearly variability in appearance times of birds and butterflies highly significantly correlated with spring temp variability (3)	Very long-term records (>50 years) demonstrate "sign-switching" (4) in which a species shifts to earlier spring events during warming periods and later spring events during cooler periods (5)	Phenological models based on temperature have high predictive power in back-casting observed phenological change (6)	Yes. Warming spring temperatures have been linked to GHG forcing at both regional and global scales (7)		multiple global meta-analyses all show from xx% to xx% of species in a given region or taxonomic group having shifted towards earlier spring timing in recent decades (8)	high level of evidence, high agreement,	*very high confidence*	(1){refs}; (2) {refs}; (3) {Sparks}, (4) {Parmesan, 2003}, (5)Crefs); (6) {refs}; (7) {WGI); (8) { cohen, 2018}, {Parmesan, 2003}, {Root, 2003}, {Rosenzweig, 2008}, Freshwater: {Blenckner et al, 2007}, {Adrian et al, 2006}, {Adrian et al, 2009} + refs in section 2.4.2.2

New studies that were not designed for CC attribution are consistent with earlier studies on attribution in demonstrating general poleward and upward shifts of species' ranges and earlier spring events in regions with significant warming. These changes have been documented in both plants and animals, in terrestrial, lake and rivers systems	Global	Varies by study. Range = xx - xx years, mean = xx years	Not assessed									yes (1)			(1) {Chen, 2011}, {Thackeray, 2012},
For species that require winter chilling, winter warming has countered spring warming, resulting in either delayed spring events or no change. When these species are taken into account, it is estimated that 92% of species in these studies have responded to regional warming trends	Northern Europe and USA	Varies by study. Range = xx-xxx years.	NA	Photoperiod and vernalization requirements interact add details) (1)	NA	orange tip, vernalization of plants (UEA group) - demonstrate high heritability (strong genetic basis). Metabolic pathways understood for some species (2)	Yearly variability in break of diapause and dormancy highly significantly correlated with variability of fall and winter temperatures (3)		Models based on seasonal temperature sensitivities of individual species have high predictive power in back-casting observed phenological change (4)	Yes. Fall and winter warming has been linked to GHG forcing at both regional and global scales (5)		none to date	medium evidence, high agreement	*high confidence*	(1) {Gill, 2015}; (2) {Stohlhanski, }, {UEA group}; (3) {Gotthard, }, {Cook, 2012}, {Cook, 2013}; (4) {Cook 2012}, {Cook, 2013}; (5) {WGI} + refs in section 2.4.2.3
Field and remote sensing measurements have detected increases in the area burned by wildfire in western North America, analyzed the contribution of the principal climate and non-climate factors, and attributed the wildfire increases mainly to increased aridity caused by anthropogenic climate change.	western North America	1979-2017	population density, roads, built area, analyzed but less important				Field and remote sensing measurements of burned area, weather station measurements of climate. Significant temperature increaase, summer precipitation decrease.		Numerical models of wildfire as a function of climate and non-climate variables, calibrated by historical data, run for actual observed values and compared to model runs in which temperature remains unchanged. Western US - nthropogenic	Increases in temperature and decreases in summer precipitation detected and attributed to anthropogenic greenhouse gases	Correlation of burned area to climate variables (temperature, precipitation, relative humidity, evapotranspiration) outweighed local human factors (population density, roads, and built-area)		high evidence, high agreement	*high confidence*	Abatzoglou and Williams 2016, Holden et al. 2018, Kirchmeier-Young et al. 2019, Mansuy et al. 2019 + refs in section 2.4.4.2

									climate change burned area 200%,. British Columbia - 2017 burned area 700-1100%						
Increases in aridity due to increases in temperature and reduced precipitation caused by anthropogenic climate change have caused tree mortality at sites in North America and Africa. [Numerous other cases aorund the world have been detected, but not formally attributed]	North America and Africa	ca. 1945-2007	multivariate and bivariate statistical analyses of population density, roads, timber harvesting, livestock grazing, increased tree density, fire suppression, toppling of large trees, analyzed but less important				Field surveys of trees, weather station measurements of climate; western U.S. - tree mortality doubled, African Sahel tree mortality 20%, Morocco tree mortality 45%; significant increases in temperature and decreases in precipitation				Correlation of climate factors significant, non-climate factors non-significant; canonical correlation analyses of climate and non-climate factors		medium evidence, high agreement	*medium confidence*	van Mantgem et al. 2009, Gonzalez 2001, Gonzalez et al. 2012, le Polain de Waroux and Lambin 2012 [many other cases detected (Allen et al. 2010, Allen et al. 2015, Bennett et al. 2015, Martínez-Vilalta and Lloret 2016, Greenwood et al. 2017, Hartmann et al. 2018) but not formally attributed + refs in section 2.4.4.3
Field surveys of vegetation at the biome level detected latitudinal and elevational biome shifts at 19 sites in boreal, temperate, and tropical ecosystems and attributed the shifts mainly to anthropogenic climate change	Global	1500-2008	some research conducted multivariate statistical analyses, population density, roads, other non-climate factors analyzed but less important; some research conducted in sites with no recorded human land use change				Field surveys of vegetation species and densities, significant temperature increases and precipitation changes			Increases in temperature and changes in precipitation detected and attributed to anthropogenic greenhouse gases	canonical correlation analyses of climate and non-climate factors		high evidence, high agreement	*high confidence*	Beckage et al. 2008, Brink 1959, Danby and Hik 2007, Devi et al. 2008, Dial et al. 2007, Gonzalez 2001, Gonzalez et al. 2010, Gonzalez et al. 2012, Kirdyanov et al. 2012, Kullman and Öberg 2009, Leonelli et al. 2011, Lloyd and Fastie 2003, Luckman and Kavanagh 2000, Millar et al. 2004, Payette and Filion 1985, Payette 2007, Peñuelas and

															Boada 2003, Settele et al. 2014, Suarez et al. 1999, Walther et al. 2005, Wardle and Coleman 1992 + refs in sections 2.4.3.2; 2.4.3.3; 2.4.3.4
beetles & moths shifting poleward and upward has brought new pest species into some forests	North America, *other?*	Varies by study. Range = xx=xx years.	Not directly assessed, but occurring in both areas of high LUC and protected areas			-		-	-	-	-	-			refs in section 2.4.3.3
shift in forest composition has occurred due to species-specific differences in response to increasing drought															{Anderegg, 2016}
Increased tree mortality has occurred globally, in boreal, temperate and tropical systems, in response to increased drought, wildfire and insect pest outbreaks	Global	Varies by study. Range = xx=xx years.	For many studies, land use change is an important driver. For some studies, LUC is minimal (1)	pest outbreaks are important drivers, but impacts have been exacerbated by heat/drought induced tree stress (2)	tree-rings show xxx (3)	controlled temperature experiments link warming winters to lower insect mortality, and increased growing season length to increased number of generations per year, which leads to large increases in insect abudances in late growing season (4)	yes - add detail (5)	??? can this been deduced from records > 50 years? Eg. evidence for tree mortality being higher than past 100 years? (6)	yes - add detail (7)	true for some studies, not others? Ie probably true for large regional studies, but not very local ones (8)		????	high evidence, high agreement	*high confidence*	refs in section 2.4.4.3

exotic species are responding differently from native species in both abundance changes and phenological changes, but not in a consistent fashion													low/medium evidence, low agreement		{Primack, },
The most-cold-adapted species have shown the large range contractions and population declines (Sea-ice dependent, mountain-top restricted, upper headwaters, coldest lakes)															
body size changes						-	-	-	-	-	-	-			
diseases wildlife/humans															
taxonomic-specific statements				n		-	-	-	-	-	-	-			
Warming has amplified the trophic state lakes are already in. Eutrophic lakes will become more productive while nutrient limitation will increase in oligotrophic lakes.	global	Varies by study. Range 20-50 years	agriculture, urbanisation			whole lake experiments, enclosure experiments	monitoring data past>40 years, remote sensing data	in nutrient poor lakes prolongation of thermal stratification limits nutrient entrainments via vertical mixing which leads to a reduction in algal biomass, while global warming reinforces eutrophication of already eutrophic lakes via oxygen depletion in the sediment near water layers which triggers release of nutrients previously bound in the sediment.	yes				high	*high / medium confidence level*	{Mooij et al. 2007}; {Adrian et al. 2009}, {De Somerpond Domis et al. 2013} {Kraemer et al. 2017}
Whole biome shifts have occurred. Boreal forests have shifted into arctic tundra, treeline	Global		Mixed. add detail												

has shifted upward into alpine tundra, temperate deciduous shrubs and forests upwards into conifer forest, xx															
Woody encroachment into open (grassland, desert) systems has occurred globally, with climate change as one of the primary drivers	global		yes - loss of browsing herbivores; fire suppression. Reviews of long term experiments demonstrate impacts (1)	yes - (2)	yes - emergence of grasslands after CO2 came down below ~500ppm (3)	Experiments manipulating CO2 benefit woody plants (4)	yes -Long-term fire and grazing trials show woody encroachment occurs even when land use is held constant or accounted for indicating a global driver. (5)		yes - indicating co2 driven increase in woody cover (6)	yes - add detail (7)		yes - consistent encroachment across all savannas (8)	*In development: high evidence that CC is one of the primary drivers, but LUC also primary driver. High amount of evidence (lots of studies) but medium agreement on CC attribution because of complex drivers*	*medium*	(1){Bakker et al 2016} {Bond and Midgley 2012} {Smit et al 2010} (3) {Ehleringer and Ceerling 2002}{Beerling and Osborne 2008}(4) {Kgope et al 2010}{Bond and Midgley 2000}{Polley et al 1997}{Hoffman n et al 2000}{Quirk et al 2019}(5) {Buitenwerf et al 2012}{Zhang et al 2019}{Venter et al 2018} (6){Scheiter et al 2018}{Moncrieff et al 2014}{Higgins and Scheiter et al 2009} (8) {Stevens et al 2017}
High arctic and high mountain tundra systems have generally experienced greater warming than adjacent regions - *statement on impacts in development*															
Widespread greening and shrubbification of tundra	High artic and mountain tundra	1900-				yes - network of warming experiments link warming to increases in shrub, grass and sedge species (4)	yes - satellite and long term repeat photos (5)			yes - IPCC		yes - widespread shrubbification (8)	high	*high*	(4) {Elmendorf et al. 2012a, 2012b, 2015}{Bjorkman et al. 2018, 2019} {Myers-Smith et al 2019}(5) {Tape et al. 2006}{ Phoenix and Bjerke 2016}

															(8) {Myers-Smith et al 2011}
Tropical forests															
Drought and warming induced diversity shifts in Mediteranean type ecosystems	Mediterranean ecosystems		insect outbreaks assoicated with drought (1); loss of fish species				yes - Field surveys of long term monitoring show reduced diversity or shift in functional due to increasing prevalence of extreme hot and dry weather often the post-fire regeneration phase(5)			yes - increase in extreme droughts in regions (8)			medium evidence changes are mediated by an increase in extreme droughts. Changes are not always direct but interact through altering the fire regime and post-fire recovery		(1) {Fettig et al. 2019}{ McIntyre et al. 2015}(5) {Fettig et al. 2019}{ McIntyre et al. 2015}{Stephenson et al. 2018} {Slingsby et al 2017}{{Harrison, LaForgia, and Latimer 2018}. {Smithers et al 2018} (8) {F. E. Otto et al. 2018}{Sousa et al. 2018} {AghaKouchak et al. 2014} {Robeson 2015}
deserts															
Med shrublands shifting to grasslands	Med ecosystems, arid shrublands		Human driven fragmentation and nitrogen deposition benefits grasses (1)				Long-term								(1) {Lambrinos 2006}{ Fenn et al. 2010}
terrestrial carbon stocks							Long term monitoring and remote sensing show grass invasions (5)								(5) {Young et al. 2019; Syphard, Brennan, and Keeley 2019, Jacobsen et al 2018}
Climate change leads to shifts in thermal regime of lakes	global	Varies by study. past >40 years	yes, trophic state, ice formation can affect mixing regime of lakes				monitoring data past>40 years	temperate polamictic lakes may become dimictic more frequently, dimictic lakes may become monomictic, monomictic lakes tend to become oligomictic	yes; see Kirillin 2010, Kirillin & Shatwell, 2016				high evidence that CC is one of the primary driver. Planktonic events can contribute to polymictic-dimictic regime shifts in temperate lakes	*medium*	{Kirillin,2010}, {Kirillin & Shatwell, 2016}, {Shatwell et a, 2016}

Rising temperatures and changes in precipitation pattern are key drivers of DOC concentrations in freshwater which cause browning in boreal lakes.	Boreal	past decades		ecosystem responses to enhanced DOC act in concert with atmospheric deposition of nitrogen and sulphur		strong hydrological modelling tools	monitoring data past decades		Analysis of a large dataset of boreal Norwegian watersheds indicated that a moderate increase in temperature (2°C average, downscaled Hadley scenario) with associated increase in precipitation and vegetation density increases DOC export substantially.				high evidence that CC is one of the primary driver. Planktonic events can contribute to polymictic-dimictic regime shifts in temperate lakes	*high*	{Weyhenmeyer et al, 2009}, {Larsen et al, 2011}, {Finstad et al, 2016}, {Creed et al, 2018}