Global Tipping Points

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Chapter 1.3 Tipping points in the biosphere

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Summary

This chapter assesses scientific evidence for tipping points across the biosphere, which comprises Earth's ecosystems. Human-driven habitat loss, pollution, exploitation and, increasingly, climate change are degrading ecosystems across the planet, some of which can pass tipping points beyond which a 'regime shift' to an alternative (and often less diverse or beneficial) ecosystem state occurs.

Evidence for tipping points emerges across many biomes. In forests, large parts of the Amazon rainforest could tip to degraded forest or impoverished savanna, while tipping in boreal forests is possible but more uncertain, and whether current temperate forest disturbance could lead to tipping is unclear. In open savannas and drylands, drying could lead to desertification in some areas, while in others encroachment by trees and shrubs could see these biodiverse ecosystems shift to a forested or degraded state. Nutrient pollution and warming can trigger lakes to switch to an algae-dominated low-oxygen state. Coral reefs are already experiencing tipping points, as more frequent warming-driven bleaching events, along with pollution, extreme weather events and diseases, tip them to degraded algae-dominated states. Mangroves and seagrasses are at risk of regional tipping, along with kelp forests, marine food webs and some fisheries, which are known to be able to collapse.

Together, these tipping points threaten the livelihoods of millions of people, and some thresholds are likely imminent. Stabilising climate is critical for reducing the likelihood of widespread ecosystem tipping points, but tackling other pressures can also help increase ecological resilience, push back tipping and support human wellbeing.

Key messages

- Evidence exists for tipping points in a variety of ecosystems, including forest dieback, tree and bush encroachment in savanna and grasslands, dryland desertification, lake eutrophication, coral reef die-off and fishery collapse.
- Several biomes (such as mangroves and the Amazon rainforest) are losing resilience and approaching key tipping thresholds, with current warming levels already triggering coral reef die-off tipping points in multiple regions.
- Ecosystem tipping points can be driven by many different drivers (including, but not limited to, climate change) that interact in complex ways across many species and feedbacks, making it harder to assess whether tipping points may be imminent.

Recommendations

- Reduce pressure on global ecosystems through the urgent phase-out of greenhouse gas emissions as well as tackling exploitation, habitat loss and pollution.
- Promote ecological resilience through adaptive management, ecosystem restoration and inclusive conservation, supporting sustainable livelihoods and rights for Indigenous peoples and local communities, and improved governance of land and oceans.
- Address deep uncertainties around feedbacks controlling ecosystem tipping and the impacts of increasingly extreme events, plant adaptability and spatial variability through more and better-integrated observations, experiments, and improved models.
- Invest in observations (field and remote sensing) and experiments to monitor and detect declining ecosystem resilience and potential early warning signals.
- Foster greater data sharing and international collaboration, and co-design research to bring together researchers across natural and social sciences and Global North and South, as well as Indigenous and traditional ecological knowledge.

1.3.1 Introduction

The Earth's biosphere describes the sum of all global ecosystems. It forms a key part of the Earth system, driving the many biogeochemical cycles that maintain the climate system and keep Earth habitable (Kump, Kasting, and Crane, 1999). Ecosystems are the complex systems composed of assemblages of living organisms and their physical environment at the local scale (e.g. an area of rainforest in the Brazilian state of Amazonas).

At a larger scale, they form regional groupings (e.g. Madeira-Tapajós moist forest ecoregion in Dinerstein et al., 2017), ecosystem functional groups (e.g. tropical/subtropical lowland rainforests), biomes (e.g. tropical-subtropical forests), and ultimately the whole biosphere (Keith et al., 2022). Humans are also an integral part of the biosphere, with social systems being so closely intertwined with ecosystems that they can be seen as joint 'social-ecological systems' in which the dynamics of both interact as a single complex adaptive system (Folke et al.,2016; 2021; Ellis et al., 2021).

Ecosystems are being globally degraded by multiple human-driven pressures. At the species level, one million animals and plants face extinction (IPBES, 2019). Extinctions are happening at up to 100 times natural background rates averaged over the last century, leading some to assess that the Earth has now entered the sixth mass extinction event in the nearly 4 billion years of life's history (Barnosky et al., 2011; Ceballos et al., 2015). The Living Planet Index indicates that populations are declining in around half of vertebrate species, with an average decline across all species of 69 per cent since 1970 (WWF, 2022). The key drivers of biodiversity loss in order of importance are land and sea use change, direct exploitation, climate change, pollution, and invasive alien species (IPBES, 2019; Maxwell et al., 2016). Climate change is not currently the leading driver, but will become a substantial threat with further warming (IPBES, 2019). Global warming moving from 1.5 to 2°C increases the number of species facing the loss of most of their ranges from 4 to 8 per cent for vertebrates (e.g. mammals), 8 to 16 per cent for plants, and 6 to 18 per cent for insects, while 3.2°C of warming would increase these to 26, 44, and 49 per cent respectively (Warren et al., 2018). Together these losses are harming many ecosystems' ability to function and so threatening the critical ecosystem services that humanity relies upon, including providing food, clean water, and removing ~31 per cent of human-emitted CO_2 (Friedlingstein et al., 2022)..

As with many other complex systems, ecosystems have been proposed to feature nonlinear changes such as tipping points, beyond which dramatic shifts to a different ecological state are expected, further threatening biodiversity and bio-abundance (Scheffer et al.,2001, 2009). Ecosystems are also subject to many co-stressors with complex interactions, with changing disturbance regimes eroding resilience (e.g. Nystrom et al., 2000; Folke et al., 2004) and making tipping points easier to reach (Willcock et al., 2023). However, complex ecological and social-ecological dynamics crossing multiple scales can make it hard to discern tipping thresholds in observations (Schröder et al., 2005; Hillebrand et al., 2020; Spake et al., 2022). Organisms have agency that enables complex network and spatial dynamics to emerge – with human agency making social-ecological systems particularly complex – making ecosystem tipping dynamics often more difficult to detect and project relative to more physical systems (Kéfi et al., 2022; Rietkerk et al., 2021; Bastiaansen et al., 2022). Furthermore, while ecosystem functions or composition can have threshold responses to biodiversity loss or environmental change, in many cases responses remain relatively linear (Cardinale et al., 2011; Meyer et al., 2017; Hodapp et al., 2018; Strack et al., 2022).

Tipping at the global biosphere scale has been discussed (Barnosky et al., 2012; Hughes et al., 2013; Lenton and Williams, 2013) but is deemed unlikely, with local ecosystem shifts globally aggregating to relatively linear changes in response to human-driven pressures (Brook et al., 2013; Montoya et al., 2017; Rockström et al., 2018). Empirical evidence for tipping has, though, been found in multiple ecosystems from the local to regional scale – for example, in lakes, coastal zones, marine food webs, rangelands and forests (Scheffer et al., 2001, 2009; Folke et al., 2004; Walker and Meyer, 2004; Brook et al., 2013; Rocha et al., 2015; regimeshifts.org), and model evidence suggests tipping is possible in some biomes across sub-continental scales (Armstrong McKay et al., 2022; Wang et al., 2023). As such, ecological tipping points remain a useful concept (alongside gradual and nonlinear change) in understanding and managing ecosystems, despite being sometimes hard to observe in practice (Lade et al., 2021; Spake et al., 2022; Norberg et al., 2022).

In this chapter we follow the wider section's tipping point definition to categorise proposed tipping systems (see Box 1.1). In ecology, the terms 'regime shift' and 'critical transition' have been used interchangeably with 'tipping points', despite differences in meaning (Dakos, 2019). A *regime shift* refers to a shift in the current state of an ecological or social-ecological system from one partially stable state to another that is often large, relatively sudden (depending on system size and feedback timescales) and long-lasting, and entails a reorganisation in the structure and functioning of the system (Biggs et al., 2009; Maciejewski et al., 2019; Cooper et al., 2020). A *critical transition* refers to an abrupt shift in a system that occurs at a specific critical threshold in external conditions (Scheffer et al., 2009). In this Chapter, we use *tipping event* to describe the crossing of a tipping point (which is equivalent to critical threshold), and *regime shifts* to describe the resulting changes that unfold (equivalent to critical transition above). *Resilience* – the ability of ecosystems to maintain functioning in response to change and regenerate in the face of shocks, sometimes adapting and transforming in the process – is also a key concept, with declining resilience being a potential precursor to tipping (see Section 1.6) (Folke et al., 2004, 2016).

1.3.2 Current state of knowledge on tipping points in the biosphere

In this section we assess available scientific literature relating to tipping points in the Biosphere, as summarised in Figure 1.3.1 and Table 1.3.1. We focus on the following biomes: forests, savannas, drylands, lakes, coastal ecosystems and marine environments.

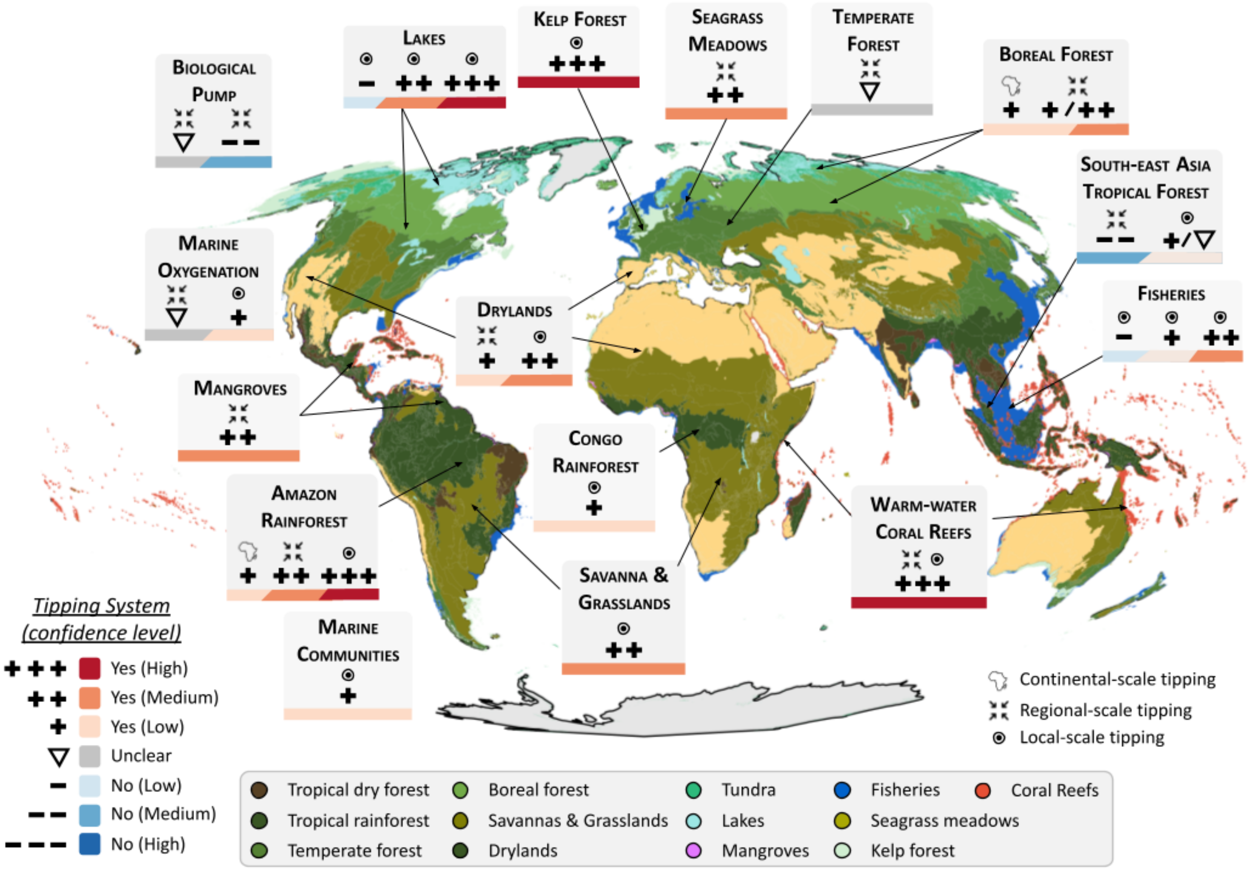

Figure 1.3.1: Map of biosphere systems considered in this chapter. Systems are marked by the coloured areas, with terrestrial biomes and mangroves based on biogeographic biomes (Dinerstein et al., 2017), and lakes and ocean biomes on IUCN functional biomes (Keith et al., 2022) (lakes are shown over other biomes for tundra only; fisheries are spread across the global ocean, but are marked only on key coastal seas for simplicity). Labels indicate which of the systems are in this report considered a tipping system (+++ high confidence, ++ medium confidence and + low confidence), which are not (- - - high confidence, - - medium confidence and - low confidence), and which are currently uncertain (▽).

Table 1.3.1: Summary of evidence for tipping dynamics, key drivers and biophysical impacts in each system considered in this chapter

System (and potential tipping point)	Key drivers	Key biophysical impacts (see S2 for societal impacts)	Key feedbacks	Evidence base	Abrupt / large rate change?	Critical threshold(s)?	Irreversible? (decadal / centennial)	Tipping system?
Forests								
Amazon rainforest (dieback)	**DC: atmospheric warming (↗)** **NC: deforestation / degradation (↗)** **DC: drying (↗)** **CA: fire frequency/intensity increase (↗)** DC: heatwaves (↗) CA: ENSO intensification (e.g. Amazon, SE Asia) (↗) CA: AMOC / SPG weakening / collapse (e.g. Amazon) (↗) CA: terrestrial greening (↘, declining)	• Biodiversity loss • Regional rainfall reduction (e.g. from Amazon dieback across Amazon Basin & Southern American Cone) • Carbon emissions (amplifying global warming) • Remote impacts on rainfall patterns all over the planet	Moisture recycling, fire, albedo	• Models • Observations (local scale)	++	1000-1250mm annual rainfall -400 to -450mm max. accumulated water deficit 7-8m dry season length ~20-40% deforestation ~3.5°C (2-6°C) global warming	++	**+++ (local)** **++ (partial dieback / regional)** **+ (full dieback / continental)**
Congo rainforest (dieback)					+	~1350mm mean annual rainfall; climate change increasing rainfall	+	**+ (local)**
SE Asia rainforest (dieback)					-	~1550mm mean annual rainfall	-	+? (local) - - (regional)

System (and potential tipping point)	Key drivers	Key biophysical impacts (see S2 for societal impacts)	Key feedbacks	Evidence base	Abrupt / large rate change?	Critical threshold(s)?	Irreversible? (decadal / centennial)	Tipping system?
Boreal forest (southern dieback)	**DC: drying (↗)** **CA: fire frequency/intensity increase (↗)** **DC: atmospheric warming (↗)** **CA: permafrost thaw (↗)** **CA: insect outbreaks (↗)** NC: deforestation / degradation (↗) DC: heatwaves (↗) CA: terrestrial greening (↘) CA: vegetation albedo (↗) CA: sea ice albedo decline (↗) DC: precipitation change (↘,↗)	• Biodiversity loss • Carbon emissions from dieback, carbon drawdown from expansion • Complex regional biogeophysical effects on warming - dieback = higher albedo (cooling) but less evaporative cooling (warming) & vice versa for expansion	Fire, albedo, moisture recycling	• Models • Observations • Experiments	++	~4°C (1.4-5°C)	+ [~100 yr]	**++ (partial / regional)** **+ (continental)**
Boreal forest (northern expansion)			Fire, albedo, moisture recycling	• Models • Observations • Experiments	+	~4°C (1.5-7.2°C)	+ [~100 yr]	**+ (partial / regional)**
Temperate forests (dieback)	**DC: atmospheric warming (↗)** **DC: droughts (↗)** **DC: heatwaves (↗)** CA: insect outbreaks (↗) CA: windthrow (↗) NC: deforestation & fragmentation (↗) CA: fire frequency increase (↗)	• Biodiversity loss • Carbon emissions • Regional warming in summer due to less evaporative cooling, less cloud cover • Less atmos. water supply • Less groundwater recharge	Moisture recycling, soil moisture -atmosphere, interacting disturbances, albedo	• Models • Observations • Experiments	++	Widespread thresholds uncertain	- [decades]	? (partial / regional)
Savannas, Grasslands & Drylands								

System (and potential tipping point)	Key drivers	Key biophysical impacts (see S2 for societal impacts)	Key feedbacks	Evidence base	Abrupt / large rate change?	Critical threshold(s)?	Irreversible? (decadal / centennial)	Tipping system?
Savanna & Grasslands (degradation)	**NC: fire suppression (↗)** **NC: overgrazing (↗)** DC: increased precipitation intensity (↗) CA: terrestrial greening (↗) NC: afforestation (↗) CA: ocean circulation shift (e.g. Sahe), ↗)	• Biodiversity loss • Groundwater depletion (with encroachment) • Nutrient cycle disruption • Reduced fires (with encroachment)	Fire, grazing	• Models • Observations (remote sensing & fieldwork)	+	Regionally variable mean annual rainfall; thresholds highly localised; Fire percolation threshold ~ 60% flammable cover	++	**++ (local to landscape)** ? (regional)

System (and potential tipping point)	Key drivers	Key biophysical impacts (see S2 for societal impacts)	Key feedbacks	Evidence base	Abrupt / large rate change?	Critical threshold(s)?	Irreversible? (decadal / centennial)	Tipping system?
Drylands (land degradation)	**DC: drying (↗)** **DC: atmospheric warming (↗)** **NC: land use intensification (e.g. livestock, agriculture, urbanisation)(↗)** DC: extreme events (heatwaves, floods) (↗) DC: increased rainfall variability (↗) CA: terrestrial greening (↘) CA: insect outbreaks (↗) CA: invasive species (↗)	• Biodiversity loss • Aridification / Desertification • Groundwater depletion (with encroachment) • Regional rainfall changes • Shift in species composition (e.g. shrub encroachment) • Vegetation recruitment	Soil fertility, / moisture / microbes, vegetation structure, veg-rainfall, fire, herbivory	• Models • Observations (current & historical) • Field experiment	++	Aridity index (0.54,0.7 and 0.8) (limited reliability of aridity measures; lack of temporal evidences for some thresholds)	+ (shorter timescales possible, e.g. via active restoration)	**++ (local to landscape)** **+ (regional)**
Freshwater								
Lakes (eutrophication-driven anoxia)	**NC: nutrient pollution (↗)** DC: atmospheric warming (↗) DC: precipitation changes (↗)	• Biodiversity loss • Water quality declineIncreased GHG emissions	Anoxia-driven P release, trophic cascades	• Observations • Models • Experiments	+++	20-30 mg P/l No clear warming/ rainfall thresholds	++ (decadal)	**+++ (localised, widespread)**
Lakes (DOM loading - 'browning')	**CA: terrestrial greening (↗)** **NC: afforestation (↗)** DC: atmospheric warming (↗)	• Biodiversity loss • Increased GHG emissions	Anoxia-driven P release	• Observations • Models	+	>10 mg DOC/l	++ (decadal)	**++ (localised, widespread in boreal)**
Lakes (appearance / disappearance)	**CA: permafrost thaw-related thermokarst formation / drainage (↗)** **CA: glacier lake formation / drainage (↗)**	• Biodiversity loss • Increased GHG emissions	(can be driven by thermokarst)	• Observations	+++	As for permafrost thaw	+++ (centennial)	- (localised, widespread on tundra)
Lakes (N to P limiting switch)	**NC: nutrient pollution (atmos. deposition) (↗)**	• Biodiversity loss	N/A	• Observations	++	Related to elemental ratio	++ (decadal)	- (localised, regions with high N-deposition)
Lakes (salinisation)	**DC: atmos. warming (↗)** **DC: drought (in arid regions) (↗)** **CA: water use intensification (↗)**	• Biodiversity loss • Reduced GHG emissions	Salt release from sediment	• Observations	+	Species-specific salinity threshold	++ (decadal)	- (localised, arid regions)

System (and potential tipping point)	Key drivers	Key biophysical impacts (see S2 for societal impacts)	Key feedbacks	Evidence base	Abrupt / large rate change?	Critical threshold(s)?	Irreversible? (decadal / centennial)	Tipping system?
Lakes (invasive species)	**CA: warming-driven range expansion (↗)** **NC: human-mediated introduction (↗)**	• Biodiversity loss	N/A	• Observations • Models	+	Cannot be defined	++ (decadal - centennial)	- (localised, widespread)
Coastal								
Warm-water coral reefs (die-off)	**DC: ocean warming (↗)** **DC: marine heatwaves (↗)** **CA: disease spread (↗)** CA: ocean acidification (↗) NC: pollution (nutrient / sediment) (↗) NC: disruption (ships, over-harvesting) (↗) CA: invasive species (↗) DC: storm intensity (↗) CA: sea level rise (↗)	• Biodiversity loss (ecosystem collapse, ~25% marine species have life stages dependent on coral reefs) • Loss of commercial & artisanal fisheries, and other sectors • Coastal protection loss	Thermal stress leading to symbiont expulsion,	• Observations • Models	+++	**Region and reef dependent:** • ~1.2°C (1.0-1.5°C) GW • Temporally variable heat stress (8-12 Degree Heating Weeks) • Long-term consequences of >350 ppm atmospheric CO^2 • Acidification threshold uncertain	++ (decadal)	**+++ (localised)** **+++ (regionally clustered)**

1.3.2.1 Tropical forests

Tropical forests cover around 1.95bn hectares (including degraded portions), and are key components of the Earth system (Pan et al., 2011) (Figure 1.3.2). They are home to a disproportionate amount of Earth's species (e.g. Slik et al., 2015; Pillay et al., 2021), store huge amounts of carbon (circa 471 ±93 GtC) in their soils and biomass, and, through evapotranspiration and their effect on cloud formation through production of aerosols and cloud condensation nuclei, have an overall cooling and moistening effect at regional scales (SPA, 2021; IPCC AR6 WG2 2021). They are also home to many Indigenous peoples and local communities, with a long history of human habitation and high biocultural diversity (Ellis et al.,2021).

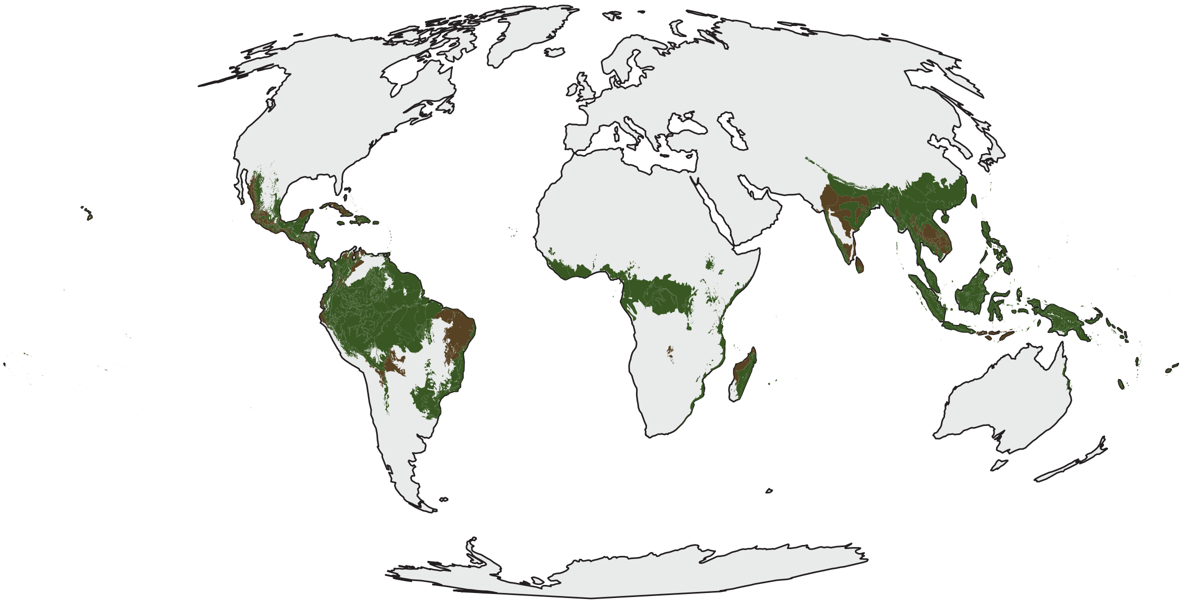

Figure 1.3.2: Top: map showing global extent of tropical forests, including tropical rainforests (dark green) and tropical dry forests (brown) (source: Dinerstein et al. (2017)) . Middle: photo of mature rainforest in Tapajós National Forest, Brazil (credit: Boris Sakschewski). Bottom: photo of arboreal Caatinga, a tropical dry forest formation in Eastern Brazil (credit: Kyle Dexter).

As well as experiencing deforestation and degradation due to land use change across the tropics (IPBES 2019), tropical forests in South America and Asia have been undergoing unprecedented climate-driven disturbances such as increasing dry season length and intensity, more intense and frequent rainfall and temperature extremes (Lapola et al., 2023; SPA, 2021). For instance, recent extreme droughts – mainly driven by climate variability modes such as the El Niño Southern Oscillation (ENSO) in 2014-2016 and the Atlantic dipole in 2005 and 2010 (e.g. Marengo et al., 2008; Marengo et al., 2011; Jimenez-Muñoz et al., 2016; see Chapter 1.4) – have caused extensive tree mortality, even up to 36 months after peak drought (e.g. Phillips et al., 2009; Phillips et al., 2010; Berenguer et al., 2021). Given the variability of forests across the tropics, their responses to global changes are likely to differ (Allen et al., 2017). Nonetheless, even subtle changes in their structure, composition and functioning could affect the global carbon and water cycles (e.g. Esquivel-Muelbert et al., 2019; Barros et al., 2019; Hirota et al., 2021).

Here we also consider deciduous and semi-deciduous forests (often referred to as dry forests) that coexist with evergreen forests in regions with around 1,000-2,000mm of annual rainfall, i.e. non-arid or dryland regions (Dexter et al., 2018). These dry forests may resemble (in terms of tree species composition) the dry forests in arid or dryland regions. However, because they exist in climates that can form continuous, high fuel-load flammable grass layers when canopies are opened (which is not the case in drylands), their dynamics are more comparable to neighbouring moist forests.

Evidence for tipping dynamics

Two positive/amplifying feedbacks are among the most plausible mechanisms that could lead to tipping dynamics in tropical forests, one at broader regional scales potentially causing large-scale forest collapse, and another at local scales potentially causing local forest collapse (Figure 1.3.3 and Box 1.3.2).

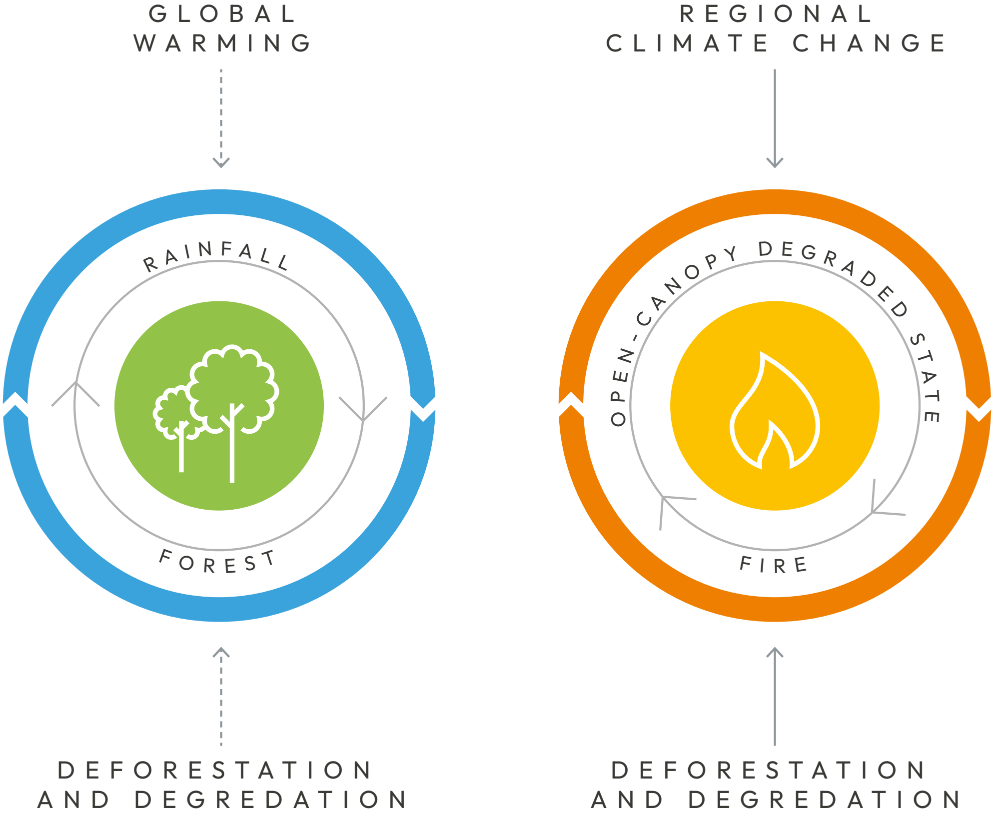

Figure 1.3.3: Diagram with positive/amplifying feedback loops that may cause large- and local-scale tipping events in tropical forests. (a) Regional climatic conditions are changing in response to global warming and also to deforestation, both of which contribute to weakening the forest-rainfall feedback. Reductions in rainfall cause water stress, increasing tree mortality and forest loss, further weakening the feedback, which could cause a large-scale forest collapse of the Amazon. (b) Interactions and feedbacks among the vegetation and fire can arrest the ecosystem in an open vegetation state, thus causing a local-scale forest collapse.

At regional scales, the forest-rainfall feedback is believed to be the dominant mechanism stabilising tropical forests by increasing annual rainfall levels and reducing its seasonal and interannual variability (Staal et al., 2020; Sternberg, 2001). However, under certain conditions it can instead amplify forest loss. Accumulated deforestation or forest loss reduces forest cover, which decreases evapotranspiration and moisture flow downwind, thus reducing regional rainfall (Smith et al., 2023). This in turn may increase tree mortality in downwind forest (Phillips et al., 2009; Berenguer et al., 2021), and beyond a threshold could lead to self-sustaining forest loss in drier areas of forest (Zemp et al., 2017; Staal et al., 2020) (Figure 1.3.3).

In the Amazon, on average, around 30 per cent of the water precipitating has been evaporated within the region beforehand at least once, but with large spatial differences: in the western Amazon, almost all precipitation has previously evaporated from the basin (Zemp et al., 2014; Staal et al., 2018). In the Congo basin, almost half of all precipitation originates from the Congo forest itself (Tuinenburg et al., 2020; Te Wierik et al., 2022). For Australian and Asian forests, evidence is still lacking, but this feedback likely has less effect on forest resilience due to less dependence on precipitation stemming from land evapotranspiration due to the major importance of monsoons (Staal et al., 2020).

At the local scale, evidence from across the tropics (Cochrane et al., 1999; Staver et al., 2011; van Nes et al., 2018) suggests that a fire-vegetation feedback can maintain the ecosystem in an open vegetation state: with less tree cover, fires spread more easily due to more flammable grassy fuels and because the air is drier in an open landscape without the local moistening effect of forest canopies. The resulting enhanced fire occurrence can in turn prevent the re-establishment of trees and maintain a more open vegetation state (Martinez-Cano et al., 2022; Drüke et al., 2023). This alternative open vegetation state could be either a natural savanna with native plant species (Flores and Holmgren, 2021; Beckett et al., 2022) or a degraded open-vegetation state when invasive plants are dominant (D'Antonio and Vitousek, 1992; Veldman and Putz, 2011; Malhi et al., 2014; Barlow et al., 2018) (Figure 1.3.3).

The effects of the fire-vegetation feedback are amplified by the regional forest-rainfall feedback (Staal et al., 2020). Moreover, forest loss may increase global warming by releasing carbon to the atmosphere, which further reduces regional moisture flows, causing more forest loss (Canadell et al., 2021). Also, climate change may change wind directions and residence times of moisture in a warmer atmosphere (Gimeno et al., 2021). Tropical forest loss also may change atmospheric circulation patterns (Portmann et al., 2022) and increase regional and global warming through reductions in cloud cover and evapotranspiration.

Among tropical forests, the Amazon forest has most evidence for potential tipping points. Analysis based on early warning signals (see Chapter 1.6) indicates that over 75 per cent of the Amazon has lost resilience since the early 2000s (Boulton et al., 2022). This decline is focused mostly closer to human disturbance, as well as in the drier south and east previously identified as 'bistable' (i.e. with two possible alternative states) due to the forest-rainfall feedback and thus is more vulnerable to tipping (Staal et al., 2020). While the Amazon has acted as a carbon sink due to CO_2 fertilisation, in mature forest this sink peaked and started declining in the 1990s (Hubau et al., 2020) and when including degraded forest (also predominantly in the drier south and east) the Amazon as a whole is now a carbon source (Gatti et al., 2021). Recent CMIP6 models indicate that localised shifts in peripheral parts of the Amazon forest system are more likely than a large-scale tipping event (IPCC AR6 WG1 Ch5, 2021; Parry et al., 2022). However, the latter cannot be ruled out (Hirota et al., 2021) because several compounding and possibly synergistic disturbances (e.g. combining an extreme hot drought with forest fires) may play a role in reducing forest resilience, with greater resilience loss closer to human activities (Boulton et al., 2022). Such synergies are generally not considered in Earth system models (Willcock et al., 2023).

A global warming threshold of ~3.5°C (2-6°C) has been estimated (Armstrong McKay et al., 2022), partly based on a few modelling studies that simulate some kind of nonlinear decrease in modelled properties of the Amazon forest, at least on small scales (Gerten et al., 2013; Drijfhout et al., 2015; Nobre et al., 2016; Boulton et al., 2017; Parry et al., 2022). However, most CMIP6 models do not include dynamic vegetation modules (Song et al., 2021; Canadell et al., 2021), which might make the forest artificially stable (Zemp et al., 2017). Models including deforestation, fire and dynamic vegetation have simulated widespread local-scale dieback (e.g. Cano et al., 2022; Parry et al., 2022), and also larger scale dieback in potential vegetation models (e.g. Salazar and Nobre, 2010).

Evidence pointing against a large-scale Amazon tipping point stems from palaeoclimate reconstructions suggesting that at least some parts of the Amazon forest have been resilient to past reductions in rainfall (Wang et al., 2017; Kukla et al., 2021) and temperatures as high as projected by climate models for the rest of the century (Steinthorsdottir et al., 2020). However, these were under more stable climate conditions (and before Pleistocene with different geographic effects on climate due to tectonics; (Brierley and Fedorov, 2016), with the current rate of warming far greater than during past climate changes (Zeebe et al., 2016; Osman et al., 2021). Geographically limited data means partial dieback elsewhere cannot be ruled out for drier intervals (Wang et al., 2017; Kukla et al., 2021), particularly in the drier south, where drying is currently leading to greater resilience loss (Boulton et al., 2022). Additionally, compounding disturbances are becoming increasingly widespread across the Amazon, even in remote central parts of the system, which is leading to resilience loss (Boulton et al., 2022) and could help trigger forest dieback at larger scales (Kukla et al., 2021; Wilcock et al., 2023).

Other tropical forests have evidence for local tipping points, but are less likely to cross them. The Congo has also been suggested as a possible tipping system (Staal et al., 2020) as it may also host a large area of bistable forest with some amplification by forest-rainfall feedback (Staver et al., 2011). However, because climate models indicate wetting across large parts of the Congo, it is not considered a tipping system in response to global warming (Armstrong McKay et al., 2022). The south-east Asian rainforests lack a strong regional forest-rainfall feedback and tend to have enough rainfall from ocean proximity for forests to remain stable, thus they are not considered a tipping system in relation to global warming (Armstrong McKay et al., 2022). Other tropical forests such as the Choco in Central America or Brazilian Atlantic Forests have not been assessed in detail.

Plants can reduce moisture transpiration in response to water limitation on very short timescales (hours to days), followed by water cycle feedbacks (weeks). Deforestation has a similarly fast effect on rainfall, as loss of trees can immediately reduce evapotranspiration. Large-scale forest dieback events in response to global warming can only be expected on the timescale of decades to centuries (Armstrong McKay et al., 2022). At a local scale, empirical evidence from the Amazon and from Africa has shown that forests can shift into savannas within a few decades after repeated fires (Flores and Holmgren, 2021; Beckett et al., 2022), and on larger scales tipping may occur faster (Cooper et al., 2020).

An Amazon tipping point would have global impacts from possibly large losses of carbon to the atmosphere. The best estimates suggest that a large-scale collapse of 40 per cent of the forest before the end of this century could lead to emissions of ~30 GtC and an additional global warming of ~0.1°C (Armstrong McKay et al., 2022). The Amazon dieback would also lead to substantial rainfall reductions across the Amazon basin and in to the Southern Cone of South America (Costa et al., 2021), and may also directly influence distant parts of the Earth system via 'teleconnections', for example to the Tibetan Plateau (Liu et al., 2023).

Assessment and knowledge gaps

The feedbacks that could contribute to tipping behaviour are relatively well understood in principle, yet there are large uncertainties surrounding the effects of climate and land use changes on these feedbacks. For instance, CO_2-fertilisation is expected to increase forest resilience locally, but it also increases water-use efficiency, reducing forest transpiration, and may thus weaken the forest-rainfall feedback and regional forest resilience (Brienen et al., 2020; Sampaio et al., 2021; Kooperman et al., 2018; Li et al., 2023). CO_2-fertilisation of tropical forests may also be overestimated in current Earth system models (Terrer et al., 2019; Hubau et al., 2020; Wang et al., 2020). Moreover, the actual thresholds and the extent to which tipping behaviour can be expected across heterogeneous landscapes and forest communities are much less certain (Levine et al., 2016; Longo et al., 2018; Sakschewski et al., 2021).

Considering only the Amazon as a rainforest tipping system, we have medium confidence in its potential for tipping of its bistable area (~40 per cent of the forest, predominantly in the drier south and east; Staal et al., 2020), with low confidence in the estimated tipping points and possibility of a large-scale collapse. The Congo may also be vulnerable to localised tipping (low confidence), but is unlikely to tip as a result of climate change, and localised tipping is possible but uncertain in south-east Asian rainforests.

Confidence in the tipping behaviour of tropical forests can be greatly improved through further development of models. Models can include dynamic vegetation modules and land use change to improve the representation of the forest-rainfall feedback, which would likely result in more drastic drying under high-deforestation scenarios (Parry et al., 2022). Incorporating fire dynamics in these modules would also likely result in a more bistable system (Drüke et al., 2023). In contrast, allowing for local vegetation adaptation (such as rooting depth) by including more plant types and traits in these modules would help better resolve the effect of landscape heterogeneity on tipping dynamics (Langan et al., 2017; Sakschewski et al., 2021), which may reduce the abruptness of the transition to an open degraded state (Levine et al., 2016). Efforts to increase ecological understanding of the feedback mechanisms and processes described here through observations (such as recent field studies on plant characteristics related to drought mortality throughout the Amazon basin (Tavares et al., 2023), or on the growth-survival tradeoff (Oliveira et al., 2021)) would help better understand forest dynamics and represent them in models.

1.3.2.2 Boreal forests and tundra

Boreal forests, also called 'Taiga', span around 1,135 million hectares, all located in the northern hemisphere (Pan et al., 2011) (Figure 1.3.4). They are vital for climate regulation, storing circa 272 (± 23) GtC, mostly below ground (Pan et al., 2011; Mayer et al., 2020). Management varies, but illegal logging constitutes a critical driver of boreal forest loss. Boreal forest growth is constrained by a short vegetation period, and their dynamics involve large-scale disturbances such as insect outbreaks and fire (with fire percolation dynamics important – see 1.3.2.4). While disturbance regimes differ in Eurasian and American forests, an overall increase in disturbances has been observed over past decades, fuelling worries about a wider loss of resilience.

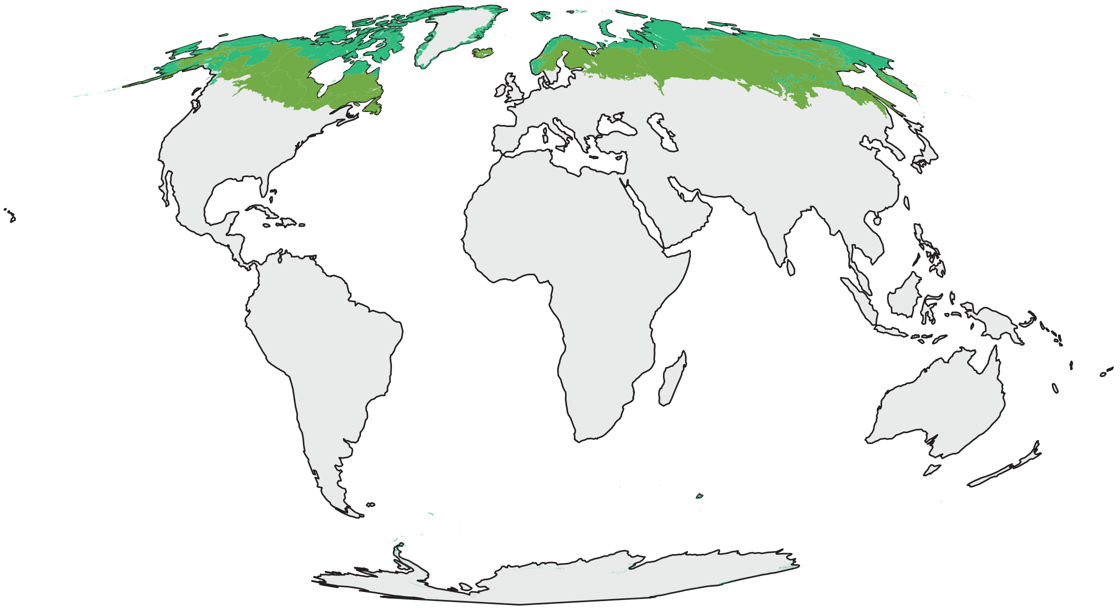

Figure 1.3.4: Top: map showing extent of boreal forests (light green) and tundra (blue-green) biomes (source: Dinerstein et al. (2017)). Bottom: photo of boreal forest and swamps, southern Norway (credit: Boris Sakschewski).

Evidence for tipping dynamics

Boreal forest dieback has already been identified as a potential tipping element in the climate system in Lenton et al., (2008) and further assessed in the IPCC AR5 WG2 Report in 2014 and in the WG1 and WG2 reports of AR6 in 2021 and 2022. The IPCC SR1.5 (Hoegh-Guldberg et al., 2018) and also the most recent assessment by (Armstrong McKay et al., 2022) differentiate between southern boreal forest and northern tundra tipping points. The southern boreal forest tipping point refers to a dieback of southern boreal forests that lead to a state-shift to an almost treeless state (to steppe/prairie), while the northern tundra tipping point refers to an expansion of tree cover into currently treeless tundra ecosystems.

There is little additional evidence for a boreal forest tipping point since the assessment of Armstrong McKay et al., (2022). Significant losses in tree cover driven by fires and logging were identified for the southern boreal forests of North America between 2000 and 2019 (Rotbarth et al., 2023). In contrast, interior boreal forests have become denser. There has been no clear sign of a northward expansion of the boreal forests of North America (Rotbarth et al., 2023). Similarly, Burrell et al. (2021) found that the forests of southern Siberia might have approached a tipping point as fire regimes have intensified, causing widespread regeneration failure. Moreover, Siberian larch foliage is sensitive to warming, with temperatures potentially exceeding a threshold by 2050 after which forest dieback can be expected (Rao et al., 2023).

A range of mechanisms contribute to the feedback processes associated with boreal tipping points (see Box 1.3.2 for more on forest feedbacks). For the southern boreal forest, the recent surge in forest disturbances, such as the extreme forest fires in Canada in summer 2023, is noteworthy because they constitute a substantial change in forest dynamics and resilience that, combined with failure to regenerate, could initiate regional tipping. In particular, the southern trailing edge of boreal forests has been identified as prone to compound and interacting disturbances, including droughts, windstorms, fires, large herbivores and insect outbreaks (Frehlich and Reich, 2010). For instance, increasing water stress reduces tree resistance against insects, and increases the size and severity of wildfires.

Southern boreal tipping points are driven by forest dieback from disturbances (Lenton et al., 2008). Empirical evidence from satellite data suggests that disturbances are responsible for switches between states rather than causing gradual change (Scheffer et al., 2012; Abis and Brovkin, 2017). Rotbarth et al. (2023) confirm that processes dominating the dieback of southern boreal forests and the northward expansion of forests into tundra diverge and that a northward expansion is not compensating for declines in the southern boreal forests of North America.

Climate change will further intensify disturbance regimes (Seidl et al., 2017), with fire regimes expected to increase significantly in boreal forests (Velasco Hererra et al., 2022). In Canada, fire frequency could increase up to 50 per cent by the 21st century under climate change (Flannigan et al., 2013). A doubling of fire frequency and increased wind activity during the 21st century will likely cause a significant decrease in coniferous forests, potentially replaced by early successional broadleaved tree species (Anoszko et al., 2022; Liu et al., 2022).

The increase in fire could potentially modify the forest microclimate, so that subsequent fires and droughts become more likely, causing a change in vegetation dynamics. For instance, Whitman et al. (2019) found that drought after fire exacerbates regeneration failure. Overall, drought-induced mortality will likely rise more in western than eastern North American regions (Peng et al., 2010). Moreover, insects, such as mountain pine beetles might expand into North American boreal forests, causing changes in ecosystem dynamics (Safranyik et al., 2010; Jarvis and Kulakowski, 2015). For instance, severe defoliation could impede birch forest recovery (Vindstad et al., 2018).

If these changes in disturbances cause widespread mortality while, at the same time, forests fail to regenerate, the forest might tip into an almost treeless state. Stevens-Rumann et al., (2022) suggest that a combination of changing climate patterns and disturbance regimes could primarily cause regeneration failure in coniferous forests. Bailey et al., (2021) highlighted the importance of temperature-moisture interactions for successful seedling establishment at the upper treeline in the Southern Rocky Mountains. However, over the past decade, no seedling establishment occurred at any site, suggesting that a threshold for regeneration may have been passed. Regeneration failure of boreal forests might occur with warming alone (+1.6°C to +3.1°C increase in one local warming experiment), but temperature thresholds are reduced if an increase in temperature is combined with reduced precipitation (Reich et al., 2022).

The sensitivity of coniferous tree recruits to climate change is overall higher than for broadleaved tree regeneration (Reich et al., 2022; Stevens-Rumann et al., 2022). In addition, natural disturbances might more likely cause state-shifts of coniferous than broadleaved-dominated boreal forest (Thom, 2023) as broadleaved tree species have an overall higher resprouting ability than conifers (Thom et al., 2021). Topographic complexity and peatlands may act as refugia from fire (Kuntzemann et al., 2023; Rogeau et al., 2018), thus reducing the likelihood of regeneration failure and state shifts. If widespread mortality becomes an increasing issue in northern forests, reduced microclimatic buffering of forests to increasing temperature might accelerate the thawing of permafrost in the boreal biome, causing additional releases of greenhouse gases – further interacting with the climate system [See Chapter 1.2.2.4 on Permafrost].

An increase in abundance of woody plants and advancing shrublines into the Arctic tundra is likely as climate changes (Mekkonen et al., 2021). This shrubification driven by warmer climate is also accompanied by northward treeline migration. A recent review of more than 400 treeline site locations suggested that at about two-thirds of treeline sites' forest cover had increased in elevational or latitudinal extent (Hansson et al., 2021). Main drivers of treeline migration are an increase in the rate of seedling success through warmer summers and increased winter temperatures. The change from tundra and peatlands to boreal forests can be nonlinear. Experimental work in boreal peat bogs reveals positive interactions between shrub cover and tree recruitment in which shrub cover favours tree seedlings and, in turn, higher tree basal area fosters shrub biomass, potentially triggering tipping towards high tree cover (Holmgren et al., 2015). As with southern dieback, interaction with permafrost thaw is also likely, but is complex and currently uncertain.

There are no clear thresholds for boreal forest dieback beyond the initial estimates already presented in Armstrong McKay et al., [2022]. With low confidence, they estimate a southern dieback tipping point at a global warming threshold of ~4°C (1.4-5°C) and a tipping timescale of ~100 (50-?) years, and a northern expansion into tundra tipping point at an estimated global warming threshold of ~4°C (1.5-7.2°C) and a tipping timescale of ~100 (40-?) years. Regeneration failure of southern boreal forests might occur with warming alone, while those thresholds are even lower if precipitation amounts also decrease (Reich et al., 2022).

Assessment and knowledge gaps

We assess with medium confidence that larger parts of boreal forests will approach a southern dieback tipping point and with low confidence that they will expand northwards as global temperatures increase by 3-4°C, if precipitation amounts and patterns remain similar. Yet, this threshold depends on multiple factors such as human and natural disturbances.

The capacity for adaptation and resilience is among the key uncertainties. Biodiversity, among other factors, might influence tipping dynamics as a diverse ecosystem may be more resistant to reaching tipping points, yet the effects of compositional and structural diversity require further investigation. Furthermore, although there is strong evidence and confidence in the increase of natural disturbances in boreal forests it remains uncertain whether they will truly lead to the transgression of a tipping point, pushing the southern range of boreal forests into an alternative, treeless state.

While in the southern boreal region the main mechanisms causing tipping points are relatively clear, for the northern tundra expansion tipping point the mechanisms sustaining large-scale abrupt state-shifts are not as evident. Disturbances in this region may be weaker and more localised, and the replacement of tundra by forest might occur more gradually. Yet, it is unclear if regeneration failure drives a self-sustaining feedback loop hindering recovery due to soil dryness or extreme conditions, causing a tipping point.

Further uncertainties linked to tipping points requiring further investigation include testing:

- interactions between climate, atmospheric forcing and disturbances;
- cascading and compounding disturbances;
- the existence of a 'fast-in, fast-out' behaviour of release and recovery in boreal forests;
- whether changes are self-reinforcing and perpetuating forest loss (or gain in the case of the northern tipping point);
- the extent of southern forest loss vs. northern forest expansion; and
- the role of human interventions, such as forest management on tipping dynamics.

Box 1.3.2: Forest feedbacks that could lead to tipping

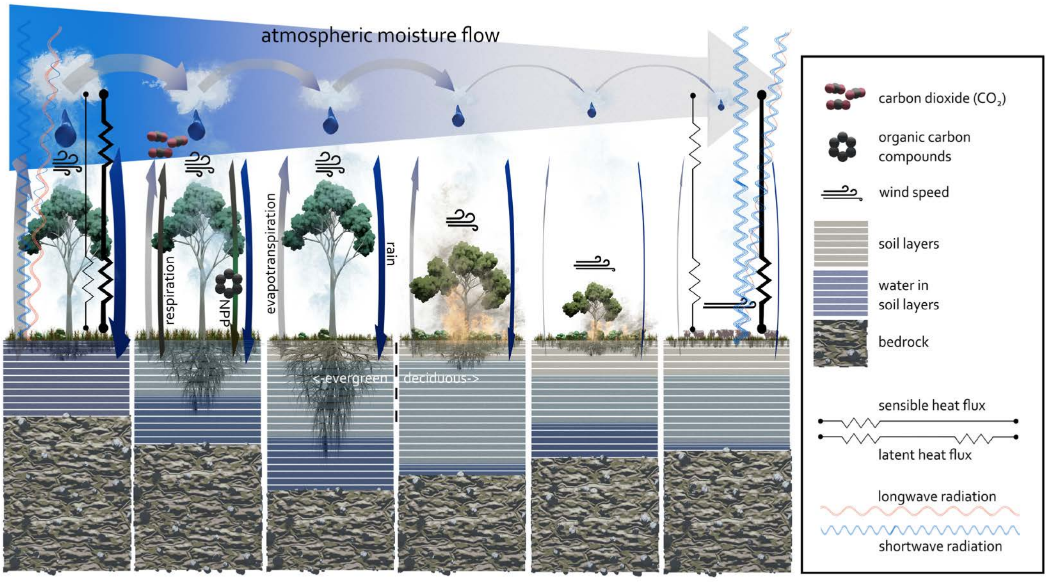

Figure 1.3.5: A conceptual regional transect from moist (left) to dry (right) localities depicting examples of local to regional feedbacks of forest cover with the land and atmosphere.

Less forest leads to ...

- less evapotranspiration (less productivity, less interception, less deep roots, etc.), hence reduced atmospheric moisture supply, and therefore reduced local and downwind precipitation, which leads to...
- less tree-produced volatile organic compounds (VOCs) serving as cloud condensation nuclei and therefore reduced local and downwind precipitation, which leads to...
- decreased roughness length of the landscape and hence increased wind speeds, leading to reduced residence time of moisture in the overall forest system, which leads to...
- decreased cloud formation due to less evapotranspiration, less VOCs, higher wind speeds leading to less reflectivity of sunlight, hence higher temperatures and therefore higher atmospheric water demand i.e. drought stress, which leads to...
- increased temperatures due to less evaporative cooling and decreased shading in canopy and ground proximity, hence higher atmospheric water demand i.e. drought stress, which leads to...
- more open canopy, drier understorey and less decomposition hence potentially larger pools of dead material to burn which all increasing fire probabilities, which leads to...
- higher windspeeds, less soil moisture and less soil retention capacity lead to higher erosion, which leads to...
- a surplus of atmospheric CO_2 by losing biomass carbon and losing a potential future carbon sink (a forest still capable of increasing biomass due to e.g. CO_2-fertilisation) and hence fueling global climate change, which leads to...

... less forest

1.3.2.3 Temperate forests

Temperate forests cover around 767 million hectares (16 per cent of the global forest area) and represent 34 per cent of global carbon sinks, storing around 119 GtC (Hansen et al., 2010; Pan et al., 2011) (Figure 1.3.6). In this report, we only consider temperate forests as defined in Figure 1.3.1. Mediterranean forests are covered under Drylands [see 1.3.2.5].

In most regions their spatial cover is highly fragmented following a long history of human land-use and forestry practices.

In fact there are only a few temperate forests which are considered 'intact' primary forest (Potapov et al., 2017; Sabatini et al., 2021) and the vast majority are managed by humans using vastly varying forest management techniques and intensities. Current managed temperate forests are often monocultures or mixtures of few tree species with relatively low biodiversity and structural diversity, optimised for high timber yields and certain wood features established under the assumption of stable climate and environmental conditions (instead of optimised for long-term forest resilience).

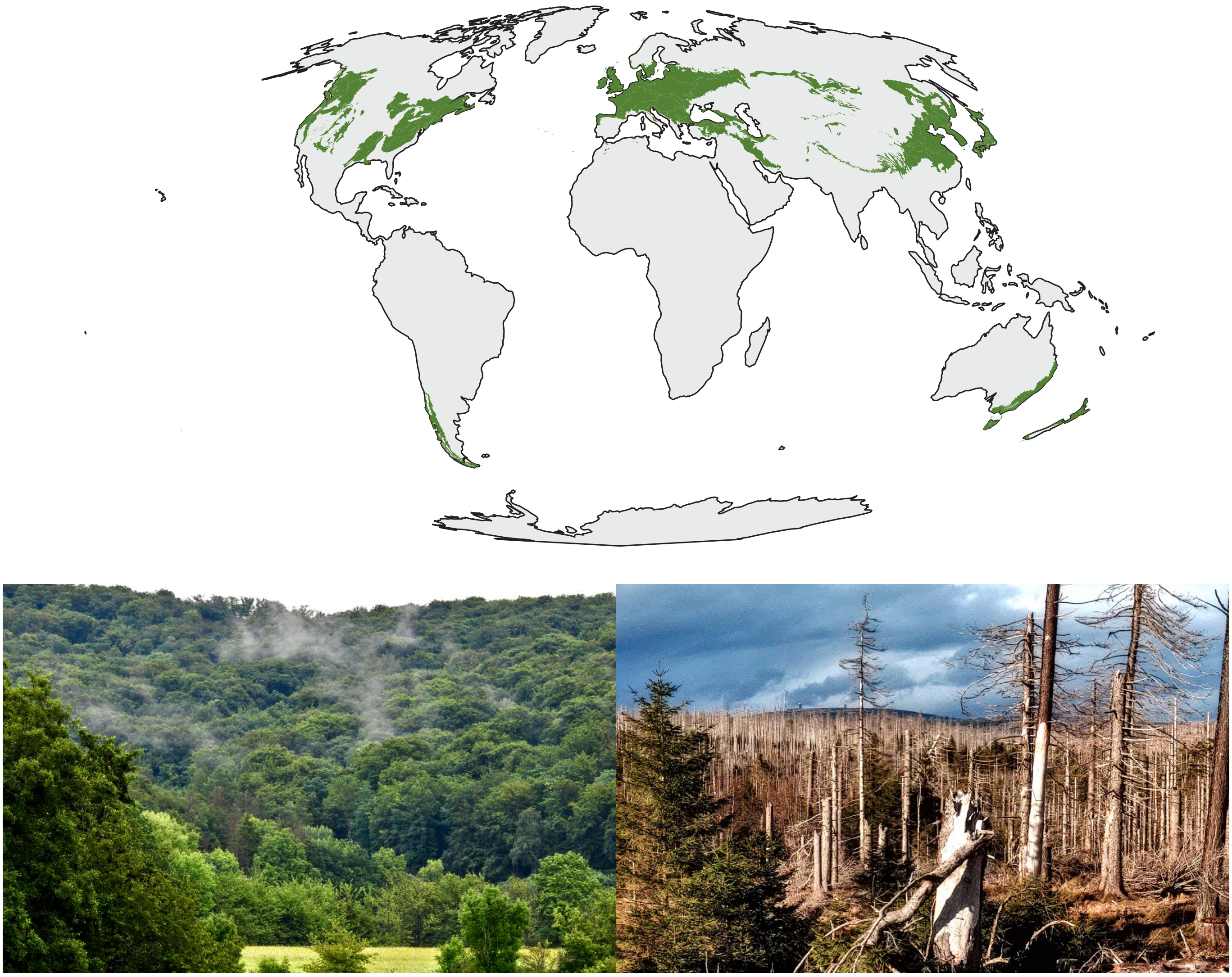

Figure 1.3.6: Top: map showing global extent of temperate forests biomes (green) (source: Dinerstein et al. (2017)). Left: photo of a mature temperate forest, Hainich National Park, Germany (credit: Boris Sakschewski,2022). Right: synchronous landscape-scale forest dieback (spruce monoculture) at Harz National Park, Germany (credit: Boris Sakschewski, 2023).

Evidence for tipping dynamics

In recent years temperate forests globally have suffered enormous damages and losses caused by extreme heat waves and droughts in combination with secondary effects like insect outbreaks and fires (Allen et al., 2010; Buras et al., 2019; Senf et al., 2020; Zhang et al., 2021; Carnicer et al., 2021; Benyon et al., 2023, Forzieri et al., 2022). As many temperate forests are effectively plantations for wood production in most parts of the world, those impacts often occurred in a similar synchronised manner on regional scales. Embedded in landscapes dominated by human land use (segregated by roads, crops, power lines, etc.), many temperate forests feature reduced connectivity and hence less exchange of species or genetic material, which reduces resilience (Sabatini et al., 2021).

More importantly, the extremely low diversity reduces the forest's ability to cope with stress through mechanisms such as portfolio insurance effects or complementarity (Billing et al., 2020). Portfolio insurance effects refer to the idea that having a diverse portfolio of species can help protect the forest against stressors by spreading the risk among different species. Complementarity refers to the idea that different species work together in a complementary way to improve the overall functioning of the ecosystem. However, when there is low diversity, these mechanisms may not be as effective and hence a potential tipping of temperate forests might also be more abrupt than in natural systems. Still, it must be noted that effective support from forest management (by regenerating an area through planting or supporting natural regeneration) can in principle also alleviate some of the pressures that natural systems face.

Besides the clear devastating signals of temperate forest damage and dieback, past assessments have had difficulties classifying temperate forests as tipping systems. In a review by (Thom, 2023) many temperate forest ecosystems were identified as resilient and/or resistant to increasing disturbance regimes and unlikely to shift towards alternative states in the very near future at large scale. However, drastic changes under intensifying future pressures such as climate change cannot be ruled out. In accordance with these findings, the recent assessment of (Armstrong McKay et al., 2022) has categorised temperate forests as an uncertain potential regional impact tipping system.

So far self-amplifying feedbacks in temperate forest dieback were described for more localised landscape-scale stressors like bark beetle attacks and fire in the Boreal forest section (see 1.3.2.2 and Box 1.3.2) (Hlásny et al., 2021; Fettig et al., 2022). On larger spatial scales it remains less clear whether temperate forests might feature self-amplifying feedbacks strong enough to induce tipping behaviour. However, just as in the tropical zone, the principles of cascading moisture recycling also apply to temperate forests. Any loss of forest cover reduces atmospheric moisture supply, hence reducing precipitation downwind and increasing sensible heat, which can amplify drying and warming in the affected areas (Pranindita et al., 2021). The average net cooling effect of temperate forests compared to grassland was found to be 1-2°C, with maxima of up to 5°C (Zhang et al., 2020). A recent study integrating data and modelling results reports continental-scale cooling effects of regrowing temperate forests on abandoned agricultural areas (Huang et al., 2020).

Related to this, cloud formation probability was found to be higher above forests in comparison to other land cover types in the temperate region (Teuling et al., 2017). Therefore, recent forest damages could have decreased cloud cover during recent droughts and heatwaves further intensifying these events. Furthermore, soil moisture-atmosphere feedbacks related to droughts and heatwaves were reported for the temperate zone (Seneviratne et al., 2010; Jaeger and Seneviratne, 2011) and could indicate that droughts might self-propagate in space and time (Schumacher et al., 2022). A recent study for the US west coast suggests cascading effects of soil moisture and biomass during a multi-year drought (Au et al., 2023). The recent large-scale forest damages and losses in the temperate zone (Senf et al., 2020; Lloret and Batllori 2021) could mark the beginning of self-amplifying and potentially self-sustaining feedbacks, but further work is required to confirm this.

The most important mediator between soil moisture and the atmosphere is vegetation, and forests especially stand out since they access water in great depths via their root systems (Sakschewski et al., 2021; Singh et al., 2020; Fan et al., 2017). Hence, larger-scale forest damage or loss means losing this mediator, further decreasing atmospheric moisture supply and downwind rainfall. This becomes particularly significant when, during droughts, precipitation becomes increasingly dependent on water evaporated from land or transpired by vegetation due to altered atmospheric patterns (Pranindita et al., 2021).

Additionally, local mechanisms or secondary effects could increase the likelihood of nonlinear responses, thereby increasing the probability of reaching tipping points. For instance in a more open forest or simply due to warmer and drier conditions at the forest floor, fire occurrences and intensities can easily increase. Moreover, the suppression of forest regeneration can occur due to the invasion of highly competitive light-demanding plant species, forming ecosystems which potentially transpire less moisture back to the atmosphere.

In combination with reduced resilience and resistance due to human interferences, abrupt large-scale damage and dieback of temperate forests is conceivable. Early warning signals in satellite-derived biomass data hint towards such a destabilisation (Forzieri et al., 2022). Yet, large-scale tipping behaviour in temperate forests is not proven. If at all, this will certainly be region-specific and recent forest damage will illuminate such potential feedbacks in the near future.

Assessment and knowledge gaps

It is uncertain if temperate forests have strong enough self-amplifying feedbacks like the Amazon rainforest and boreal forest to result in tipping, hence there is no evidence for larger-scale tipping and confidence is low. There is, however, a lot of evidence and medium confidence for abrupt changes with changing disturbances regimes.

Human forest management practices may have made temperate forests less resilient and therefore more susceptible to abrupt changes, but improved management can assist resilience and adaptation to climate change. Based on the impacts of current extreme events on temperate forests, it can be inferred that an increase in the intensity and/or frequency of such events could severely threaten existing forests in many areas, even without further climate change (Senf et al., 2020, Lloret and Batllori, 2021). The potential feedback to the water cycle requires further investigation. In particular, modelling studies should fully account for extreme events such as droughts, heatwaves and other important disturbances, their increasing frequencies and intensities as well as their potential impact on simulated vegetation and the resulting land-atmosphere feedbacks (Kolus et al., 2019).

1.3.2.4 Savannas and grasslands

Savannas and grasslands are characterised by the ecological dominance of grasses, sometimes with a substantial tree or shrub component (Figure 1.3.7). Savanna ecosystems are biodiverse and home to many people, but are being lost to a range of threats globally, especially because they are extensively targeted for agricultural conversion (Stevens et al., 2022; Strömberg and Staver, 2022). Even intact savannas are under threat, largely due to forest invasion or afforestation and woody encroachment, driven by grazing intensification and active fire suppression, and exacerbated by increasing atmospheric CO_2 (Stevens et al., 2017) and changing rainfall regimes (Kulmatiski and Beard, 2013). Active tree planting efforts further increase the threat to savannas from afforestation and woody encroachment.

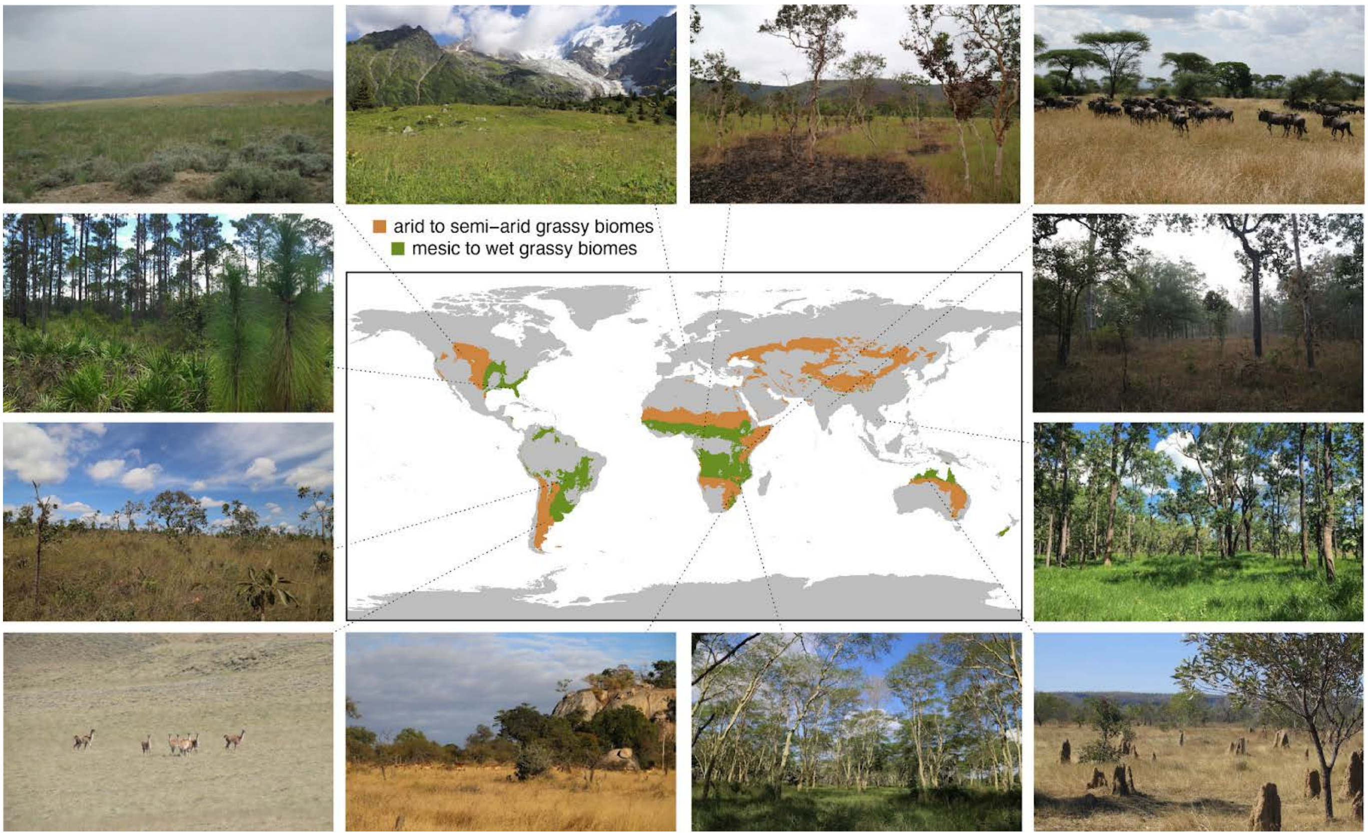

Figure 1.3.7: Global distribution of savannas and grasslands, showing semi-arid vs. mesic distributions (centre, from: Strömberg and Staver (2022), replotted from Dinerstein et al., (2017). Pictured, clockwise from top left, are native grassy ecosystems in 1) Montana near Dillon, USA; 2) Alps near Mont Blanc, France; 3) Pool Department, Republic of Congo; 4) Serengeti NP, Tanzania; 5) Pench NP, India; 6) Chhaeb Wildlife Sanctuary, Cambodia; 7) Kidman Springs Ranch, Australia; 8) Gorongosa NP, Mozambique; 9) Kruger NP, South Africa; 10) Santa Cruz Province near Lago Argentino, Argentina; 11) Instituto Brasileiro de Geografia e Estatística Reserve, Brasilia, Brasil; 12) Apalachicola National Forest, Florida, USA. Photo credits: Carla Staver, Caroline Strömberg, Naomi Schwartz.

Although the converse issue receives extensive attention (e.g. Amazon rainforest collapse), the issue of savanna vulnerability to tipping points is recognised (Staver et al., 2011b) but generally neglected in literature and assessments of tipping points in the Earth system (Armstrong McKay, 2022; Wang et al., 2023). Savanna vulnerability to desertification (corresponding to a self-sustaining loss of ecosystem productivity) is sometimes cited in tipping point syntheses, but the generality of this feedback has been questioned. For example, aridification observed in western Africa's Sahel during the 1970s and 80s has since reversed across much of the Sahel in response to a cyclic increase in rainfall (Nicholson et al., 1998; Prince et al., 2007).

Evidence for tipping dynamics

Savanna and forest are widely considered to be alternative stable ecosystem states in some climates (Staver et al., 2011a, 2011b; Hirota et al., 2011; Dantas et al., 2015; Aleman et al., 2020). In savannas and grasslands, an open tree canopy permits high grass productivity and thus the accumulation of grass fuel for frequent fires (Hennenberg et al., 2006; Lloyd et al., 2008). Fires in turn limit tree establishment (Higgins et al., 2000; Hoffmann et al., 2009), keeping the canopy open and creating a positive/amplifying feedback that potentially stabilises savannas in regions where forest is also a viable stable ecosystem state (Beckage and Ellingwood, 2008; Staver et al,. 2022a), although some apparent bistability may be the result of spatial climate variability (Good et al., 2015; Higgins et al., 2023).

The maintenance of savannas is thus dependent on fires across large parts of their range. This has meant that widespread fire suppression (active or passive via agricultural fragmentation or grazing intensification) has triggered woody encroachment and, in extreme cases, forest invasion (Stevens et al., 2017). These feedbacks between vegetation and fire frequency and intensity have also been implicated in accelerating the invasion of alien grasses that are more flammable and also tolerate higher fire intensities than native grasses (D'Antonio and Vitousek 1992; D'Angioli et al., 2022) (Figure 1.3.8). Fire-related feedback loops may not be as significant in drier savannas where herbivores or low water availability limit the accumulation of grass and thus fuel (Archibald and Hemson, 2016; Dexter et al., 2018), and further research is needed on the tipping dynamics of arid savannas and their potential alternate states (see Drylands 1.3.2.5).

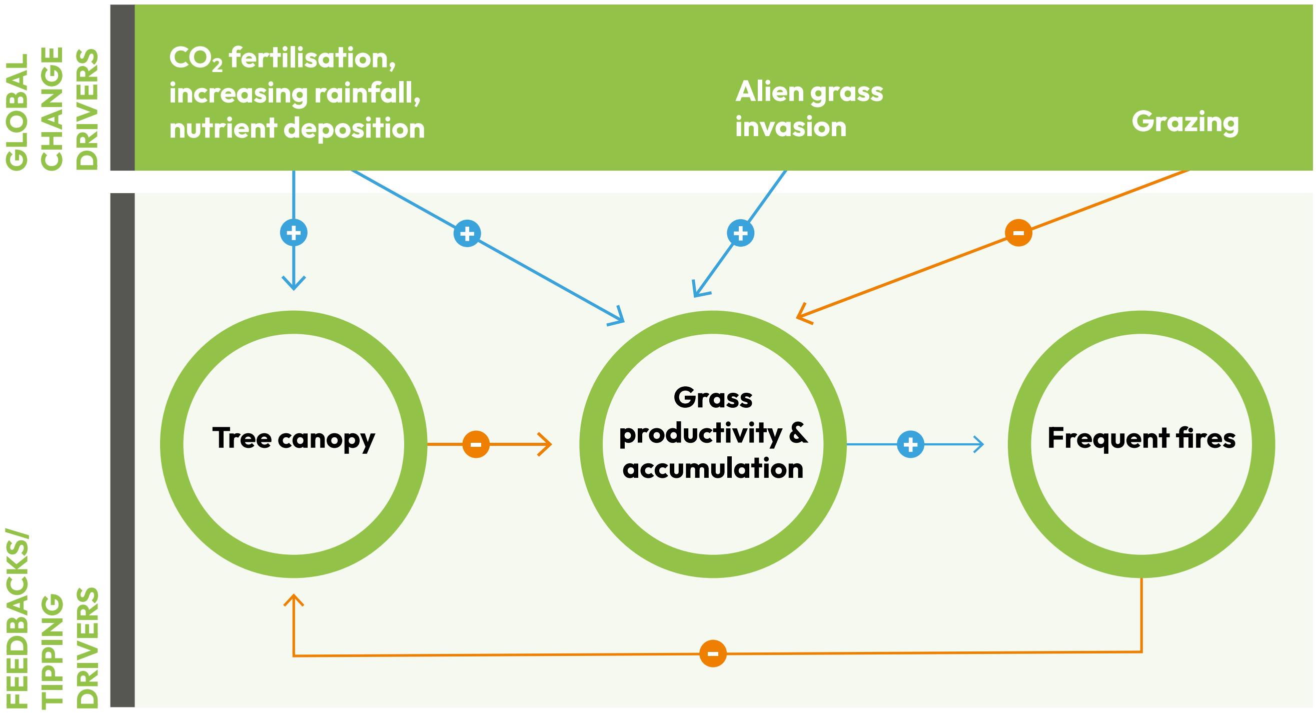

Figure 1.3.8: Key feedbacks that could lead to savanna tipping.

Several important thresholds are involved in this tipping point. First of all, fire spread is widely described as a percolation process (Loehle et al., 1996; Favier, 2004) – whereby a burning patch infects neighbouring or nearby flammable patches, thereby propagating fire in flammable landscapes. However, when not enough of the landscape is flammable (in this case, if trees shade grasses to prevent fuel accumulation), fires extinguish, with a clear threshold in fuel cover between 'connected' flammable vs. 'unconnected' non-flammable landscapes (Cardoso et al., 2022). In theory, this threshold can depend on the model used, but in practice, there appears to be a threshold in fuel cover of ~50-60 per cent, below which fire does not successfully spread (Archibald et al., 2009; Cardoso et al., 2022). Thus, fire suppression initiates woody encroachment or forest invasion, which can in turn decrease landscape flammability further, creating a cascade that results in the irreversible loss of open-canopy savannas.

The rate at which this happens – and ultimately the environmental space in which closed vs. open-canopy ecosystems are viable – depends also on environmental thresholds, but these are more widely disputed. A range of studies has defined the minimum required to sustain a closed forest canopy as ranging between 750 and 1,000mm mean annual rainfall (Sankaran et al., 2005; Staver et al., 2011b; Aleman et al., 2020), but more open but still fire-suppressing canopy can also form at much lower rainfall, for example in the Caatinga (Charles-Dominique et al., 2015; Dexter et al., 2018). The high-rainfall limit for savanna persistence is even less defined, as savannas can occur in areas with well over 1,600mm mean annual rainfall – for example, in the Llanos of Venezuela and Colombia (Huber et al., 2006) or the Beteke Plateau in the Republic of Congo (Nieto-Quintano et al., 2018).

Moreover, increasing atmospheric CO$_2$ is changing the relative photosynthetic efficiencies of 'C4' grasses vs. 'C3' trees (with C4 being the more efficient photosynthesis process) (Ehleringer and Björkman, 1977; Bond and Midgley, 2012) and is increasing plant water use efficiency across different plant types (Leakey et al., 2009; Norby and Zak, 2011). This has increased the rate of woody encroachment and forest invasion into savannas, suggesting that vulnerability of savannas to tipping points is accelerating and is not stationary with respect to climate (Higgins and Scheiter, 2012). For this reason, defining exactly how much global change might trigger savanna tipping points is not feasible (and indeed a single global tipping point may not exist).

Several lines of evidence provide support for the irreversibility of savanna-to-forest transitions. First, palaeoecological studies have suggested that reversible increases in rainfall can result in irreversible shifts from savanna to forest, consistent with hysteresis (i.e. where reversing the driver of change does not lead to recovery; see Glossary) (Karp et al., 2023). Second, and more directly, fire experiments have demonstrated that, while fire suppression causes savannas to transition to forest-like systems, introductions of fire into forests have much smaller effects (Gold et al., 2023), likely because closed forest canopies prevent fuels from accumulating to fuel intense savanna fires. This demonstrates that managed fire reintroductions are not sufficient to reverse forest encroachment (Gold et al., 2023).

Extreme fires can help reverse encroachment by forests when trees are fire sensitive (Silvério et al., 2013, Brando et al., 2014, Beckett et al., 2022) but extreme fires do not reverse woody encroachment (Strydom et al., 2023). In the case of savanna invasions by non-native grasses, irreversibility of transitions may be further exacerbated by resulting changes in nutrient cycling (Bustamante et al., 2012; D'Angioli et al., 2022). Together, these diverse lines of evidence suggest that savanna invasions, once initiated, may be rapid and irreversible.

The timescale of woody encroachment varies depending on environmental controls, but can happen in less than a decade, with accelerating vulnerability across savanna ecosystems due to rising CO_2 (Stevens et al., 2022) and widespread enthusiasm for climate mitigation via tree planting (Bastin et al., 2019, Fagan et al., 2022).

The climate impacts of woody encroachment and forest invasion are uncertain, however, due to substantial carbon in belowground pools in savannas (Zhou et al., 2022) and large uncertainty in how belowground carbon pools (root biomass and especially soil organic carbon) will respond to increasing woody cover (Veldman et al., 2019; Zhou et al., 2023). Hydrologically, there is evidence that an increasing tree fraction can increase rainfall interception and accelerate ecosystem water use, depleting groundwater recharge and streamflow, with implications for downstream water availability (Jackson et al., 2005, Honda and Durigan, 2016). Feedbacks with albedo (with woody vegetation being 'darker' than grass) have also been discussed, but little studied (Stevens et al., 2022).

Assessment and knowledge gaps

We have high confidence that Savannas are undergoing widespread degradation from woody encroachment, forest invasion, afforestation and alien grass invasion, high confidence that this is related to grazing intensification and active fire suppression and medium confidence that this is exacerbated by increasing CO_2 levels. These changes are increasingly difficult to reverse with the reapplication of fires (medium confidence), although sensitivity of invading vegetation to climate extremes is variable or unknown (Zeeman et al., 2014; Case et al., 2020). Compounded by agricultural conversion and tree planting, this is rapidly eroding endemic savanna and grassland biodiversity (high confidence) (Smit and Prins, 2015, Andersen and Steidl, 2019, Wieczorkowski and Lehmann, 2022).

Overall, savannas are likely to feature tipping dynamics at local to landscape scales (medium confidence), although large-scale synchrony may be observed if global change drivers trigger tipping points. However, Earth system feedbacks associated with savanna degradation are highly uncertain (low confidence), with particular knowledge gaps about carbon and hydrological cycle outcomes. Potential tipping points in savannas and grasslands associated with herbivory represent another major knowledge gap.

1.3.2.5 Drylands

Drylands are hyper-arid, arid, semi-arid and dry-sub-humid climate zones (Figure 1.3.9) where rainfall is less than 65 per cent of the 'potential evapotranspiration' (i.e. the amount of evaporation that would occur if enough water were available) (Middleton and Thomas, 1992). They occupy over 46 per cent of the Earth's surface and host 38 per cent of the world's human population (more than 2 billion people) (Cherlet et al., 2018). Vegetation types include deserts, grasslands, shrublands, woodlands, savannas, Mediterranean forests and tropical dry forests (see 1.3.2.1 and 1.3.2.4 for tropical dry forests and savannas). Due to their extent and the chronic water deficit, these areas are of particular concern in the face of global changes, and so we assess them separately here, despite some overlap with the tropical forest and savanna and grassland biomes above.

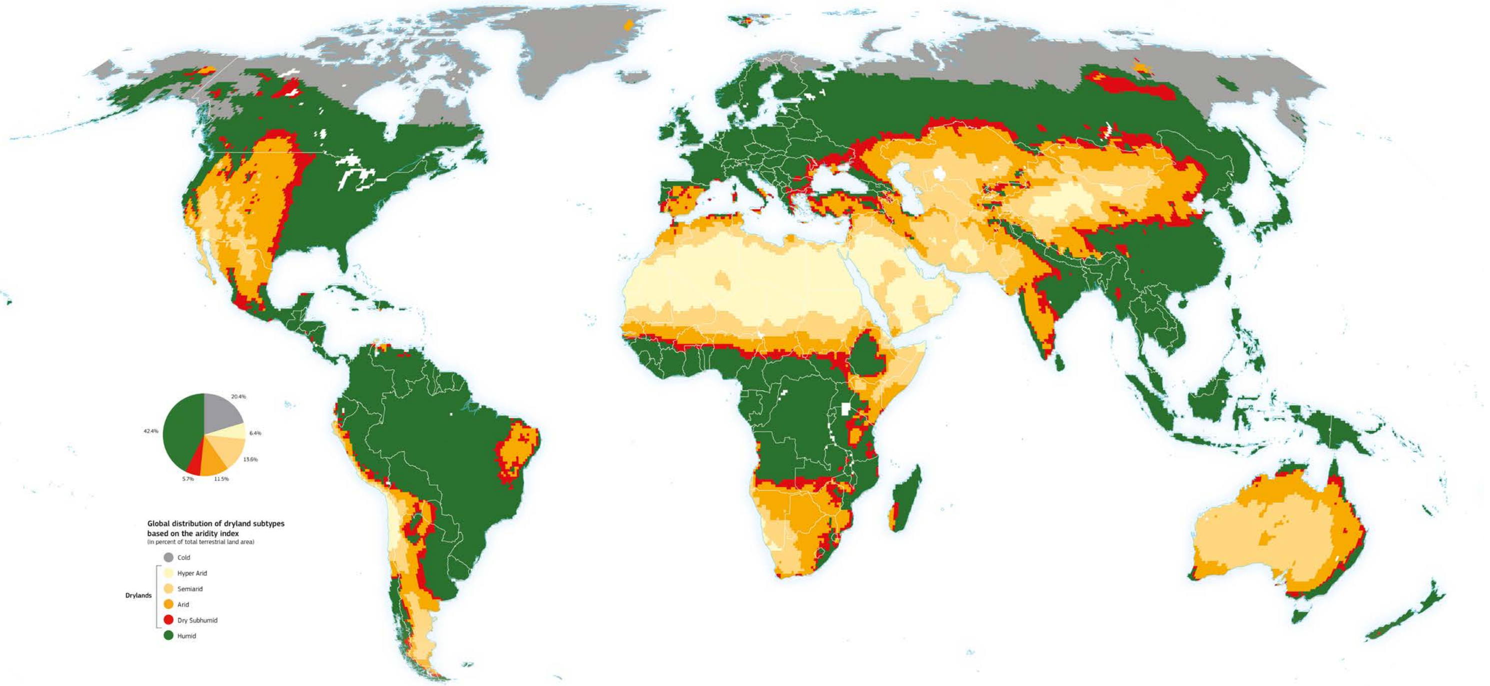

Figure 1.3.9: Global distribution of dryland subtypes based on the aridity index. Source: WAD3-JRC (Cherlet et al., 2018).

Recent estimates suggest that one-fifth of drylands are degraded as a result of climatic variations and human activities (Burrel et al., 2020; about 9 per cent in IPCC SRCCL, 2019). Major pressures on drylands (Cherlet et al., 2018) include:

- Climate change – for example, changes in precipitation, temperature, seasonal and interannual variability and frequency of extreme events. Projections indicate that some drylands might become more humid, whereas others may become drier (Huang et al., 2016; Pravalie et al., 2019). These expectations are uncertain though (Lian et al., 2021).
- Land use intensification – for example, grazing (the main use of drylands, at 62 per cent) (Cherlet et al., 2018), water extraction, deforestation, agriculture and urbanisation.
- Perturbations – for example, fires, insect outbreaks and biotic invasions.

The dynamics of drylands depend strongly on the interaction between these pressures, such as climate change and local perturbations (Rilig et al., 2023).

Evidence for tipping dynamics

Different lines of evidence point toward the existence of tipping dynamics in drylands, including past and current ecosystem transitions, bistability of dryland states at the global scale, thresholds along environmental gradients, and feedback mechanisms maintaining persistent dryland states.

Abrupt transitions have historically occurred in several dryland systems. Palaeo evidence reveals abrupt shifts into and out of African Humid Periods (Pausata et al., 2020), including a notable greening of the Sahara during the early to mid-Holocene, followed by its abrupt desertification around 5,500 years ago (Shanahan et al., 2015; Claussen et al., 2017; Hopcroft and Valdes, 2021, Claussen et al., 1999). Positive/amplifying feedback mechanisms between vegetation and the monsoon in North Africa are thought to be important (Charney et al., 1975). Climate change projections suggest 'Sahel Greening' might partially occur again in the future (Erfanian et al., 2016; IPCC SR1.5, 2018; Dosio et al., 2021). In dune systems, stratigraphic records covering 12,000 years have found coexistence of a vegetated, stabilised state and a bare active state in dune systems in northern China, with occasional sharp shifts in time between those contrasting states and hysteresis (Xu et al., 2020).

Shrub encroachment may also reflect tipping dynamics. Long-term data from Jornada Experimental Range (northern Chihuahuan Desert, New Mexico, USA) showed abrupt transitions from grasslands to shrublands triggered by a combination of climatic and human (i.e. overgrazing) factors during the last 150 years (Beltelmeyer et al., 2011; D'Odorico et al., 2012). Transitions from Mediterranean forests to shrublands have been reported under a combination of dry conditions, wildfires (Baudena et al., 2020; Acacio et al., 2009, Mayor et al., 2016) and herbivory (van der Wouw et al., 2011).

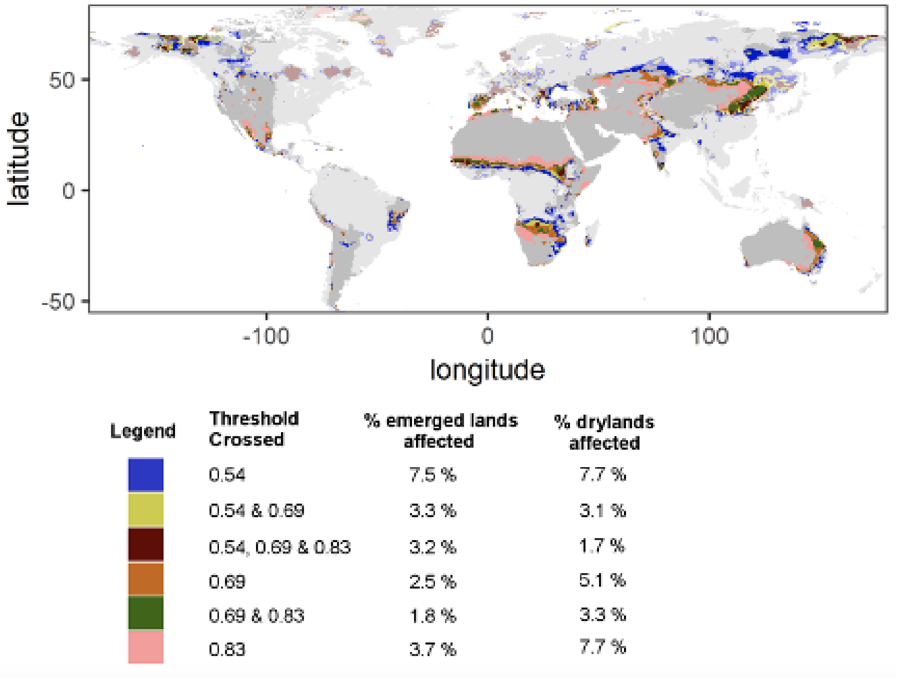

Figure 1.3.10: Map of drylands vulnerability to predicted changes in aridity for 2100 based on the IPCC RCP8.5 scenario (i.e. under the assumption of sustained increase in CO$_2$ emissions). Abrupt decays in plant productivity, soil fertility and plant cover were identified beyond aridity threshold values of respectively 0.54, 0.7, and 0.8 (Berdugo et al., 2020). The map displays areas that are expected to cross one (or several) of those thresholds in aridity level. Light-grey areas are areas that are not drylands today. Figure from (Berdugo et al., 2020). Satellite observations indicate that 5 per cent of drylands have experienced an abrupt loss of vegetation cover over the last 20 years, while 18 per cent underwent an abrupt increase in vegetation (Berdugo et al., 2022).

Evidence suggests that, in drylands, sequential abrupt shifts in plant productivity, soil fertility and plant cover occur at increasing aridity thresholds, respectively corresponding to aridity values of 0.54, 0.7, 0.8 (Berdugo et al., 2020) (Figure 1.3.10). A higher dependence of vegetation on water has been reported at aridity values of around 0.6 (Nemani et al., 2003), producing a decline in productivity with increasing aridity (Berdugo et al., 2020) and an increase in tree mortality events with hotter droughts (Hammond et al., 2022).

At aridity levels around 0.7, abrupt declines in vegetation are related with losses of soil fertility (Delgado-Baquerizo et al., 2013; Berdugo et al., 2020), changes in vegetation spatial structure, (Kéfi et al., 2007, 2011; Berdugo et al., 2017; Berdugo et al., 2019) which influences soil hydrological connectivity and resource loss at the landscape scale (Mayor et al., 2013; Rodriguez et al., 2018, Mayor et al., 2019), increases in the dominance of shrublands (Berdugo et al., 2020), and rapid shifts in the composition of soil microbial communities and soil functionality (Maestre, 2015; Lu, 2019; Delgado-Baquerizo, 2020; Zhang et al., 2023).

At aridity thresholds of 0.8, abrupt decays in plant productivity and vegetation cover occur (Berdugo et al., 2020) and can lead to a nonlinear increase in soil erosion (Mora and Lázaro, 2013; Elwell and Stocking, 1976; Francis and Thornes, 1990; Mayor et al., 2013).

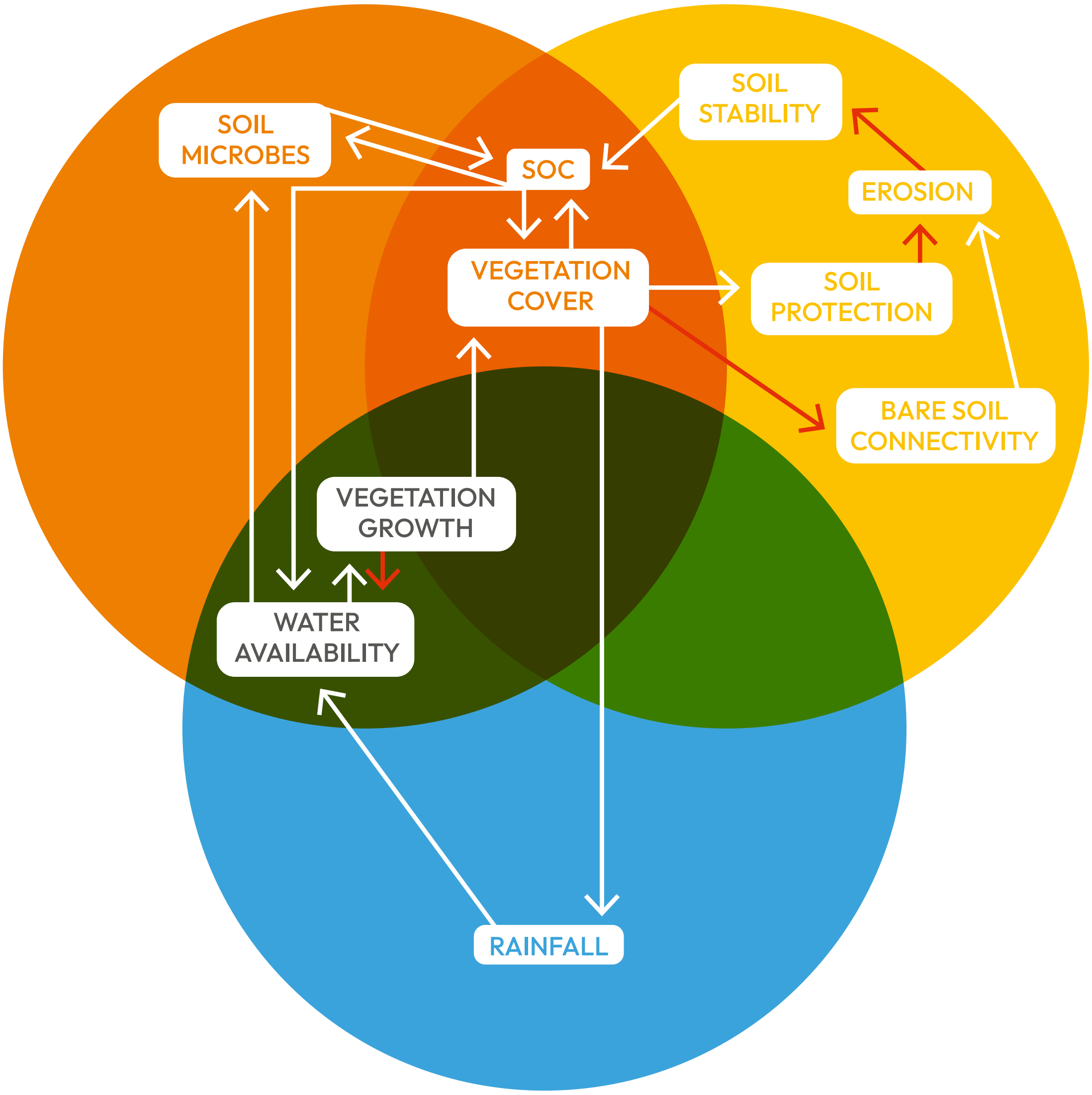

Figure 1.3.11: Schematic showing key feedbacks that could lead to dryland tipping. Coloured disks represent some of the main feedbacks described in the text (vegetation-rainfall in blue, biogeochemical feedback in red and ecohydrological feedback in blue). White arrows represent positive effects (an increase in the variable at the source of the arrow leads to an increase of the variable at the end of the arrow) and red negative effects. SOC stands for Soil Organic Carbon. See (Mayor et al., 2019) for a more detailed version of the ecohydrological feedback.

A number of feedback mechanisms are known to occur in drylands, operating across ecosystem elements and at different spatio-temporal scales. Theoretically, such feedbacks can lead to bistability and abrupt transitions between stable states in drylands (Holling, 1973; Noy-meir, 1975; May, 1977; Scheffer et al., 2001, Walker et al., 2004), although such alternative states do not necessarily exist in all regions (Ma et al., 2023). Several key feedbacks can be identified (Figure 1.3.11):

- **Soil microbial communities (biogeochemical feedback; small scale):** Microbial biomass and diversity in drylands are intricately linked to variations in water availability and organic matter (which change along the global aridity gradient; Zhang et al,.2023). Soil microbes, such as bacteria, fungal decomposers and mycorrhizal fungi, are fundamental for the breakdown of complex litter and organic matter. By decomposing organic matter, microbes are critical in the build-up of soil carbon stocks, which is essential in the maintenance of moisture in dry soils. Soil moisture, in turn, is needed for organic matter decomposition.
- **Plant-plant interactions:** In drylands, plants are known to facilitate the recruitment and growth of other plants, leading to the formation of vegetation patterns. The positive interactions between plants, i.e. facilitative effects, involve effects on microclimate, soil conditions and herbivores impacts:
  - **Plant-soil feedback (small to medium scale):** Plants enhance local soil conditions through several means, such as nutrient and water retention ('islands of fertility'), microclimate influence and erosion prevention (Aguiar and Sala 1999; Schlesinger et al., 1990; Rietkerk et al., 2000; D'Odorico et al., 2007). These processes boost vegetation growth and contribute to the formation of spatial patterns.
  - **Ecohydrological feedbacks (medium scale):** Plants aggregate and form spatial patterns of vegetation patches interspersed in a matrix of bare soil (Aguiar and Sala, 1999). The spatial connectivity of the bare soil (runoff-source areas) affects the redistribution of water, nutrients and sediments at the patch and landscape scale, which in turn shapes vegetation cover and spatial pattern (Mayor et al., 2013, 2019). These local (patch) and global (landscape) connectivity-mediated feedbacks affect the productivity and resilience of the ecosystem (Mayor et al., 2019).
  - **Vegetation-herbivore feedback (medium scale):** Herbivores graze/browse on palatable plants, which stimulates regrowth (McNaughton 1983); they then keep eating at the same places because the resprouts are soft and more easily digestible. This in turn allows the recruitment of unpalatable plants in areas without herbivores. An excess of grazing on palatable plants can prevent regrowth and lead to vegetation transitions from diverse, palatable to unpalatable dominated plant communities (Cingolani et al., 2005).
- **Vegetation-fire feedback (medium to large scale):** Fire can facilitate a transition from forest to shrublands. Shrublands recover faster and burn easier, generating a positive/amplifying feedback (e.g. dry Mediterranean regions in Portugal (Acacio et al., 2009), Spain (Baudena et al., 2020)). Replacement of native Mediterranean forests by pine forest plantations or invasion by exotic non-woody plants can contribute to this feedback (e.g. central Chile) (Pauchard et al., 2008; Gomez-Gonzalez et al., 2018) (see also: Tropical [1.3.2.1] and Boreal [1.3.2.2] forests).
- **Vegetation-rainfall positive/amplifying feedbacks (large scale):** Vegetation is largely controlled by local climate, but modelling studies suggest that it can also influence regional precipitation by modifying the atmospheric energy and water budget (Charney, 1975; Dekker et al., 2007). This large-scale albedo-precipitation and evapo-transpiration–precipitation feedback could have significant implications for ecosystem resilience. (see also: Tropical [1.3.2.1] and Boreal [1.3.2.2] forests).

Global dryland assessments suggest two different ecosystem states can exist at intermediate aridity levels ('bistability'). Drylands with aridity levels between between 0.75 and 0.8 (i.e. in the transition zone between semi-arid and arid drylands) may be in one of two different states, with higher and lower vegetation cover, with large contrasts in soil fertility, nutrient capture and nutrient cycling (Berdugo et al., 2017). Observing different ecosystem states across an area with similar conditions does not in itself prove those ecosystems are bistable. However, the global tendency for these two states to emerge, combined with our understanding of feedbacks in these ecosystems and observations of threshold responses, suggests that these could represent alternative stable states in these ecosystems.

Hysteresis, where reversing the driver of change does not lead to recovery (see Glossary), can also be evidence for alternative stable states and tipping dynamics in dryland ecosystems. In Spain (NE, Ebro Valley), past overgrazing was found to interact with droughts to explain the lack of secondary succession or even decreasing normalised difference vegetation index (NDVI, a remote sensing index for vegetation cover) trends (Vicente-Serrano, 2012). Some long-term field studies provide evidence for hysteresis in drylands. For example, in the northern Chihuahuan Desert (US), grasslands shifted into shrublands dominated by Creosote Bush (***Larrea tridentata***) during a prolonged drought combined with overgrazing, but the recovery of grass productivity did not occur in subsequent wet years (Bestelmeyer et al., 2011). Results also suggest the possibility of crossing critical thresholds for irreversible degradation (i.e. 20 per cent plant cover in Gao et al., 2011).

Long legacy effects are consistent with the existence of hysteresis in drylands. For example, palaeoclimatic legacies, e.g., from the Last Glacial Maximum, influence soil biodiversity (Delgado-Baquerizo et al., 2017), function (Ye et al., 2019) and forest distribution (Guirado et al., 2022). For example, drylands with a wetter past now have greater levels of function and forest coverage than what would be expected for current climatic conditions (Ye et al., 2019).

The reversibility of ecological transitions in drylands is challenging because plant growth rate is strongly limited by water scarcity and local disturbances. However, it is noteworthy that fast vegetation recovery during rainy periods has been observed at local and regional scales (Holmgren et al,.2006a, 2013). Studies have also found recovery of drylands to strong grazing pressure even at low cover levels in case of favourable weather conditions (Bestelmeyer et al., 2013). Coupling passive and active restoration of drylands to favourable climate swings can open windows of opportunity for dryland recovery (Holmgren and Scheffer 2001, Holmgren et al., 2006b, Sitters et al., 2012).

Timescale for transitions are about weeks to months for tree heat and grazing, months to decades for shrub encroachment Bestelmeyer et al., 2011; Tabares et al., 2019 and abrupt vegetation loss due to droughts Berdugo et al., 2022), and a few decades for the desertification of the Sahara (Shanahan et al., 2015; Claussen et al., 2017; Hopcroft and Valdes 2021; Claussen et al., 1999).

Assessment and knowledge gaps

Dynamical evidence of tipping points in drylands is challenging to find due to the slow dynamics of these ecosystems. Altogether, the knowledge of past transitions shows that relatively rapid changes have occurred in drylands, in particular in terms of vegetation cover, species composition and soil communities, leading to important changes for biodiversity and ecosystem functioning. Further, evidence from positive/amplifying feedbacks between different components of ecosystems, thresholds values in stressors (aridity, fire frequency, grazing) and hysteresis (lack of recovery) suggests the likelihood of future recurrence. We assess that dryland ecosystems can feature local to landscape-scale tipping points towards land degradation (medium confidence) with climate and land use change.

Core ecological questions remain, mainly on the mechanisms by which abruptness appears in drylands. We need long-term dynamical records. This is in particular true for soils; which is very relevant given that several thresholds involve soil transformations (particularly soil fertility losses). This lack of evidence in soils is even more difficult to address given that soils are themselves a slow component in an already slow ecosystem type and, unlike vegetation, can not be assessed with remote sensing. Quantification of the thresholds for herbivory pressure, fire frequency, and logging along aridity gradients is also necessary. Crucially, we need to improve our description and incorporation of social-ecological feedbacks in drylands (Reynolds et al., 2007). Indeed, dryland ecosystem transitions are associated with important social pressures and livelihood dependency, especially in developing countries, making social-ecological feedbacks critical to understand (see Section 2; Walker et al., 2004; Reynolds et al., 2007).

Several biotic mechanisms (e.g. local negative plant-patch feedbacks – Mayor et al., 2019) that confer resilience to dryland ecosystems are still not sufficiently explored, such as plant plasticity or adaptability to drought. Some mechanisms might be able to counteract abrupt changes; for example CO_2 fertilisation may confer higher water use efficiency to plants, thus opposing stress caused by lack of water (Zhu et al., 2016) and possibly counteracting aridification (Peñuelas et al., 2017; Zhang et al., 2022). Also, we can refine our understanding of windows of opportunity for restoration in drylands (e.g. taking advantage of temporarily favourable climatic conditions; Holmgren and Scheffer 2001; Holmgren et al., 2006b; Sitters et al., 2012; Walker and Salt 2012).

1.3.2.6 Freshwater ecosystems

The scientific content of this chapter is closely based on the following scientific manuscript: Hessen et al., (in review) Lake ecosystem tipping points and climate feedbacks, Earth System Dynamics Discussion.

Freshwater bodies such as lakes are common across most biomes, forming unique and sometimes isolated ecosystems (Figure 1.3.12). In natural sciences, the hysteretic behaviour of lakes (Scheffer et al., 2007) has informed the concept of tipping points at the ecosystem level, leading to the development of the alternative stable states theory in shallow lakes (Scheffer et al., 1993; Carpenter et al., 1999; Carpenter 2005). They represent archetypal case studies for how tipping points relate to theories of ecological stability and resilience that can underpin preventative management approaches (Andersen et al., 2009; Spears et al., 2017). Despite this, significant uncertainty remains on the geographical extent of tipping points in lakes and the wider relevance for the Earth's climate system.

Lakes are also good examples of social-ecological systems, with their ecological dynamics closely intertwined with the socio-economic dynamics of surrounding populations who often depend on them for key ecosystem services and adaptively respond to changes in lake condition (Martin et al., 2020). Given the global vulnerability of freshwaters and the pervasive nature of major pressures acting upon them (e.g. nutrient pollution and climate change), tipping points in these systems could have significant societal impacts, including on human and environmental health, food production and climate regulation. The capacity to detect discontinuous ecosystem responses to pressure changes in natural systems has been challenged (e.g. Hillebrand et al., 2020). Nevertheless, there are several studies that have reported real tipping points, i.e. shifts from one stable state to another in small shallow lakes (the most common lake type globally, Messager et al., 2016).

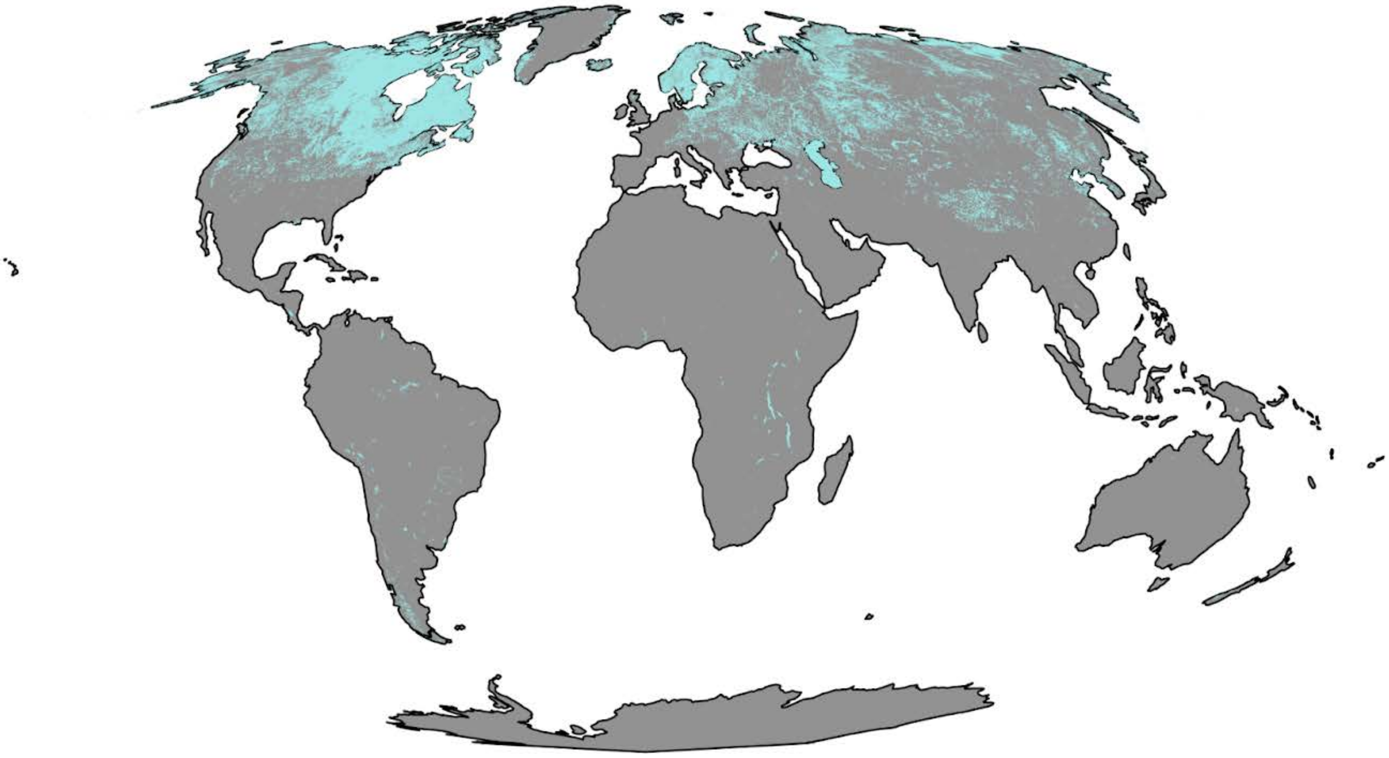

Figure 1.3.12: Top: map showing global distribution of lakes (light blue) (source: (Keith et al., 2022). Middle left: eutrophic urban lake receiving high organic matter loading leading to elevated CH_4 emissions, Bellandur Lake, Bengaluru City, India (photo: Laurence Carvalho). Middle right: boreal, brown water lake with deepwater anoxia and high emissions of CO_2 and CH_4 (photo: D.O. Hessen). Bottom left: Arctic pond at Svalbard, recently formed by permafrost thaw below Zeppelin mountain (photo: D.O. Hessen). Bottom right: thermokarst lakes in Yukon Flats, Alaska (photo: Sebastian Westermann).

Empirical analyses, process modelling and experimental studies are advanced for shallow lakes providing a good understanding of ecosystem behaviours around tipping points, typically starting with positive/amplifying feedback loops, then entering a runaway phase before finally the tipping point brings the system into a different stable state (Nes et al., 2016). For example, the well documented increase of phosphorus (P) loading across European lakes in the last century (e.g. from agricultural and waste water pollution) has uncovered critical loading thresholds beyond which lakes can shift rapidly from a clear water, macrophyte rich state to a turbid, phytoplankton rich state (Scheffer et al., 2001; Jeppesen et al., 2005; Tátrai et al., 2009), and vice versa when nutrient loading decreases.

Adding to such well-described and mechanistically well-understood changes, there is a wide range of local or single lake shifts that may be categorised as tipping points. The question remains as to whether tipping points are merely isolated phenomena in single lakes, or specific types of lakes, or whether they manifest, or will in the future, across geographically distinct populations of lakes experiencing similar environmental change, with the potential for regional or global extent (Figures 1.3.12 and 1.3.13).

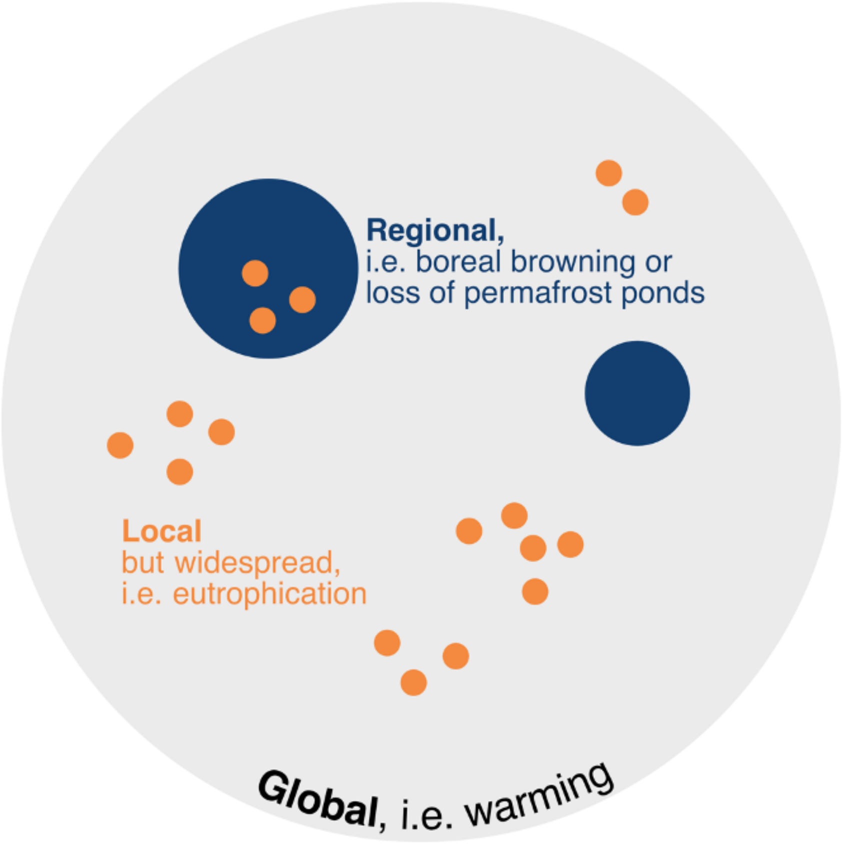

Fig. 1.3.13: Impacts at levels that may qualify for tipping points at relevant scales. Regional or biome-wise, effects could be loss of ponds and lakes due to permafrost thaw and/or increased loadings of DOM in the boreal biome or salinisation. Also, local but widespread changes such as anthropogenic eutrophication of lakes in populated areas would have large-scale impacts. Lakes worldwide show a warming trend, hence a global impact. Source: (Hessen et al., 2023, in review).

It is well established that lakes are sensitive to the effects of climate change, including warming and changes in precipitation and storminess (Meerhoff et al., 2022). Emerging evidence suggests that they may also play an important role in climate regulation, through both the emission of greenhouse gases (predominantly methane – Downing et al., 2021) and carbon burial (Anderson et al., 2020). It is therefore relevant to consider the extent to which potential tipping points may drive, or be driven by, climate change, leading to higher-level feedbacks to the Earth's climate system. In this context we will constrain the discussion to potential tipping points that are more generic, at least with some regional or biome-wise impact, and that could feedback to the climate, while not necessarily being driven or triggered by climate change per se.

Here, we adhere to tipping points as defined in this report (and matching Nes et al., 2016). Based on this we discuss candidate tipping points in freshwaters (Table 1.3.2), focusing on lakes and ponds, with the potential for global or at least regional or biome-scale relevance.

Evidence for tipping dynamics

Eutrophication-driven anoxia and internal P-loading

The mobilisation of P from sediments, a process known as internal loading (Sondergaard et al., 2001), is well described and plays a key role in hysteresis in preventing lakes recovering from human-driven eutrophication (Boström et al., 1982; Jeppesen et al., 1991; Spears and Steinman 2020).

The process may be enhanced by lake warming, and there are feedbacks to climate since water anoxia and internal P-loading (which features the actual tipping point) could offset CO_2-fixation by increased release of GHGs. Consequent changes in biota also strengthen hysteresis (Brabrand et al., 1990), not least when cyanobacterial blooms develop. The phenomenon is local but widespread, and likely to increase as a result of global warming (Meerhoff et al., 2022). Increases in precipitation, and high-intensity rainfall events, are also expected to significantly increase runoff of P from agricultural catchments to surface freshwaters (Ockenden et al., 2017), further promoting eutrophication and its manifestations. Warming increases stratification and thermal stability promoting anoxia (Maberly et al., 2020; Woolway et al., 2020), internal fertilisation and increased GHG emissions. In addition to anoxia, there are other feedback mechanisms for lake eutrophication tipping points, such as the macrophyte-nutrient-algae-turbidity and macrophyte-zooplankton/fish-algae-turbidity loops (Wang et al., 2022). Shifts in trophic cascades, i.e. a top-down control of zooplankton and reduced grazing on phytoplankton, could also help drive eutrophication (Carpenter et al., 1985; Carpenter and Kitchell 1988). However, feedback to the climate is primarily related to anoxia.

Increased loading of DOM and anoxia

Increased export of terrestrially derived dissolved organic matter (DOM) to lakes and rivers in boreal regions ("browning") is a widespread phenomenon partly linked to reduced acidification, but also driven by land use changes (notably afforestation) and climate change (CO_2-fertilisation of forests, warming and hydrology) (de Wit et al., 2016; Creed et al., 2018; Monteith et al., 2023). Wide-scale regime shifts in boreal lakes caused by increased loadings of DOM can promote a prolonged and more intensified stratification period (implications summarised above, described for DOM by Spears et al., 2017), amplified by warming. Increased terrestrial DOM loadings intensify net heterotrophy in the systems (i.e. through increased light attenuation and increased access to organic carbon) (Karlsson et al., 2009; Thrane et al., 2014; Horppila et al., 2023). While at present the thresholds around these effects have not been well constrained, the impacts may be significant at the global scale for GHG emissions (Tranvik et al., 2009) and regionally for coastal productivity (Opdal et al., 2019)

Both eutrophication and browning are to some extent driven by climate change, and warming of lakes will promote the effects by increasing thermal stratification, promoting anoxia which again promotes internal loadings of phosphorus, leading in some cases to self-sustaining change (i.e. tipping). Increased release of GHGs will serve as another feedback to the climate (Fig. 1.3.14).

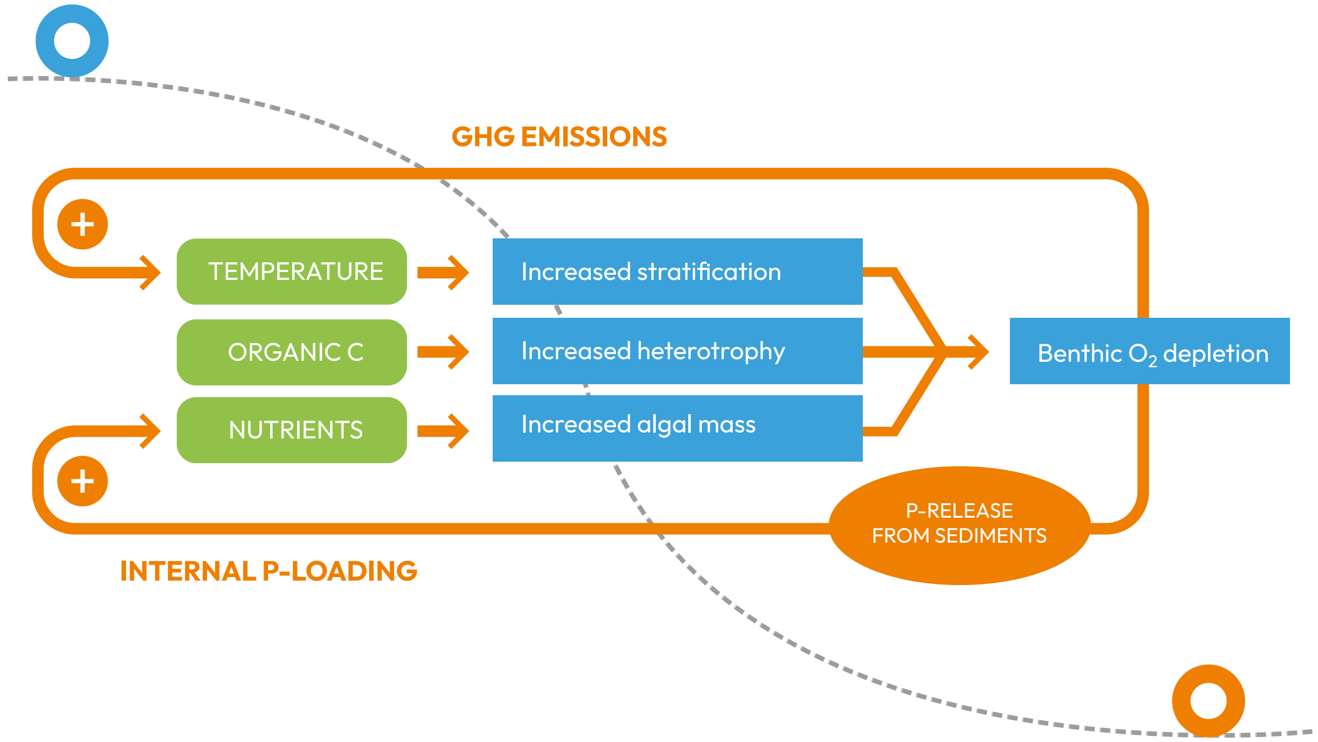

Figure 1.3.14: The interactive role of eutrophication, DOM-export (browning) and warming on lakes. Separately or combined they promote benthic O_2-depletions which cause an internal feedback by P-loading from sediments and a climate feedback via release of greenhouse gases. The potential shift between states (blue to red circle) is indicated. Adapted from: (Hessen et al., 2023, in review).

Disappearance/appearance of waterbodies

A global reduction in lake water storage (Yao et al., 2023) and climate-related creation or, more frequently, disappearance, of water bodies is a large-scale concern (Woolway et al., 2022). For example, current and future permafrost thaw and glacier melting can both create new and drain old waterbodies, providing a strong link to the fate of the cryosphere (Smith et al., 2005; Olefeldt et al., 2021). Such small but numerous waterbodies over vast areas in the high Arctic may also serve as major conduits of greenhouse gases and historical soil carbon stocks to the atmosphere (Laurion et al., 2010) and play an important role in mediating nutrient delivery to the polar oceans (Emmerton et al., 2008), potentially affecting global productivity (Terhaar et al., 2021).

Despite the scale considered here, the extent of open water globally is relatively easy to quantify using remote sensing, and loss of waterbodies can be predicted from water balance and thresholds for permafrost thaw with high confidence. However, while representing a binary shift between two states, driven by climate, this should not be classified as tipping events as in most cases no self-sustaining feedback is involved. Lake appearance or disappearance can be driven by cryosphere tipping points though – for example, thermokarst lake formation or abrupt drainage due to permafrost thaw (Turetsky et al., 2020; Teufel and Sushama, 2019) (see Chapter 1.2) – and in such cases the lake forms part of a coupled thermokarst system capable of tipping.

Switch from N to P-limitation

Regions receiving increased nitrogen (N) deposition may shift from prevailing P- to N-limitation (Elser et al., 2009). Conversely, increased N-loss by denitrification, eventually associated with increased internal P-loading, may shift systems from P to N-limitation (Weyhenmeyer et al., 2007). Changes in N- versus P-limitation of productivity are associated with changes in community structure, both for the phytoplankton and macrophyte communities, which could involve ecological tipping points. However, while the switch between N and P-limitation represents a binary switch with ecological consequences, it is not itself classified as a tipping point according to our criteria, as self-sustaining feedbacks have not been identified. There is currently weak evidence for this shift's impact on climate feedbacks.

Salinisation

Salinisation is a prevalent threat to freshwater rivers, lakes and wetlands and is caused by a range of anthropogenic actions including water extraction, pollution and climate change (Herbert et al., 2015). It has severe consequences for aquatic communities (Short et al., 2016, Cunillera-Montcusí et al., 2022) with salinity thresholds likely strongly impacted by other stressors – including eutrophication (Kaijser et al., 2019). Salinisation has a strong societal impact, particularly related to domestic and agricultural water supply in arid and semi-arid regions (Williams et al., 1999). Salinisation tends to decrease CH_4 emissions (Herbert et al., 2015) and, in that sense, is a negative/damping feedback with respect to climate change. Salinisation may induce ecological regime shifts, for example leading to microbial mat dominance (Sim et al., 2006), and results in some hysteresis, with salinised sediments remaining salty also after the system is flushed with fresh water (Van Dijk et al., 2019), but is not in itself driven by self-sustaining feedbacks.

Spread of invasive species

Freshwaters are especially vulnerable to species loss and population declines as well as species invasions due to their isolation. Substantial ecosystem changes by reinforcing interactions between invasive species and alternative stable states (i.e. macrophyte – aquatic plant – ***versus*** phytoplankton dominance, as described above) may occur (Reynolds and Aldridge 2021). The spread of several invasive species can be facilitated by climate change (Rahel and Olden, 2008) and may have some self-sustaining properties. Such changes could thus drive a regime shift for a given system, but in most cases are hypothetically reversible if the original driver (the invasive species) were removed. Species invasion is hard to predict and difficult to quantify, despite the risk of species ingress as ranges expand with climate change.

Assessment and knowledge gaps

Table 1.3.2: Candidate tipping events from the literature with potential to occur at local to regional scales, their association with climate change, and whether tipping points and hysteresis have been associated with them. Brackets indicate higher uncertainty. Bold entries represent categories that qualify as tipping points in this context, while the others are either simply binary shifts between states, threshold effects, or similar.

Type of event	Local	Regional	Climate driver	Climate feedback	Tipping event	Hysteresis
Eutrophication-driven anoxia and internal P-loading	**x**		**x**	**x**	**x**	**x**
Increased loadings of DOM		**x**	**x**	**x**	**x**	**(x)**
Disappearance/ appearance of waterbodies		x	x	x	(x) (linked to cryosphere tipping)	(x)
Switch between N and P limitation		x	x	(x)		
Salinisation		x	x	x		(x)
Spread of invasive species	x	(x)	(x)			(x)

Abrupt changes driven by warming, eutrophication or increased loadings of organic matter, leading to changes in the production to respiration ratio (i.e. systems shifting from net autotrophic to net heterotrophic), and/or onset of bottom-water anoxia have clear tipping dynamics (high confidence) and strong feedback to the climate via GHG emissions (Meerhoff et al., 2022) (Table 1.3.2). Whether the widespread effect of increased loading of organic matter (browning) in boreal lakes can drive tipping points is more of a knowledge gap, yet the feedback of lake browning to climate through increased GHG emissions is evident.

Loss of waterbodies residing on permafrost or suffering negative water balance and eventually complete disappearance represents a binary shift, which has major ecological consequences (Woolway et al., 2022) but is not considered a tipping event ***sensu stricto***. The same holds for other types of binary shifts, threshold effects or local changes. The role of warming as a catalyst on the changes driven by eutrophication and browning is a critical knowledge gap. Quantification of GHG release from lakes represents major feedbacks to climate, and to quantify the impact of eutrophication, browning and warming in this context should have high priority.

1.3.2.7 Coastal ecosystems

In this section we consider ecosystems bordering the land and ocean, covering the 'littoral' intertidal and subtidal zones. These zones include some of the most biodiverse and human-depended ecosystems on Earth, despite occupying globally tiny areas: warm-water coral reefs, mangrove forests and seagrass meadows. However, all face increasing pressures from increasingly frequent climate change-induced extremes compounded by habitat destruction, pollution and sea level rise.

Warm-water coral reefs

Warm-water coral reefs span the Earth's tropical and subtropical ocean, and are estimated to support over half a billion people for their livelihoods and over a quarter of marine species for part of their lifecycle (Wilkinson et al., 2004; Plaisance et al., 2011) (Figure 1.3.15). They can cross a threshold of ecosystem collapse when they cease to have sufficient cover (typically ~10 per cent) and diversity of hard corals to support the wide diversity of species taxa and ecological interactions typical of a coral reef (Bland et al., 2018; Darling et al,.2019; Sheppard et al,.2020; Perry et al,.2013; Vercelloni et al., 2020). Coral reef collapse is an ecological phenomenon at local scales; here we explore where localised coral reef collapse aggregates to the scale of regions, potentially irreversibly, and potentially to a global scale.

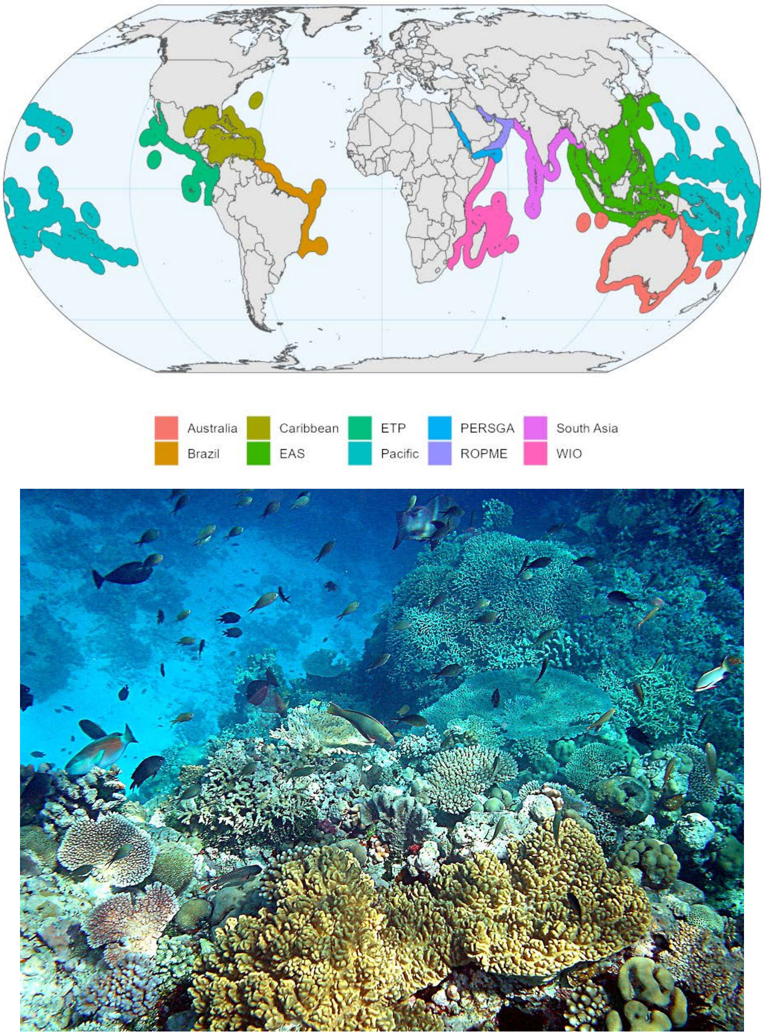

Figure 1.3.15: Global distribution of warm water coral reefs and key reef regions (top). ETP is the Eastern Tropical Pacific, PERSGA is the area included within the Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden, ROPME is the sea area surrounded by the eight Member States of the Regional Organisation for the Protection of the Marine Environment, and WIO is the Western Indian Ocean. A coral reef ecosystem in Papua New Guinea in 2003 (bottom). Credit: (Souter et al., 2021) and (top), Brocken Inaglory via Wikimedia (bottom).

Thermal stress, driven by increasingly warmer oceans and superimposed El Niño extreme events, is the primary driver of regional-scale mortality of hard corals (Hughes et al., 2017; Houk et al., 2020). Coral 'bleaching' occurs when thermal stress causes corals to expel the symbiotic algae that provides them with food (resulting in a characteristic loss of colour), and can result in death if it occurs frequently enough to prevent recovery (Hughes et al., 2018a, 2018b, Obura et al., 2022).

However, a wide variety of interacting and synergistic threats co-occur (e.g. ocean acidification, overfishing, pollution, invertebrate predators and sea level rise), generally lowering the thermal threshold for bleaching and/or mortality, bringing forward timing of collapse, or even surpassing thermal stress in local importance (Ban et al., 2013; Edmunds et al., 2014; Darling et al., 2019; Cramer et al., 2020; Dixon et al., 2022; Setter et al., 2022). Coral mortality may play out over weeks to a few months (for e.g. thermal stress-induced bleaching, for example), or years (for chronic threats such as diseases and land-based impacts), but prolonged failure to recover over a decade or more is necessary to qualify a coral reef as 'collapsed'.

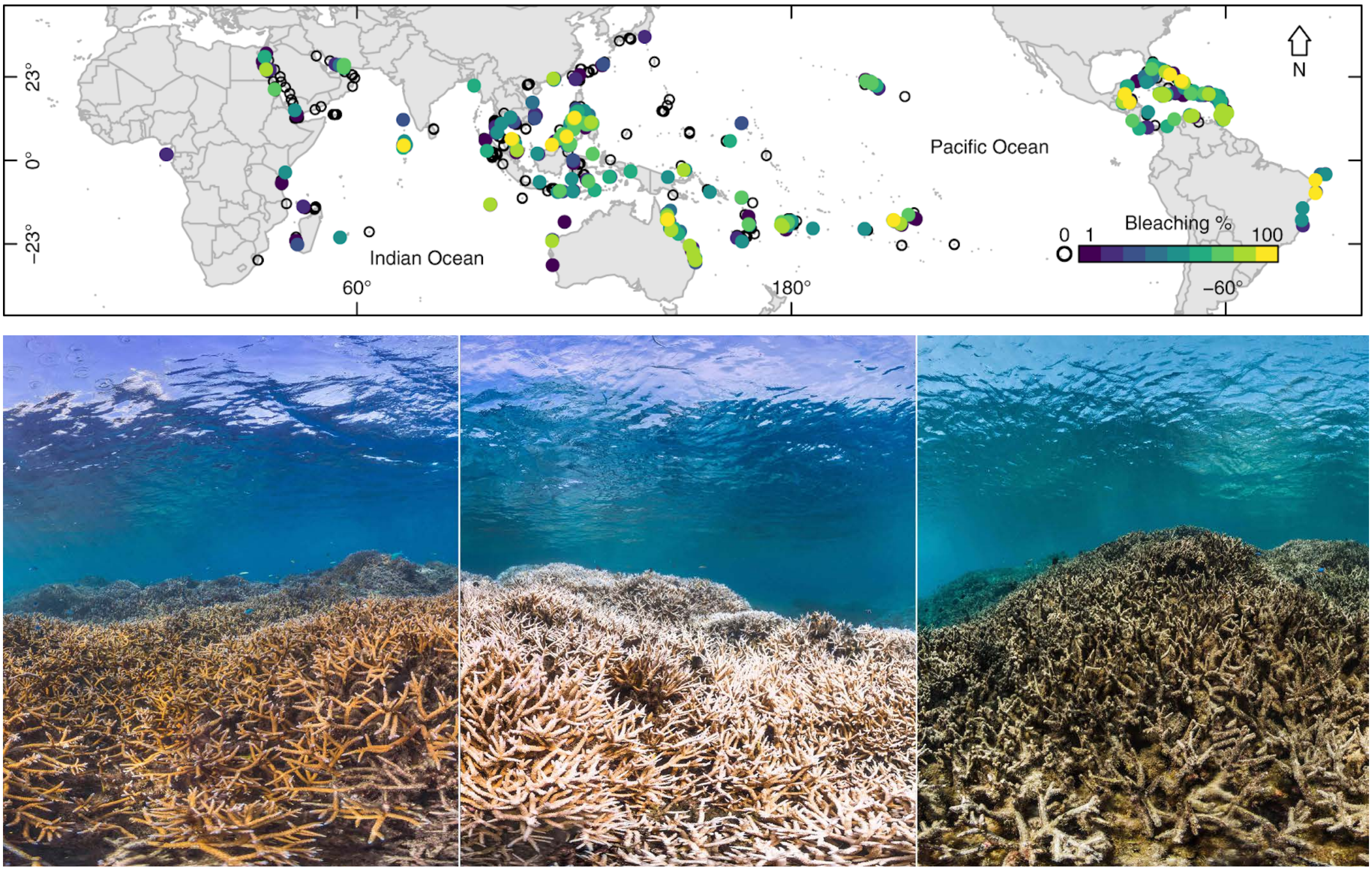

Figure 1.3.16: Map of recent coral reef bleaching distribution (as a percentage of the coral assemblage bleached at surveys from 1998 to 2017, with white circles indicating no bleaching. and coloured circles from 1% (blue) to 100% bleaching (yellow)) (top). Photos showing impact of coral bleaching in American Samoa before (left), during (middle), and after (right) the 2015 bleaching event (bottom). Credit: (top) (Sully et al., 2019), (bottom) from The Ocean Agency.

Localised coral responses to increasing stressor magnitude and intensity are now aggregating at scales exceeding 1,000km and manifesting as regional die-offs (e.g. western and central Indian Ocean, Great Barrier Reef, Mesoamerican Reefs) (Le Nohaïc et al.,2017; Amir, 2022; Muñiz-Castillo et al., 2019; Obura et al., 2022), with most reef regions having experienced multiple mass coral bleaching and die-off events (Darling et al., 2019; Cramer et al., 2020) (Figure 1.3.16). Around 50 per cent of global coral reefs are estimated to have been lost over the past 50-150 years (IPBES 2019), with estimated loss of 16 per cent in 1998 (Wilkinson et al., 1999) and measured loss of 14 per cent from 2009-2018 (Souter et al., 2020), but with high variance among regions.

Projected loss of coral reefs has been estimated in varied ways. Dominant projections are of 70-90 per cent loss of coral reefs at 1.5°C and ~99 per cent at 2°C warming (Cooley et al., 2022). The average year for projected global annual severe bleaching under SSP2-4.5 (a trajectory close to current projections) is 2045, which is delayed 30 years if corals can adapt to an additional 1°C of warming (UNEP 2020). A shift occurs from 84 per cent of reefs globally having 'good' thermal regimes in 1986-2019 to 0.2 per cent in 2100 at projections of 1.5°C, and 0 per cent at 2°C warming (Dixon et al., 2022). Finally, the proportion of reefs facing 'unsuitable conditions' increases from 44 per cent in 2005 to, under worst case scenarios, 100 per cent by 2055 under any one of several stressors, but by 2035 for cumulative stressors (Setter et al., 2022). Continued ocean warming over several decades (due to lagged ocean heat uptake) and sea level rise over centuries to millennia (due to thermal expansion and ice sheet melt, see 1.2.3) mean some reefs and other coastal ecosystems (see also Mangroves and Seagrassees) may be committed to eventually passing tipping thresholds even if emissions ceased soon (Abrams et al., accepted).

Evidence for tipping dynamics

Failure to recover from mass mortality shows evidence of having crossed a threshold for recovery, which we address for scales above approximately 1,000km, to regional and global scales. A key question is if coral reef decline globally is just an aggregate of regional events, so a linear/chronic decline process (Souter et al., 2021), or if there may be a global tipping point.

Observations on coral reef tipping points include the following:

- The first reported global bleaching event in 1998 was associated with atmospheric warming of ~0.6°C (corresponding to c. 350 ppm CO_2) with a strong El Niño on top (Veron et al., 2009), past which more frequent, intense and widespread coral bleaching and mortality has occurred.
- A very high risk of impact to corals was assessed by the IPCC as global mean warming levels crossed around 1.2°C (IPCC SR1.5 2018).
- Thermal bleaching tipping points are already being passed in the majority of coral reef regions (Cooley et al., 2022 – see Figure 1.3.17).
- The risk of ecosystem collapse is already predicted at high levels in all coral reef regions assessed. The MesoAmerican Barrier Reef is Endangered (Bland et al., 2018) and Western Indian Ocean coral reefs are Vulnerable to collapse, with two thirds of subsidiary ecoregions being Endangered or Critically Endangered due to projected warming (Obura et al., 2022).

Where are we reaching tipping points in the ocean and what can we do about it?

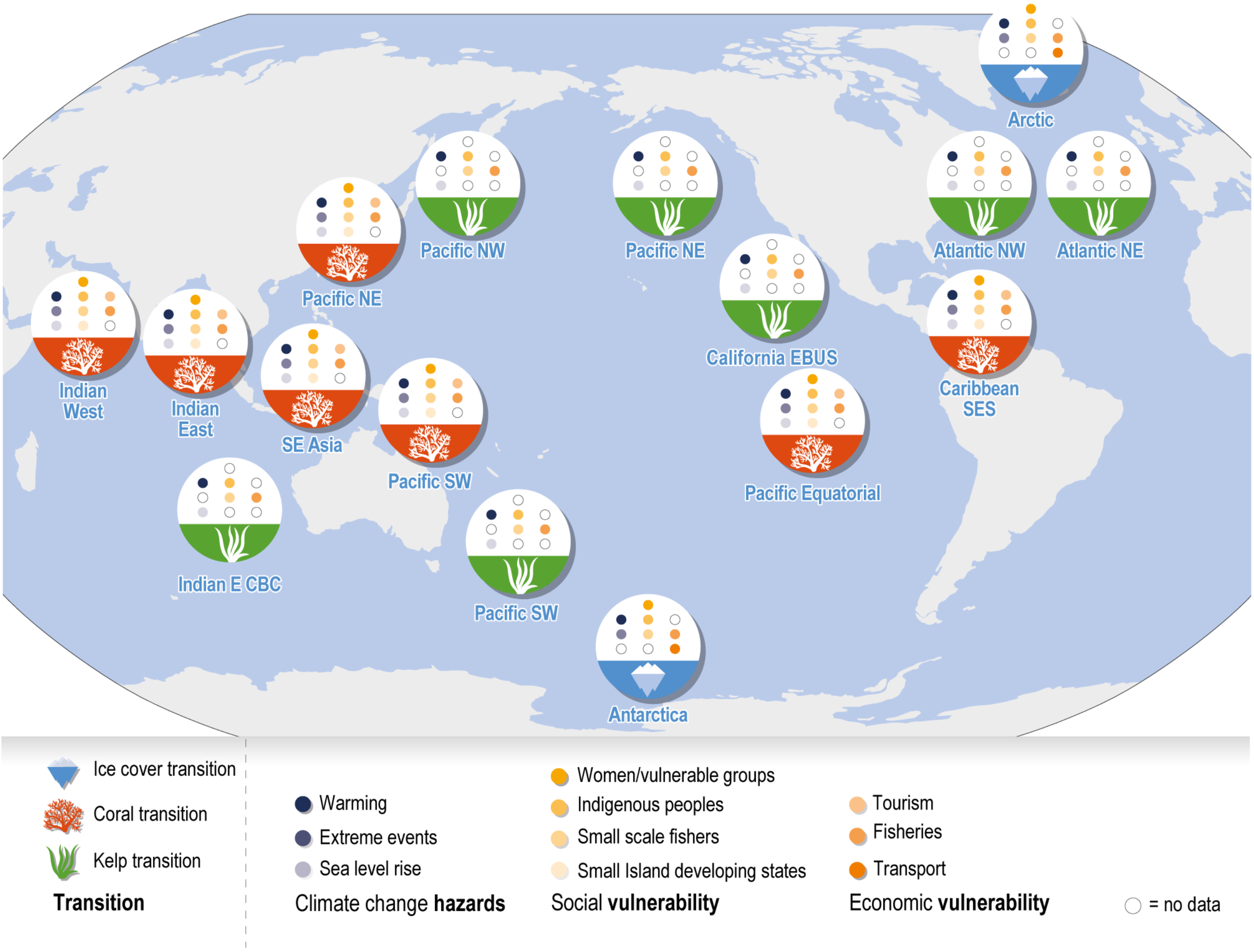

Figure 1.3.17: Tipping points have been passed in many ocean ecosystems, including coral reefs, kelp forests, and those associated with sea ice, with diverse socio-economic implications. [From FAQ3.3.1 in (Cooley et al,. 2022).

Elevated summer ocean heat maxima (over 1-2°C above site-specific individual coral acclimation thresholds) for weeks to months, and larger acute temperature spikes for several days, cause severe coral bleaching and mass mortality. Mass-mortality bleaching thresholds have been proposed at eight “Degree Heating Weeks” (a measure of how long and how much ocean temperatures are above normal) which is likely by ~2°C global warming (McWhorter et al., 2021), or at two bleaching events per decade (likely by ~1.5°C) (Frieler et al., 2013). Mass coral mortality repeated more than twice per decade and over hundreds to thousands of kilometres and larger, is increasingly recognised as giving insufficient time for recovery of impacted populations, and of ecological interactions (Hughes et al., 2018a, 2018b, Obura et al., 2022). However, estimating globally consistent warming thresholds is challenging given variation from individual corals to species and across all spatial scales in acclimation and adaptation ability. Other stressors reduce the ability of corals to resist thermal stress, thus bringing down tipping thresholds.

Increasing frequency and intensity of regional-scale coral mortality events past 1°C warming are suggestive that these coral reef regions have already passed regional bleaching tipping points (Cooley et al., 2022). The potential for thermal refuges for corals under likely future scenarios is doubtful (Beyer et al., 2018; Dixon et al., 2022; Setter et al., 2022) as very few or no reef areas are projected to remain below tipping thresholds of key stressors. The existence of putative refuges at greater depths (Bongaerts and Smith 2019) or higher latitudes (Yamano et al., 2011; Setter et al., 2022) are not strongly supported by recent work (Hoegh-Guldberg et al., 2017; Cooley et al., 2022). Ecological and biogeographical (spatial) positive/amplifying feedback loops prevent local recovery of coral reefs and promote expansion of reef collapses from local to regional scales when surviving corals and coral patches become too spatially separated for successful reproduction of adults, and supply of larvae from surviving to damaged reefs (Hock et al., 2017).

Coral reef decline does not substantially feedback to the climate system on policy-relevant timescales. However, localised surface cooling may arise through increased low level cloud albedo induced by sulphur compounds released by reef metabolism. Consequently, extensive coral die-offs could amplify local warming (Jackson et al., 2020).

Assessment and knowledge gaps

Warm-water coral reefs have localised tipping points (high confidence) and are now experiencing regionally clustered tipping points (high confidence). Based on the evidence collected here, we suggest that the critical threshold of 1.5°C (range 1-2°C) (Armstrong McKay et al., 2022) should be adjusted, narrowing and lowering the range to 1-1.5°C, with a middle estimate of 1.2°C, marked by the multi-year global coral reef bleaching events of 2015-2017 (Cooley et al., 2022; Hoegh-Guldberg et al., 2018; Dixon et al., 2022; Setter et al., 2022). The co-occurence of additional synergistic drivers also support lowering the critical threshold (Willcock et al., 2023) and there is evidence of accelerating collapses at increasing spatial scales (Cooper et al., 2020).

The combined effects of long-term warming, sea level rise, ocean acidification and other stressors bears more investigation to identify the lower critical threshold for the coral reef tipping point. The potential for coral adaptation to warming is a critical but poorly known factor, and subject to high levels of variation locally. The potential effectiveness of restoration for coral reefs at scale, and with enhanced capacity to resist future threats, are both currently poor. The effect of climate migration on coral recovery is not known, with potentially positive effects at higher latitude (with in-migration), but negative at lower latitudes (with out-migration, but no replacement; Herbert-Read et al., 2023).

Mangroves and seagrasses

Mangroves and seagrasses play vital roles in coastal societies and economies. They provide fundamental and hard-to-substitute ecosystem services such as support to fisheries, nutrient cycling, coastal protection and sediment trapping (Malik et al., 2015; Nordlund et al., 2016; Menéndez et al., 2020; Nabilah Ruslan et al., 2022; doAmaral-Camara et al., 2023, James et al., 2023). Located between the sea and the land, their unique dual nature exposes mangroves and seagrasses to climate drivers that arise in both systems (Lovelock et al., 2017a; Duke et al., 2017a, 2019), making them particularly vulnerable to climate change (Duke et al., 2022). Recent attention has focused on their climate mitigation services ('blue carbon') linked to their high productivities and long-term (millennia) storage of organic matter in their sediments, which positions them among the most dense carbon sinks on Earth (Donato et al., 2011; Alongi et al., 2016; Macreadie et al., 2021; Serrano et al., 2021).

While they occupy small areas (c. 140,000 sq km and uncertain c. 266,562 sq km for mangroves and seagrasses respectively in 2020; Bunting et al., 2022; McKenzie et al., 2020; Figure 1.3.18 and 1.3.19), they store up to 12.3 GtC and 3.8 GtC respectively (Macreadie et al., 2021). These ecosystems are natural sinks of CO$_2$, but when degraded they can release CO$_2$, NO$_2$ and CH$_4$, adding to the emissions of the estuaries they are embedded in (Rosentreter et al., 2022). Emissions derive from carbon stored long-term in sediments, which cannot be recovered in a lifespan and is therefore additional to the current atmospheric balance (Lovelock et al., 2017b; Schorn et al., 2021; Romero-Uribe et al., 2022).

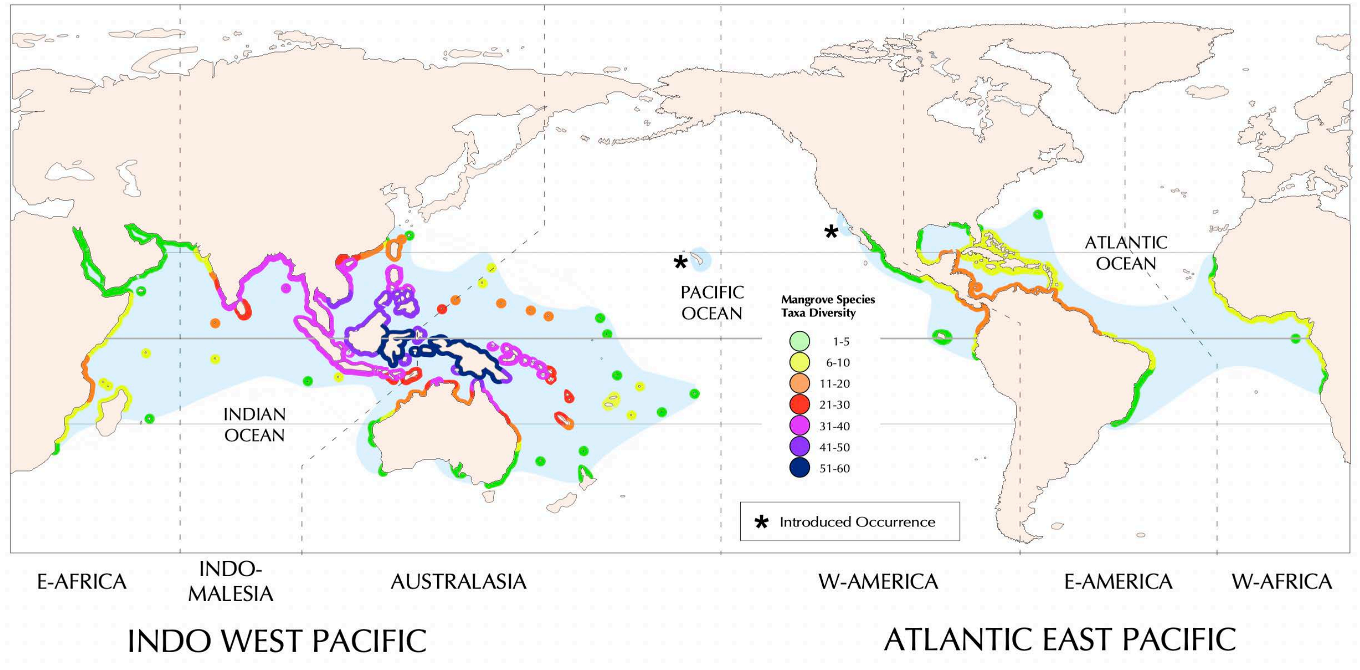

Figure 1.3.18: Upper panel: floristic distribution of mangroves in the world, with a marked diversity in the Wallacea region (Indo Pacific). Lower panel: white mangrove (*Laguncularia racemosa*) from Yucatan, showing the intricacy of mangrove roots, and their service as fish habitat, coastal protection against storms and sediment trapping. Source: (Duke et al., 2017) (top) and Jorge Herrera, CINVESTAV (bottom).

Mangroves and seagrasses are historically among the most human-threatened ecosystems in the world (Valiela et al., 2001; Waycott et al., 2009), with 35-50 per cent of mangroves' original cover now lost, mainly to aquaculture and agriculture (Richards and Friess, 2016, Goldberg et al., 2020; Hagger et al., 2022), while other factors including nutrient overload, invasive species, and ocean warming have led to a 19-30 per cent decrease of the original seagrass surveyed area (Waycott et al., 2009; Dunic et al., 2021).

In spite of this, the magnitude of their past and current feedback to global warming remains uncertain (Rosentreter et al., 2022). Under current rates of deforestation, estimates of global mangrove emissions by the end of the century range between 0.24 to 0.34 Gg CO_2e if foregone soil carbon sequestration is also included (Adame et al., 2021), which is comparable to the European Union's emissions in 2022. Southeast and South Asia (West Coral Triangle, Sunda Shelf and the Bay of Bengal) are projected to lead the emissions, followed by the Caribbean (Tropical Northwest Atlantic), the Andaman coast (West Myanmar), and northern Brazil (Adame et al., 2021).

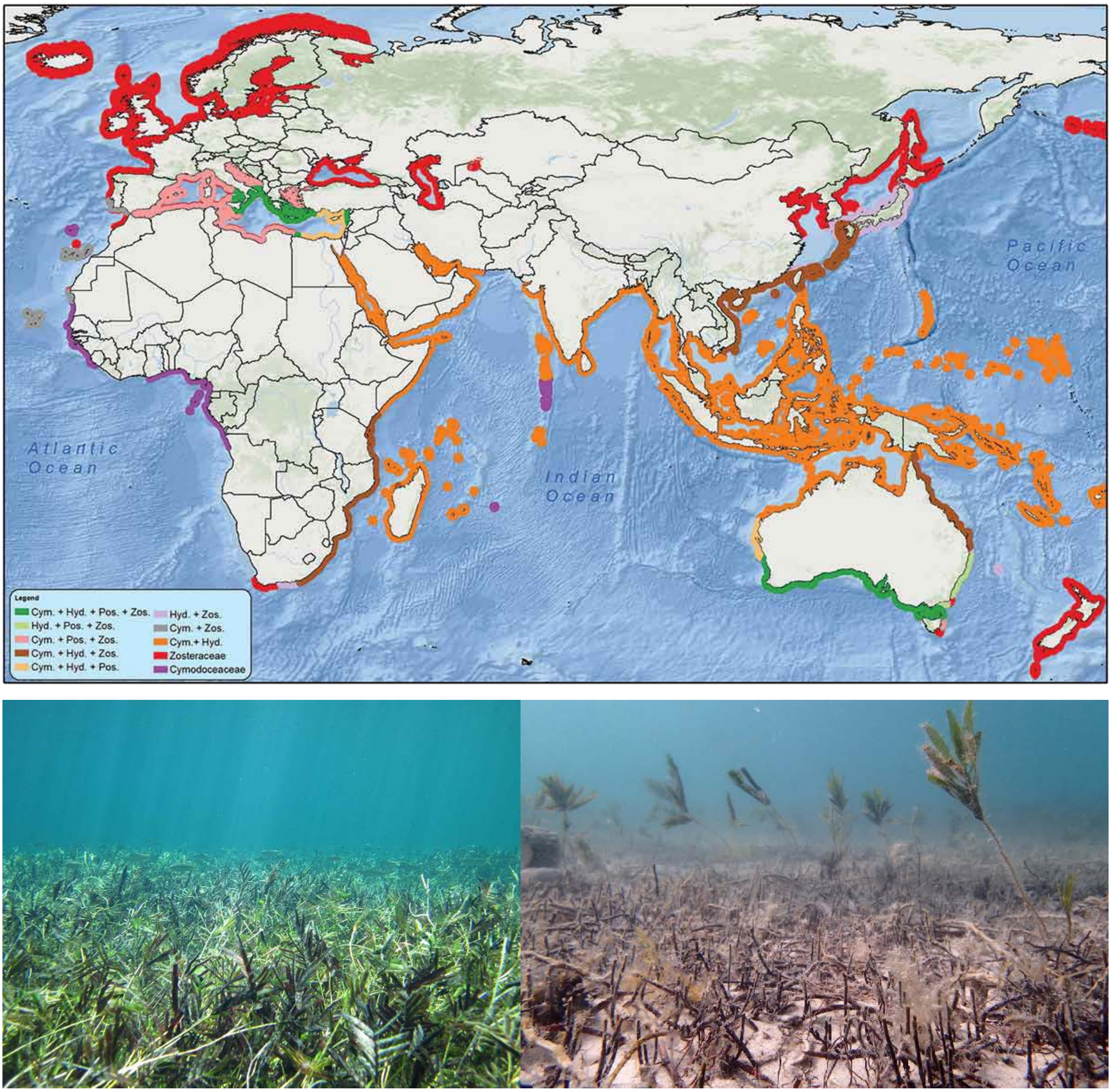

Figure 1.3.19: Upper panel: global distribution of seagrasses. Lower panel: Shark Bay temperate seagrass (*Amphibolis antarctica*) before the 2011 heatwave and after (2013). Revisits from 2012 to 2014 verify poor recovery of A. antarctica, and the slow expansion of the tropical seagrass *Halodule uninervis*, in sites with no recovery (30% of cover three years later). Source: IUCN, map created by T. Bakirman. Seagrass die-off: credit goes to the Shark Bay Ecosystem Research Project and (Nowicki et al., 2017).

Evidence for tipping dynamics

In spite of major historical habitat loss and degradation, there are not yet generalised signs of irreversible global transitions of mangroves towards alternative states such as tidal flats, and the remaining systems have so far retained large-scale stability in the tropics. Bistability is, however, observed in northern subtropical distributions with mangrove encroachment over tidal marshes where freezing events are now rarer (Feller et al., 2017; Hesterberg et al., 2022). Observational data also suggests rainfall-induced bistability of mangroves and salt marshes (Duke et al., 2019).

Scarce global monitoring prevents robust analyses of seagrass trends, but transitions (>50 sq km) towards unvegetated sediments have intensified in many coastal regions in the last two decades (e.g. Europe, Australia, US, Caribbean) (Waycott et al., 2009; Carr et al., 2012; Arias-Ortiz et al., 2018; Duarte et al., 2018; Kendrick et al., 2019; Cooley et al., 2022; MacLeod et al., 2023) (Fig 1.3.19). For temperate regions, bistability and tropicalisation of temperate seagrass species are observed in edge-of-range meadows, with uncertain stability trends (Bartenfelder et al., 2022). For tropical seagrasses, local resilience after disturbance has been observed when enough time and reduced pressures apply (MacLeod et al., 2023).

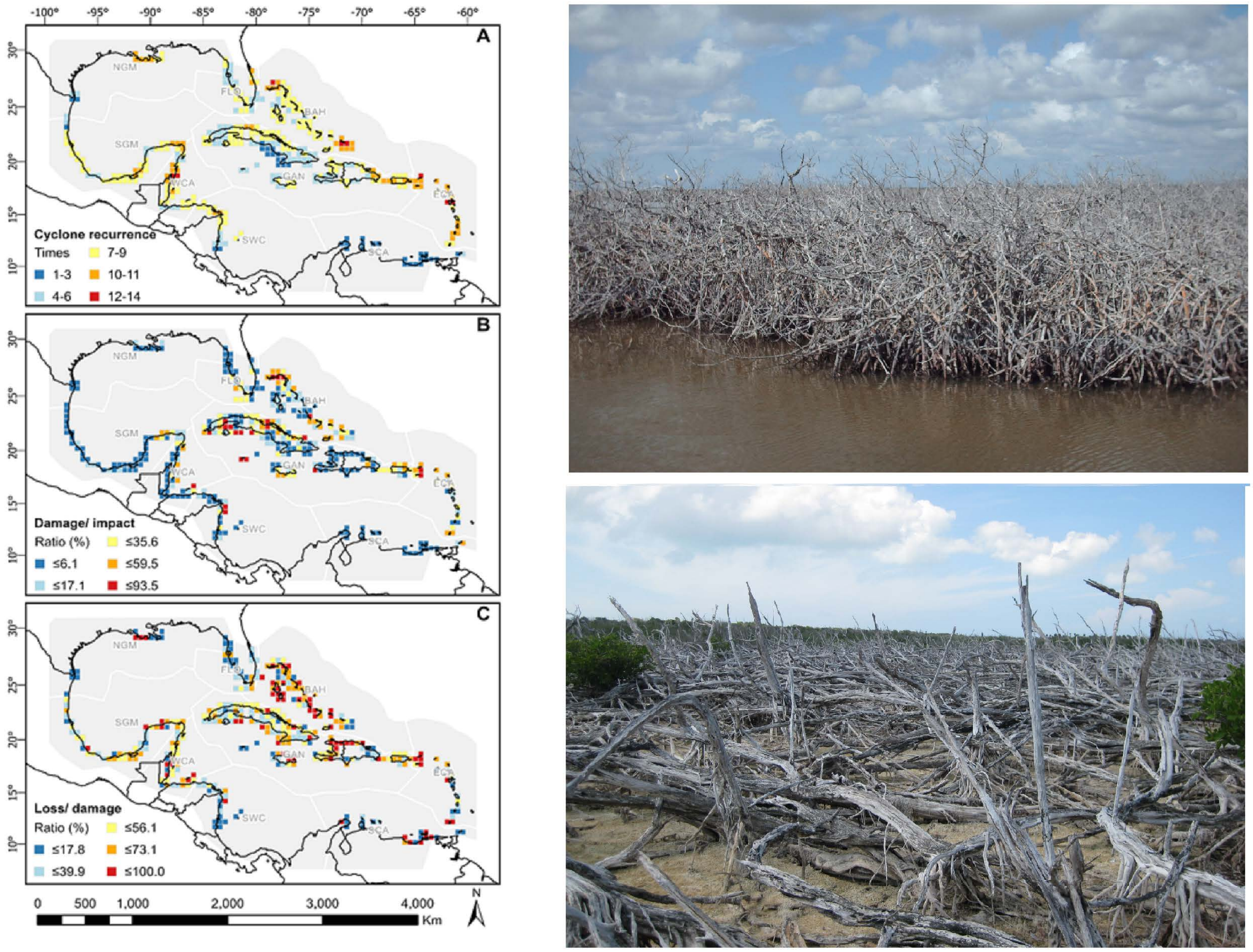

Figure 1.3.20: Left panel presents (A) the recurrence of tropical cyclones (from tropical storms to hurricanes category 5) in different subregions of the North Atlantic Basin (Caribbean, Gulf of Mexico, Mesoamerica), (B) percentage of pixels hit by a tropical cyclone where mangroves show damage six months after the pass of the storm (vulnerability), and (C) percentage of pixels that showed damage after a storm that do not show signal of recovery one year after being damaged (resilience). Right panel includes photos from mangroves hit by hurricanes in Yucatan. Sources: (Amaral et al., 2023) (left panel), Jorge Herrera, CINVESTAV (right panel).

While the resilience of these systems (particularly mangroves) does not yet seem compromised at the global scale, there is increasing evidence of region-dependent declines in resilience for both seagrasses (Dunic et al., 2021; Turschwell et al., 2021) and mangroves (Bergstrom et al., 2021, Friess et al., 2022; Amaral et al., 2023, Duke et al., 2023 in press). These responses relate to:

i) An increased exposure to more frequent and intense extreme events such as hazardous cyclonic activity (Figure 1.3.20), more frequent and intense El Niño (Figure 1.3.21) and marine heatwaves (Fig 1.3.19), which add to the long existing human pressures (nutrient overloads, land use changes, sedimentation rates, etc) and to the long-term environmental impacts that promote mangrove and seagrass mortality (including sea level rise, ocean acidification, ocean/atmosphere warming, regional drought, salinity, hypoxia, diseases and invasive species) (Waycott et al., 2009; Krauss et al., 2014; Lovelock et al.,2015; Feller et al., 2017; Duke et al., 2021; Friess et al., 2022; MacLeod et al., 2023).

ii) Shortened recovery times below re-establishment needs. Post-disturbance recovery has been reported to take ca. 10-20 years depending on the ecosystem service considered (Lugo 1980; Jimenez et al., 1985; MacLeod et al., 2023), with mangrove recovery taking c. 20 years (more on arid climates), and c.10 years for seagrasses. A decade has been considered the absolute minimum successful re-establishment time for both systems, if pre-disturbance conditions (hydrological stability and seed sources) were retained (Lugo 1980; Teutli-Hernandez et al., 2020, Duke et al., 2023 in press; MacLeod et al., 2023). Revisiting times are currently below these thresholds in many regions,

iii) Unprecedented increases in compound extreme events that precede, succeed, or coincide in time and space and amplify ecosystem responses (Allen et al., 2021). Along this line, magnified mangrove mortality due to drought-hurricane duos has already been reported in the Caribbean (Taillie et al., 2020; Amaral et al., 2023).

iv) Exposure to multivariable extreme pressures (Fig 1.3.22). While models frequently focus on a few independent-forcing variables, in reality multiple amplifying, synergistic or antagonic effects occur among stressors. As an example, El Niño combines multiple variables such as heat, drought, flooding, more extreme oscillations in sea level (e.g. Taimasas in the Indo-Pacific), and marine heatwaves, whose combined interaction amplifies mangrove and seagrass mortality.

Decreasing resilience enhances damages in coastal habitats, including severe losses of biodiversity, collapse of regional fisheries and aquaculture, and reduced capacity of habitat-forming species to protect shorelines, preventing re-establishment (Cooley et al., 2022). These make mangroves and seagrasses likely candidates for regional tipping points, with major social and economic consequences.

Additionally, lagged ocean warming (over decades) and sea level rise (over centuries) mean coastal ecosystems will continue to face increasing pressure after atmospheric warming stabilises, meaning tipping can be committed decades before it is realised (see warm-water coral reefs above).

Figure 1.3.21: Mangrove die-off in physiologically stressed mangrove systems after intense El Niño-driven droughts (2015-2016, 2019) combined with other interacting stresses (prolonged ocean retreat in the Indo Pacific, previous eutrophication in the Bay of Panama, timber extraction, etc). a) El Niño 2015-2016 effects over Australia's Gulf of Carpentaria (8,000 hectares of affected mangroves), b) mangrove die-off in the Maldives has been reported in 11 islands since mid-2020, c) mangrove die-off in the Bay of Panama (Juan Diaz site) after the 2015-2016 El Niño on an eutrophic, rapidly sedimented and colonised site. Sources: Norman Duke (James Cook University), Steve Paton (STRI-Panama), Save Maldives Campaign and Neykurendhoo Island Council (2020).

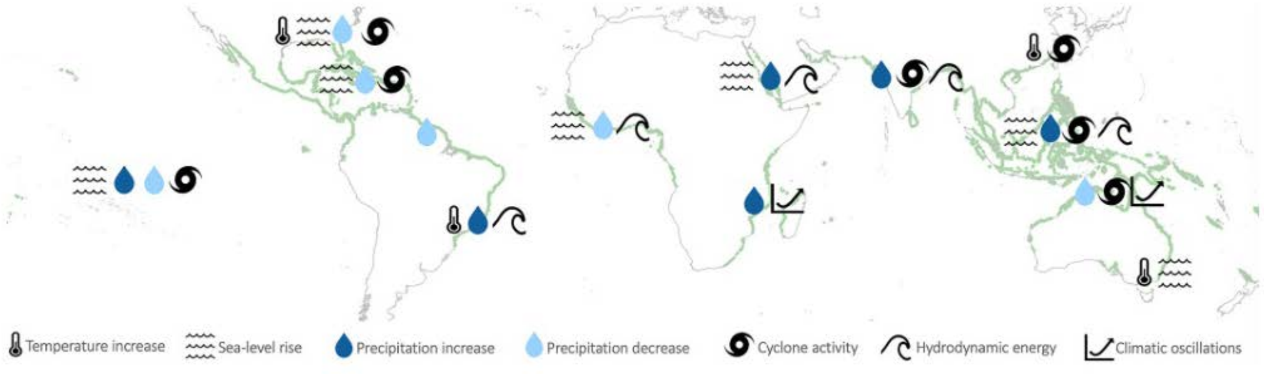

Figure 1.3.22. Regional differences in climate drivers (long-term trends and extreme events) leading to mangrove impacts. Combined with human and other environmental impacts, they are expected to lead to different regional tipping timings and degradation speeds. Source: (Friess et al., 2022).

On the potential tipping dynamics of coastal systems, the IPCC AR6 chapter on ocean and coastal ecosystems (Cooley et al., 2022) noted “irreversible phase shifts with global warming levels >1.5°C, making both systems at high risk this century even in <1.5°C scenarios that include periods of temperature overshoot beyond 1.5°C (high confidence). Mangroves, under SSP1-2.6, are expected to be unable to keep up with sea level rise by 2050, with ecological impacts escalating rapidly beyond 2050”.

(Saintilan et al., 2020, 2023) found it very likely that mangroves were unable to initiate sustained accretion when relative sea level rise rates exceeded 6.1 (4-7) mm/year. This threshold is likely to be surpassed on low-latitude tropical coastlines within 3-5 decades under high-emissions scenarios (Sweet and Park 2014; Saintilan et al., 2020, 2023). For seagrasses, the IPCC AR6 (Cooley et al., 2022) projects contractions of temperate edge-ranges (e.g. ***Zostera costera*** seagrasses in the US would retract by 150-650km under RCP2.6 and RCP8.5, respectively and ***Posidonia oceanica*** in the Mediterranean Sea, which might lose as much as 75 per cent of their habitat by 2050 under RCP8.5 and become functionally extinct by 2100). Marine heat waves will escalate seagrass responses, with moderate responses to sea level rise (Cooley et al., 2022).

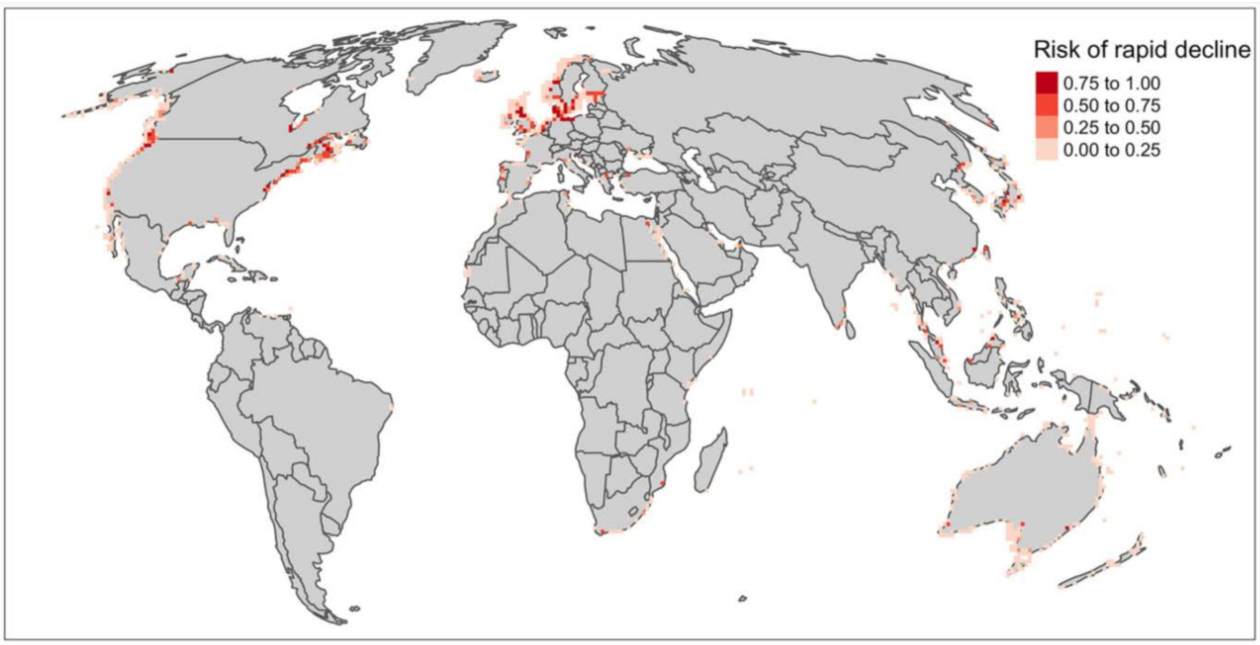

Fig 1.3.23: Rapidly declining trajectories of seagrass meadow extent (>25% loss from 2000 to 2010) predicted in 100×100 km grid cells. Sites are coloured by the probability of a site being ranked among the 10% of sites most likely to have a rapidly decreasing trajectory. Predictions were most strongly associated with high pressures from destructive demersal fishing and poor water quality. Source: (Turschwell et al., 2021).

Assessment and knowledge gaps

We conclude with medium confidence that, under current relative sea level rise projections, subsidence, expected increases in extreme events and coastal development (Cooley et al., 2022), tipping responses for mangroves are likely to be regionally visible by 2080 at temperature thresholds between 1.5-2°C (starting with physiologically stressed regions that host increasing extreme events – also medium confidence). Seagrasses are likely (medium confidence) to show region-dependent die-off responses earlier (by mid century) due to more intense and recurrent marine heatwaves, nutrient pollution and turbidity, at global temperature thresholds closer to 1.5°C (medium confidence).

We have high confidence that tipping responses will be region and site-dependent with diverse timings and degradation speeds. For mangroves, physiologically stressed regions such as arid or highly seasonal climates like the Middle East or the dry corridor of Central America, karstic systems such as the Caribbean, small islands, northern Australia, or the northern Coral Triangle are likely (medium confidence) to show tipping responses earlier than other regions such as the Indo-Pacific, South America or parts of the Indian Ocean, whose systems either have more species, are less exposed, or are less vulnerable to hazard exposure (e.g. there is more space for encroachment, or more refugia).

For seagrasses, temperate regions are predicted to be more vulnerable to tipping than warmer regions (Turschwell et al., 2021; Green et al., 2021; Cooley et al., 2022) (Fig. 1.3.23). Seagrasses in warm regions that are more exposed to water pollution, turbidity, extreme events (marine heat waves and cyclones), coastal development, salinity or invasive species are expected to tip earlier than seagrasses in other warm regions.

Compared to the IPCC AR6 report (Cooley et al., 2022), we highlight a higher confidence on the directional effects of storms on both mangroves and seagrasses towards regionally synchronous mortality (Carlson et al., 2012; Wilson et al., 2019; Taillie et al., 2020; Amaral et al., 2023; Duke et al., 2023 in press). Evidence also exists on decreased regional resilience in mangroves after cyclones (Amaral et al., 2023) and transitions to mudflat shifts in areas where storms combine with erosion co-stressors (Bhargava and Friess 2022). Similarly, warming responses in mangroves have a clearer directional trend, with extreme El Niño hot-droughts superimposed onto global warming and regional drought leading to well-known extended mangrove mortality in many regions (Jimenez et al., 1985), including recent reports of die-off in Australia (Duke et al., 2017a), Panama (Fig. 1.3.21) and the Maldives (Save Maldives Campaign and Neykurendhoo Island Council, 2020).

Current modelling does not yet properly cover extreme events or multiple drivers, nor their interactions (Cooley et al., 2022). These gaps are likely leading to an underestimation of their impacts on ecosystems and their long-term resilience thresholds. Resilience responses to enhanced stressors will be region- and site-dependent, but models still need data to properly represent key drivers per region and their interactions, as well as the thresholds of survival of regional ecosystems (Marba et al., 2022).

1.3.2.8. Marine ecosystems and environment

Climate change, pollution and overexploitation are affecting the marine environment at the physical, chemical and biological levels (e.g. Heinze et al., 2021; Jouffray et al., 2020; Bindoff et al., 2019). Pelagic marine ecosystems (defined as the water column from the surface ocean to the seafloor) as well as benthic marine ecosystems (defined as restricted on the seafloor) from the organism to the community level are changing at the same time as the ocean waters are becoming more warm, acidic and deoxygenated. In this section, we outline five potential tipping systems ranging from fisheries collapse and regime shifts in marine communities to ocean water hypoxia and the nonlinear weakening of parts of the ocean's biological pump (Figure 1.3.24).

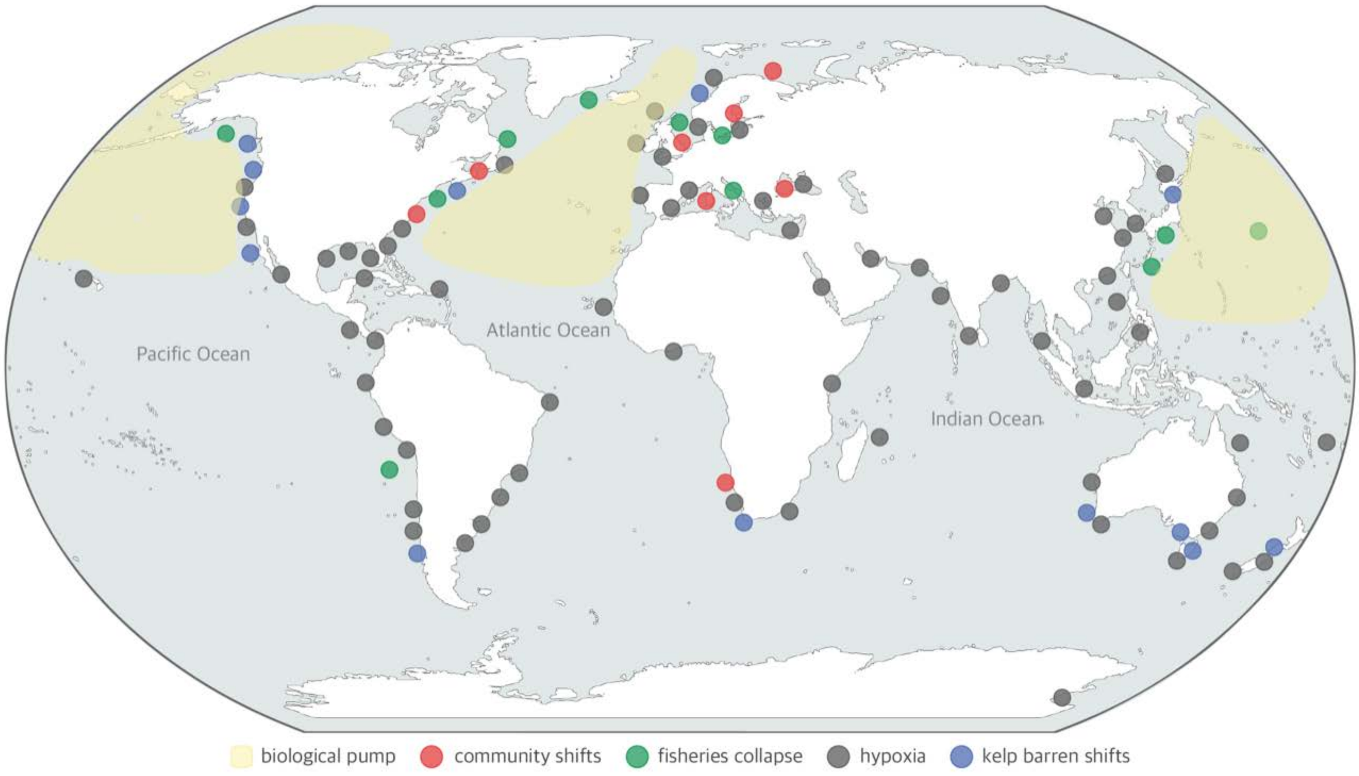

Figure 1.3.24: Locations of reported regime shifts and potential tipping points in the global marine environment. Redrawn and updated from (Blenckner and Niiranen, 2013).

Evidence for tipping dynamics

Fisheries collapse

Over the past decades many fisheries have collapsed primarily due to over-exploitation, but they are increasingly threatened by climate change.

Fish stocks are defined as management units of a species; thus one fish species can have multiple stocks (e.g. more than 20 stocks in the North Atlantic are assessed for Atlantic cod, *Gadus morhua*).

Figure 1.3.25: A school of fish: Credit: iStock.com/armiblue.

Among more than 200 exploited fish stocks, 23 per cent of the species showed at least one stock collapse (biomass below sustainable reference points) (Pinksy et al., 2011). Concerningly, 40 per cent of the collapsed stocks present different regimes of productivity (different relationships between fishing and biomass at different productivity stages) (Vert-pre et al,.2013) that potentially indicate the presence of regime shifts and hysteresis. But, while for some species there is clear evidence of regime shifts (Atlantic cod stocks), for others more studies are needed (Frank et al., 2016; Sguotti et al., 2019).

Fish stock collapses can be due to different feedback mechanisms. The collapse of a stock can induce food web changes (i.e. trophic cascades) that, by modifying the other species of the community and their interactions, can maintain the population at a low level through predation or competition. For instance, large predators such as Atlantic cod may be successful because of the 'cultivation effect': adult cod prey on the juveniles of forage fishes (small pelagic fish which are preyed on by larger predators) that are competitors or predators of juvenile cod. Once the collapse in the biomass of cod occurs, the predation on the forage fish is released and these species start to thrive. Forage fish then prey on juvenile or recruit cod, thus maintaining the population in a depleted state. Examples of this particular dynamic can be found in Newfoundland and also the Baltic Sea (Walters and Kitchell, 2001). Another possible mechanism of hysteresis is the so-called Allee effect, which takes place when recruitment of a population (the process by which new organisms are added to a population) is positively correlated with its biomass. This means that a minimum population size is needed for the population to grow; otherwise it collapses. Thus, if biomass collapses, recruitment will also drastically decline, limiting the capacity of the population to recover. The Allee effect has been shown to be one of the possible hysteresis mechanisms of 13 stocks of Atlantic cod (Winter et al., 2023).

It is difficult to detect specific thresholds in fisheries in general, since every species and every stock within each species is impacted by different levels of the same driver and may experience different pressures. However, it has been shown that, for Atlantic cod stocks, the threshold was created by the combination of multiple drivers, especially warming and fishing (Sguotti et al., 2019; Beaugrand et al., 2022). Specific thresholds need to be detected for every stock.

Beaugrand et al., (2022) have shown that rebuilding cod stocks may depend upon the fishing-environment interaction. When the environment becomes unsuitable at the same time fish stocks collapse, rebuilding the stock may take time or even be impossible so long as adverse environmental conditions persist. This provides an explanation as to why, despite the fishing moratorium near Newfoundland, a partial recovery took more than two decades (DFO 2018). Long-living, slow-growing species might be more prone to irreversibility. For instance, 16 out of 19 Atlantic cod stocks present regime shift dynamics due to fishing and warming and their recovery is hindered by the presence of hysteresis (Sguotti et al., 2019; Möllmann et al., 2022., Frank et al., 2016).

Marine community shifts

Marine community shifts take place when abrupt changes cascade through several species or functional groups of an ecosystem, i.e. the change is not limited to a single species, as in a fish stock collapse, but can cascade all the way from top predators to phytoplankton (Figure 1.3.26).

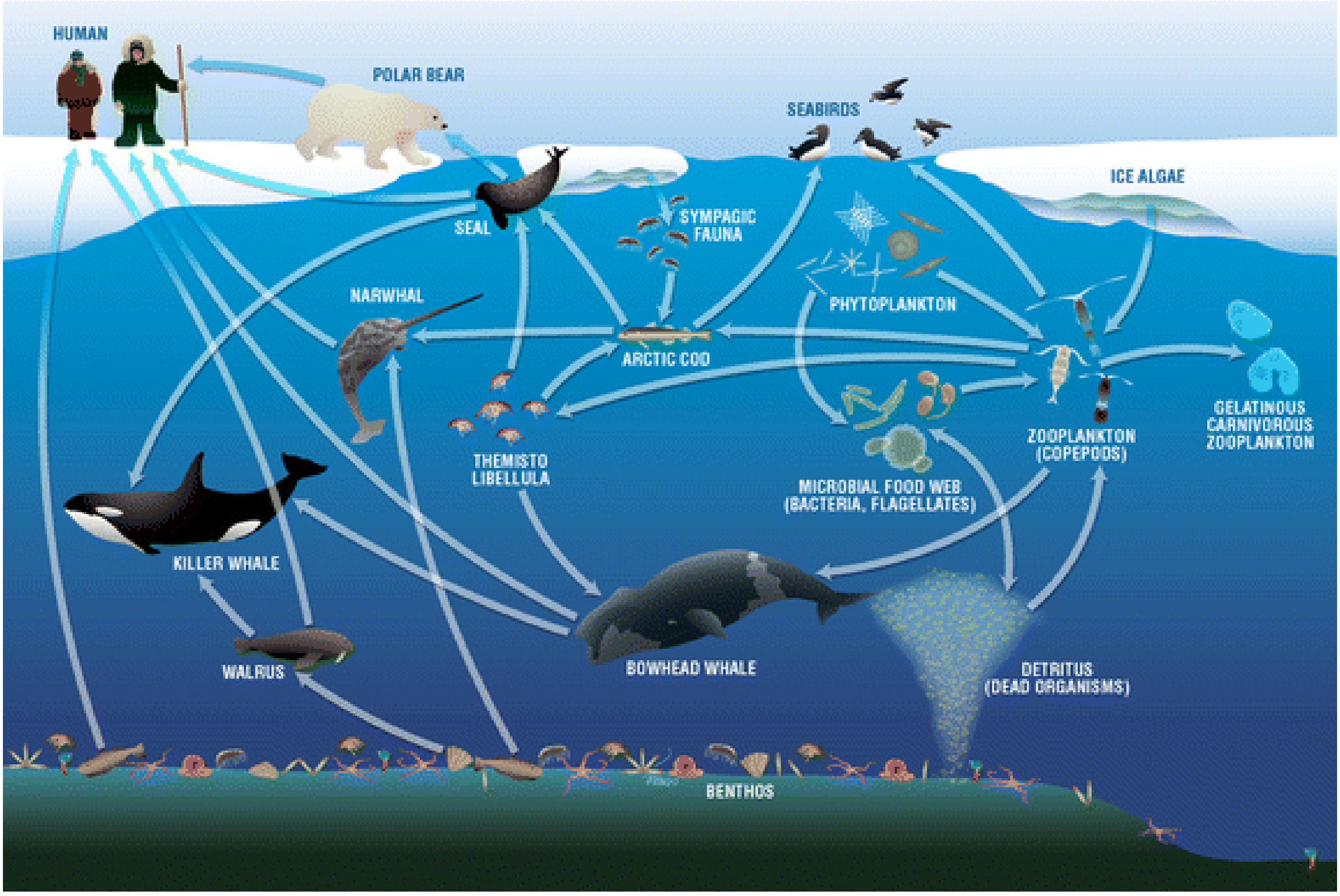

Figure 1.3.26: Schematic of a marine food web. Source: (Darnis et al., (2012).

Many community shifts have been reported in marine ecosystems (Conversi et al., 2015; Beaugrand et al., 2019; Möllmann et al., 2021; Ban et al., 2022; Sguotti et al., 2022). Some ecosystems have even experienced several marine community shifts, such as the Black Sea and Baltic Sea. In the Black Sea, the first major shift started in the end of 1960s with the overfishing of pelagic top predators, enabling surplus phytoplankton and jellyfish production during the following decades, and resulting in increased hypoxia (lack of oxygen necessary for life) followed by collapse of small pelagic fish and domination of jellyfish (Daskalov et al., 2017). In the Baltic Sea, the increased inflow of nutrients and organic matter resulted in the eutrophication of the main basins around the 1950s, enabling higher biological production, but also worsening hypoxia (Österblom et al.,(2007).

Community shifts related to tipping responses mostly occur when the system is controlled by a few key species through trophic cascade (Beaugrand et al., 2015; Daskalov et al., 2007, 2017). Trophic cascades can be environmentally induced or induced by anthropogenic pressures such as pollution or overfishing (Casini et al., 2009). The mechanisms at the origin of the apparent synchronicities among marine community shifts have been debated (Conversi et al., 2010a, Beaugrand 2015). Möllmann and Diekmann, (2012) suggested that multiple drivers, such as climate and overfishing, may interact in triggering ecosystem community shifts between alternative states. Reid and Beaugrand (2012) observed that, in many cases, the reported shifts coincided with major temporal changes seen in marine temperature anomalies. The interaction between climate-induced environmental changes and species' ecological niches (Beaugrand 2015; Beaugrand et al., 2019) may lead to a community shift. For such shifts, the existence of tipping is not needed as an explanation.

Another region of potential climate change-induced regime shifts is the Arctic Ocean. As summer sea ice declines, spring phytoplankton blooms are becoming possible, leading to Arctic ecosystems becoming more like the present North Atlantic and productivity increasing by 30-50 per cent (Yool et al., 2015). Warming and circulation changes can also lead to the spread of invasive species – for example in the Barents Sea and from the Pacific (Kelly et al., 2020; Neukermans et al., 2018; Oziel et al., 2020) (see 1.4.2.1). However, while these changes may trigger regime shifts, it is currently difficult to predict whether they will feature self-sustaining tipping dynamics.

Empirical thresholds for marine communities have been estimated in specific cases using ecosystem model-derived indicators of community status (e.g. Samhouri et al., 2010), but are in general challenging to identify. Evidence for irreversibility is anecdotal and case-specific. One example is shifts in the anchovy-sardine cycles (Schwartzlose et al., 1999) that occur worldwide. Such shifts appear to be triggered by changes in short and long-term climate conditions. In the Peruvian upwelling system, switches in climate cycles can thus correspond to tipping points for the community (Alheit and Niquen 2004; Chavez et al.,2003), with effects on the middle (decadal) to long (centuries) timescale (Salvatteci et al., 2018). Evidence for this system suggests that natural fluctuations and anthropogenic climate change may pose an increased risk of tipping toward irreversible changes to a community characterised by less desirable (from a social-ecological perspective) and less productive features (Salvatteci et al.,2022).

Kelp forests

Kelp forests are mostly coastal ecosystems dominated by dense populations of large brown macroalgae (Figure 1.3.27). In recent decades, a significant number of these forests have undergone devastating collapses, resulting in their transformation into desolate and unproductive communities, called barrens. These collapses are primarily driven by overgrazing by sea urchins (Ling et al., 2015). However, additional pressures, such as marine heatwaves (McPherson et al., 2021), nutrient concentration (Boada et al., 2017) and sedimentation (Foster and Schiel, 2010), also contribute to its formation.

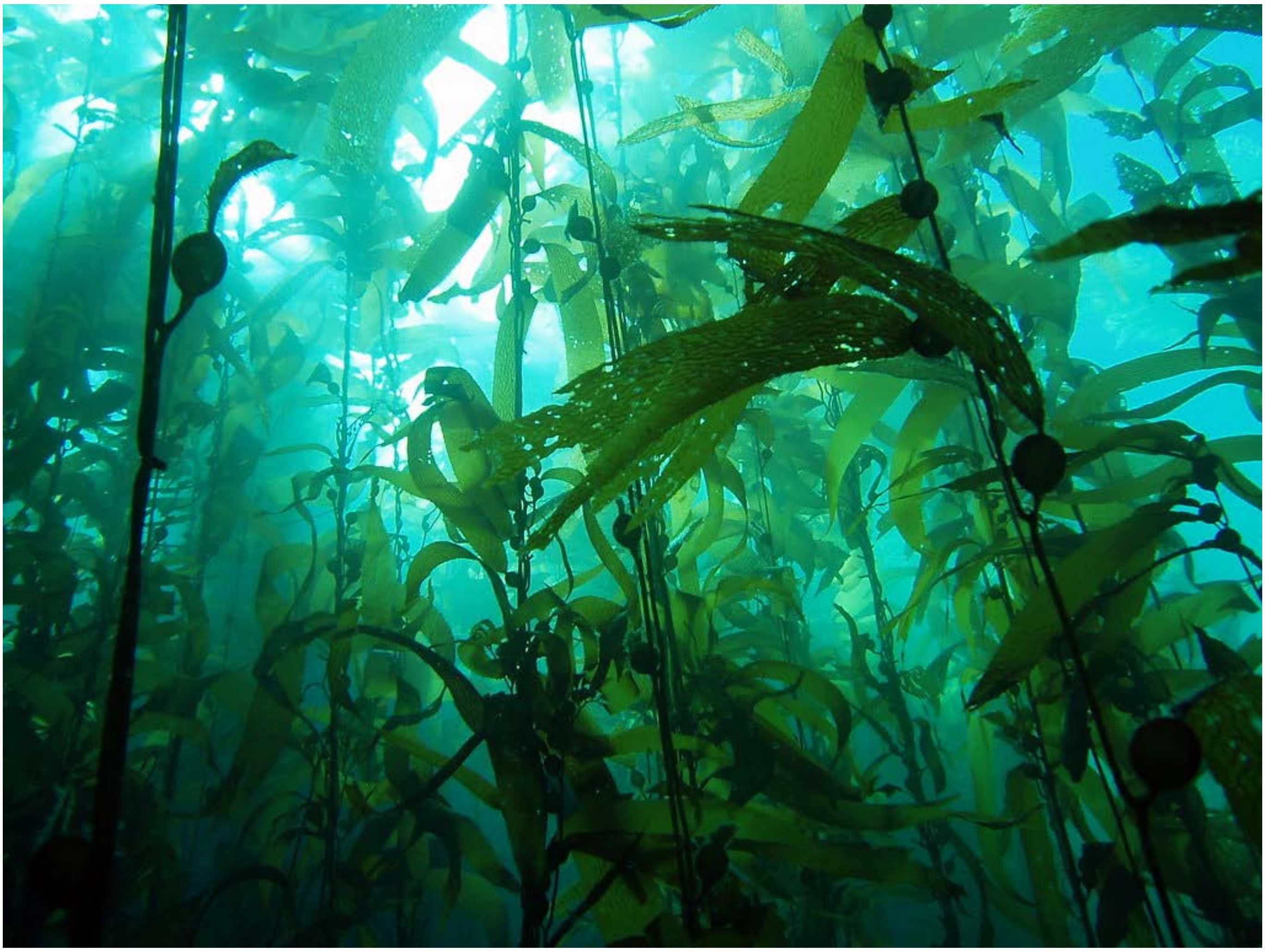

Figure 1.3.27: Kelp forest at Anacapa Island, California, 2010. Source: Dana Roeber Murray, flickr

Persistent, catastrophic regime shifts in coastal rocky communities transitioning between productive macroalgal beds and impoverished sea urchin barrens have been shown to occur worldwide (Ling et al., 2015). In many cases, such regime shifts exhibit nonlinear dynamics with hysteresis, where the transition shifts exhibit tipping points (Filbee-Dexter and Scheibling, 2018). Thresholds can be estimated empirically through a critical density of sea urchins (Ling et al., 2015), but such thresholds are influenced by biotic and abiotic factors.

Two feedbacks promote the stability of the barren state: processes that reduce kelp recruitment on barrens and processes that allow sea urchins to maintain high densities on barrens (Filbee-Dexter and Scheibling, 2018). For example, adult sea urchins seem to provide shelter and facilitate survival of urchin recruitment, offering a reinforcing mechanism. Similarly, barren conditions are kept open by intense grazing, reducing the chances of kelp recruitment.

Empirical studies have demonstrated the possibility of kelp forest recovery once sea urchin densities are limited (Smith and Tinker, 2022; Galloway et al., 2023). However, such recovery is influenced by abiotic factors such as marine heat waves, making kelp forest reversibility uncertain.

Biological carbon pump

The biological carbon pump (BCP) refers to the suite of processes that remove ~50 Gt of carbon annually from the atmosphere and into marine biomass, transferring ~10 per cent of this into the deep ocean (Carr et al., 2006; Westberry et al., 2008; Fu et al., 2016). Without this flux, atmospheric CO_2 would likely be ~200 ppm higher than the present-day concentration (Henson et al., 2022).

The largest component of the BCP, the gravitational pump, is driven by sinking of organic matter, mostly from dead plankton and detritus such as faecal pellets (Figure 1.3.28) (Nowicki et al., 2022). This part of the BCP is expected to decline with warming as a result of reduced mixing between warming surface and colder deep waters (thermal stratification) leading to reduced nutrient supplies for surface algae (i.e. phytoplankton), as well as warming favouring smaller plankton species that contribute less sinking matter (Armstrong McKay et al., 2021). However, there is no known mechanism that would enable this decline to become self-sustaining, with changes scaling quasi-linearly with emissions in models, and it is therefore not considered to show tipping-point behaviour (Armstrong McKay et al., 2022).

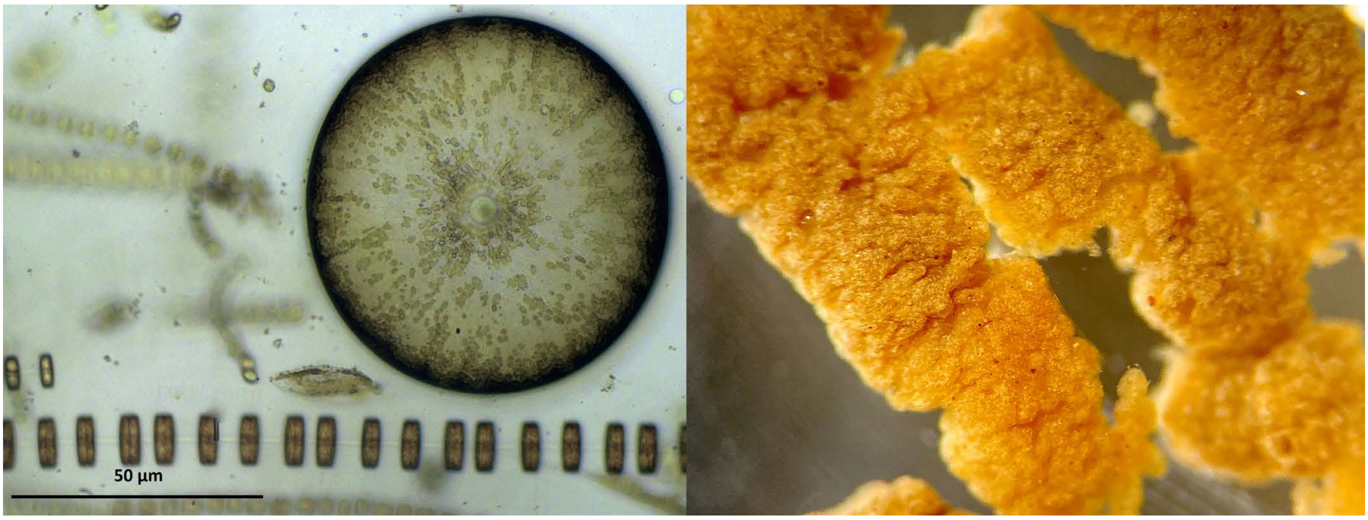

Figure 1.3.28: Left: the centric diatom *Coscinodiscus* sp. which is a large, lipid and carbohydrate-rich species that capitalises on peak nutrients during early spring. Image courtesy of Amanda Burson (British Antarctic Survey). Right: organic detritus produced by jellyfish from the subtropical South Atlantic, March 2023. Approximate width of pellets is 1.5mm. Image: Daniel Mayor on Instagram (accessed 2023).

A system that is more likely to show tipping-point behaviour is the seasonal lipid (fat) pump (SLP) (Jonasdottir et al., 2015). The SLP mainly occurs in high latitude oceans and is driven by the seasonal vertical migration of lipid-rich zooplankton (Figure 1.3.29) into the deep ocean, where they overwinter for ≥6 months, directly injecting carbon below the winter mixed layer.

A dramatic reduction in primary production via diatoms, for example, driven by changing nutrient supply patterns via increased stratification due to ocean warming, could result in zooplankton not consuming enough lipids to successfully overwinter and reproduce the following spring. Arresting the SLP would irreversibly change the ecological and biogeochemical functioning of high latitude ecosystems.

Figure 1.3.29: The marine copepod, *Calanus finmarchicus*, with its lipid sac outlined in red. Reproduced from (Mayor et al., (2020) and (Anderson et al., (2022).

Deep ocean warming will increase rates of respiration, meaning that lipid reserves may become exhausted before returning to the surface. This will interrupt recruitment and halt the SLP. The poleward migration of non-diapausing species (i.e. those that do not form an inactive life-form for parts of the year), as polar conditions ameliorate, could eventually mean that lipid-storing deep-diapausing zooplankton eventually disappear and the SLP collapse will be irreversible. However, the SLP was only described <10 years ago, and so our nascent understanding of its scale and complexity currently precludes the establishment of thresholds.

Other parts of the ocean biological pump could also result in nonlinear dynamics or tipping points. A recent paper found evidence that 'mixotrophs' – plankton that can both photosynthesise like algae and consume other plankton – can switch between a photosynthesis-dominant carbon sink state to a consumption-dominant carbon source state, with warming pushing them towards the latter and nutrient pollution making tipping dynamics more likely (Wieczynski et al., 2023). Mixotrophs are common in the ocean but their role in ocean and ecosystems and the biological pump is under-studied (Ward, 2019), making the impacts of these potential tipping dynamics unclear.

Marine oxygenation

Coastal hypoxia is a regime shift that occurs when dissolved oxygen in water diminishes below levels detrimental to marine life. As a consequence, one of their symptoms are 'dead zones', areas of the oceans where fish and many other marine organisms (particularly in benthic communities) migrate outwards or die due to low oxygen levels.

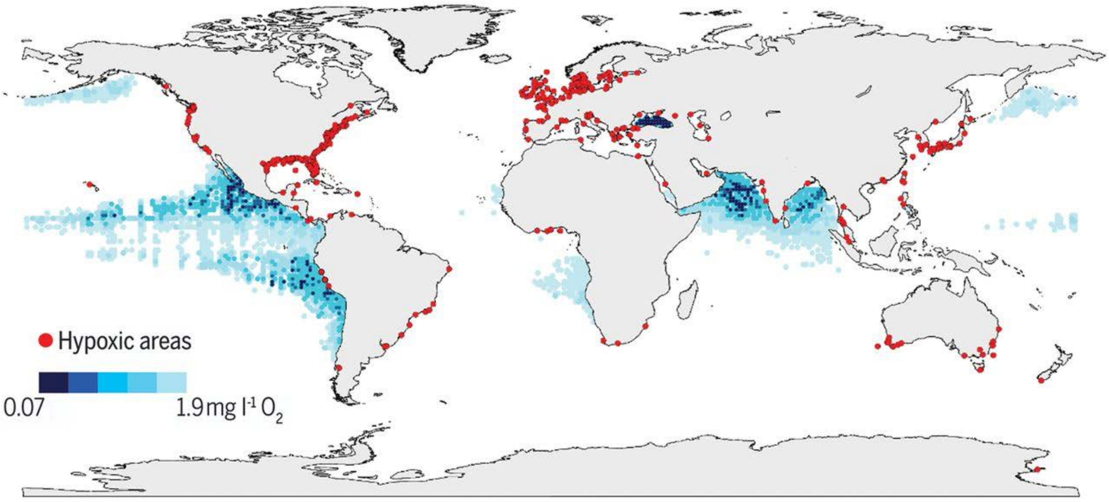

Figure 1.3.30: Map of known oceanic oxygen minimum zones (at 300m depth, blue) and coastal sites where anthropogenic nutrients have exacerbated or caused O_2 to decline to <2 mg/litre (red dots), becoming 'dead zones'. Source: (Breitburg et al., 2018).

While hypoxia is naturally occurring in some areas, hypoxic events have been increasing over the last few decades. A first global assessment of ocean deoxygenation documented over 300 cases mainly in the Atlantic coast of North America, the Caribbean, Mediterranean and Baltic seas (Diaz and Rosenberg, 2008). Subsequent assessments expanded to >500 case studies, from occasional hypoxic events to severe anoxia (Breitburg et al., 2018) (Figure 1.3.30).

The main mechanisms underlying coastal hypoxia are related to over-enrichment of nutrients like phosphorus and nitrogen coming from agricultural fertilisers, sewage or upwelling currents in the ocean. The latter are natural currents that bring nutrient-rich waters from the deep ocean to the surface, powering the primary producers (i.e. algae) and in turn productive food webs. In high-nutrient waters, algae can become over-abundant, consuming the available oxygen and causing the death of fish or other oxygen-dependent organisms. As they die, decomposers then further decrease available oxygen as they break down extra organic matter. Additional nutrients from fertiliser and sewage runoff on land is amplifying this process, increasing the number of hypoxic events and sites (Breitburg et al., 2018; Heinze et al., 2020).

At the same time, phosphorus can be released from sediment under low oxygen conditions, acting as a positive/amplifying feedback by further amplifying the growth of algae and the consumption of oxygen (Conley et al., 2002; Adhikari et al., 2015). Besides nutrients, climate change can exacerbate hypoxia by reducing oxygen solubility in water (Breitburg et al., 2018), and is projected to cause widespread deoxygenation over coming centuries to millennia via warming and enhanced land weathering delivering more phosphorus (Watson et al.,2017; Battaglia and Joos, 2018). Even if warming peaks and falls ('overshoot') Earth system models indicate that deoxygenation in the upper 1000 metres of the ocean is irreversible for multiple centuries (Santana-Falcon et al., 2023). Sea surface temperature can also change the strength of upwellings and thus the inflow of nutrients in coastal ecosystems.

Marine ecosystems with dissolved oxygen higher than >2 mL per litre sustain diverse ecological communities, and this level is considered normal (also known as 'normoxia'). Below this level the symptoms of hypoxia appear, including hypoxic events and dead zones. Anoxia occurs when levels of dissolved oxygen are below 0.5 mL per litre, which only a few microbial species are able to survive (Diaz and Rosenberg, 2008). Some dead zones and hypoxic events are reversible in scale of months to years. However, more and more areas are reported as chronically hypoxic, possibly irreversible in the timescale of ecosystem managers (centuries). Examples of severe hypoxia are dead zones in the Gulf of Mexico, central Baltic, Kattegat, Black Sea, and East China Sea (Breitburg et al., 2018).

Assessment and knowledge gaps

Table 1.3.3: Summary table of marine environment tipping points considered in this section.

System	Tipping system?	Timescale	Biophysical Impacts	Confidence	Gaps
Fisheries **[Small, fast-growing fish]**	No	Decades	Changes in entire trophic assemblage. A regime shift in one species could propagate the regime shift in many components of the ecosystem. Important especially in bottom-up and wasp-waist ecosystems.	Low confidence because too many different species	Need more coherent statistical approaches to identify tipping points and the presence of hysteresis. Also need more analyses on single species that look at tipping points in fisheries.
Fisheries **[Large, slow-growing fish]**	Depends on the stock	Decades	Changes in entire trophic assemblage. A regime shift in one species could propagate the regime shift in many components of the ecosystem. Especially important in top-down ecosystems.	Low confidence because too many different species and many different areas	Need more coherent statistical approaches to identify the tipping points and the presence of hysteresis. Also need more analyses on single species that look at tipping points in fisheries.
Fisheries [Cod]	Yes (in 16 out of 19 stocks)	Decades	Changes in the entire trophic assemblage, trophic cascade.	High confidence	In some cases there is the need to better understand feedbacks of hysteresis.
Community shifts	Yes	Decades	Changes in ecosystem function. structure and feedbacks that may affect how to best manage the system.	Low confidence - complexity from many different species and interacting drivers	Better understanding required on interplay of multiple drivers and species interactions. Tipping points difficult to identify and predict.
Kelp forests	Yes	Months to decades	Changes to community composition of fish and macroinvertebrates scaling up to trophic disassembly.	High confidence	Necessary to understand how key ecosystem properties, e.g. resilience or stability of kelp forests, evolve over the years.
Ocean hypoxia	Yes	Months/years to centuries. Reversible at surface, irreversible at depth for centuries to millennia	Major changes in ocean productivity, biodiversity and biogeochemical cycles.	Low confidence	Degree of self-sustaining change and hysteresis; influence of future climate change and nutrient use.

Table 1.3.3 summarises our assessment of tipping dynamics (with confidence levels) along with biophysical impacts, timescales and knowledge gaps for marine ecosystems. We have high confidence that cod fisheries and kelp forests can pass tipping points, low confidence that some other large-fish fisheries, marine communities and potentially the lipid pump could also tip, and medium confidence that marine hypoxia could feature tipping dynamics. Knowledge gaps include limited understanding of complex species and driver interactions, limited ability to detect and project marine tipping points in practice, and how ecosystem resilience can change over time.

1.3.3 Final remarks

In this chapter we have assessed evidence for tipping dynamics across the biosphere, finding that many ecosystem tipping points are possible. Compared to tipping points in the cryosphere (Chapter 1.2) and ocean/atmosphere circulations (Chapter 1.4), biosphere tipping points tend to feature more co-drivers, including habitat degradation and loss, direct exploitation and nutrient pollution with often complex interactions (IPBES, 2019). Along with strong spatial variability, this often makes ecosystem tipping thresholds and risks more difficult to assess. However, these complexities also provide opportunities for action to avert tipping.

While climate change is a common leading driver, requiring urgent global emissions phaseout, compared to the cryosphere or ocean circulation it is more possible to directly increase the resilience of some at-risk systems. Actions such as ecological restoration and inclusive conservation, adaptive management and improved governance can help protect biodiversity and bio-abundance and so help to maintain key stabilising feedbacks that can help counter tipping (see Chapter 3.2). Such restoration and regenerative land use practices would also help to draw down some carbon from the atmosphere, helping to slow climate change (Girardin et al., 2021; Rockström et al., 2021). Such 'nature-based solutions' would not be enough to stop climate change though, which can only be achieved with a rapid cessation of greenhouse gas emissions.

Most ecosystems considered in this chapter can also be considered social-ecological systems, with people living within, and being integral to, the dynamics of these systems (Folke et al., 2016, 2021). While in some heavily degraded ecosystems restoration might entail minimising human impacts, in most places actions like supporting sustainable livelihoods for local communities can better help promote both ecological restoration and support human wellbeing in a way that makes both more sustainable in the long term (IPBES, 2019). The rights of Indigenous peoples – whose territories cover more biodiverse area globally than officially protected areas (ICCA Consortium, 2021) – must be respected, and their knowledge recognised as critical. Many other societal shifts are also necessary to underpin ecological restoration, including transformative changes to the global food system and commodity consumption (which together are key drivers behind much habitat loss and pollution – IPBES, 2019).

From a research perspective, we have identified several critical areas where improved knowledge could help us better understand biosphere tipping dynamics. In particular, deep uncertainties exist around the relative strength of feedbacks controlling ecosystem tipping dynamics, such as the complex interactions between ecohydrological and fire feedbacks in forest, savanna and dryland biomes. The role of increasing extreme event frequency and intensity in reducing and overcoming ecological resilience is also critical for ecosystems such as coral reefs, mangroves and forests, but it is not well resolved in models. Plant adaptability and spatial variability are also not well represented in models, despite being key factors adding complexity to ecosystem tipping dynamics. More observations, experiments and improved models, and integrations across these, are all required to address these issues.

Observations from field and remote sensing can also help monitor and detect declining ecosystem resilience, as well as potential early warning signals (see Chapter 1.6). Greater data sharing and international collaboration would improve both monitoring and understanding. Lastly, co-designing research with researchers from across the natural and social sciences, Global South and North, and from multiple knowledge systems including Indigenous and traditional ecological knowledge is critical for fully understanding ecological dynamics and the potential for tipping.