



Freshwater Resources and Biodiversity: From Competition to Synergy



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Freshwater resources and biodiversity: from competition to synergy

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significance

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1 Introduction

Klement Tockner

Disturbing news about the state of the world's water resources appears in the headlines almost daily. The summer of 2026 provided a stark illustration. Prolonged drought and recurrent heatwaves affected large parts of Europe, with rivers like the Loire, Po, Rhine and Danube reaching record-low water levels in some sections. Navigation and industrial supply chains were disrupted, water-use restrictions were imposed across affected regions, and exceptionally low flows in the Danube forced the successive shut-down of both reactors at Romania's Cernavodă nuclear power plant. At the same time, high water temperatures and low flows placed freshwater ecosystems under severe stress, contributing to declining oxygen concentrations, algal blooms, deteriorating water quality and fish mortality. The events demonstrated how rapidly water scarcity can cascade across ecosystems, economies and societies (JRC, 2026; Toreti, 2026).

Such events are manifestations of a much broader long-term trajectory. Extreme dry periods are contributing to exceptionally low water levels in rivers and lakes, declining groundwater levels, and water shortages for domestic supply, agriculture, industry, and power generation. At the same time, a warmer atmosphere can hold more water vapour, intensifying heavy precipitation, while accelerated snow and glacier melt alters the timing and magnitude of runoff.

Together with land-use change and other human modifications of the water cycle, these processes lead to an increase in hydrological extremes in many regions, although with substantial regional variation. Flash floods threaten people and infrastructure, mobilise pollutants and pathogens, and can overwhelm water and sanitation systems. Droughts and heatwaves, by contrast, reduce water availability and habitat connectivity and increase thermal and oxygen

stress in freshwater ecosystems. These interacting pressures erode freshwater biodiversity and impair ecosystem processes that underpin essential benefits to people.

Water is among humanity's most essential natural assets and common goods, but it is far more than a resource to be allocated among competing users. It is a fundamental and irreplaceable basis of all life on Earth and a key component of the Earth system. Through its close interconnections with climate, the biosphere, soils and biogeochemical cycles, the water cycle plays a critical role in regulating the Earth's system function. Human modification of the water cycle is therefore increasingly recognised as a matter of concern for the Earth's system as a whole. Ultimately, maintaining the integrity of the water cycle is essential for sustaining the conditions that make large parts of our planet habitable for humans and other forms of life.

1.1 Freshwater resources

Historically, the spread of humans across the globe was primarily driven by climate and access to freshwater. In fact, our ancestors lived close to forests and trees for shelter and along rivers, lakes and seashores for resources (e. g., Finlayson, 2014). Human civilisations evolved along large rivers such as the Nile, Euphrates, Indus, Mississippi and Huang He and around lakes such as Lake Victoria. Today, about 50 % of the global human population lives within 3 km to a surface water body, and less than 10 % live farther than 10 km away (Kummu et al., 2011).

Freshwater on Earth is highly unevenly distributed, with different water stores differing enormously in size, accessibility and turnover time. The largest stores are ice sheets and glaciers (~25.8 million km³) and groundwater within the upper ~2 km of the continental crust (~22.6 million km³, including saline groundwater), much of which is non-renewable on human timescales or otherwise not readily accessible. Other important stores include permafrost and ground ice (~207,000 km³), freshwater lakes (~108,000 km³), soil moisture (~54,000 km³), wetlands (~14,000 km³), reservoirs (~11,000 km³), seasonal snow (~2,900 km³), rivers (~1,900 km³) and living organisms (~1,000 km³). By comparison, the

atmosphere contains ~13,000 km³ of water at any given time (Abbott et al., 2019).

Despite their limited volumes, dynamic freshwater stores and fluxes play a disproportionate role in sustaining the global water cycle and terrestrial and freshwater ecosystems. Approximately 110,000 km³ of precipitation falls on land each year, around 70,000 km³ of which returns to the atmosphere through evapotranspiration, while roughly 40,000 km³ becomes runoff and ultimately contributes to ocean discharge through surface and groundwater flow. Freshwater availability therefore depends not simply on the total volume of water stored on land, but critically on the rates of replenishment, depletion and exchange among water stores. Characteristic turnover times span many orders of magnitude, from days to weeks for atmospheric water and weeks to months for rivers, and from decades to centuries or millennia for groundwater, glaciers and ice sheets (Abbott et al., 2019; Gleeson et al., 2016).

Since the mid-20th century, commonly referred to as the start of the “Great Acceleration”, humans have profoundly altered water stores and fluxes. Beyond localised water abstractions,

inter-basin water transfers and the hydrological repercussions of climate change, society has created new anthropogenic pathways in the water cycle. In particular, large volumes of virtual water are transported across the globe through global trade, establishing a complex, consumption-driven network of anthropogenic water flows.

1.2 Freshwater ecosystems

Freshwater ecosystems comprise rivers and streams, lakes and ponds, wetlands, and groundwater of natural or artificial origin (Fig. 1.1). The water they contain may be fresh, brackish, or saline. Embedded within terrestrial landscapes as dots or linear features, and generally occupying their lowest-lying areas, freshwater ecosystems collect, transport, store, and transform not only water but also sediments, nutrients, organic matter, and pollutants, thereby connecting terrestrial, marine, and atmospheric realms. Their water flows and storage, spatial extent and depth, as well as their connectivity and consequently functioning, vary over time in response to climatic dynamics and human impacts. (Throughout this report, we use the terms *freshwater resources* and *freshwater ecosystems* for simplicity and consistency, although *continental aquatic ecosystems* (or *inland waters*) would be more accurate, as our assessment also encompasses brackish and non-marine saline systems.)

Rivers and streams form highly connected networks that transport water, sediments, nutrients, pollutants and organisms from headwaters to floodplains, lakes, and oceans. Rivers and streams occupy only a small fraction of the

Earth's surface – an estimated 773,000 km² at mean annual streamflow, equivalent to only 0.58 % of the non-glaciated land surface (Allen & Pavelsky, 2018). Despite their limited spatial extent, flowing waters are ecologically and biogeochemically important at the global scale (Battin et al., 2023; Bernhardt et al., 2022). Streamflow, together with stream water quality, forms the integrated response to all processes, natural or anthropogenic, occurring in the upper drainage basin.

Lakes represent considerably larger and more persistent freshwater stores. Messenger et al. (2016) identified 1.43 million lakes and reservoirs >0.1 km², covering 2.67 million km² worldwide. Including smaller water bodies, an estimated 117 million lakes and ponds ≥0.002 km² cover about 5 million km², or 3.7 % of the non-glaciated land surface (Verpoorter et al., 2014). At the other end of the size spectrum, a very small number of large lakes account for a disproportionate share of global lake-water storage, with Lake Baikal in Siberia alone accounting for 20 % of the total.

Wetlands form the transition between terrestrial and aquatic ecosystems and include marshes,

peatlands and floodplains. Estimates of their global extent vary with definitions and mapping approaches, but recent assessments place total wetland area at approximately 12–16 million km² (Davidson et al., 2023), equivalent to 8–11 % of the global land area. Wetlands regulate water flows, retain sediments and nutrients, store carbon and provide important habitat for biodiversity. Human transformation has been extensive: an estimated 3.4 million km² of inland wetlands, or about 21 % of their original global area, have been lost since 1700, predominantly through conversion to agriculture (Fluet-Chouinard, 2023).

Groundwater is not merely a water resource but constitutes the most extensive freshwater habitat. Groundwater harbours diverse microbial communities as well as a wealth of specialised invertebrate fauna, and even vertebrates such as fish and amphibians. Groundwater contributes to water purification, including the provision of water for human consumption, nutrient and carbon cycling and the maintenance of surface-water ecosystems such as springs, streams, and wetlands (Griebler et al., 2014; Koch et al., 2024; Mammola et al., 2026). Furthermore, terrestrial vegetation heavily depends on groundwater. A

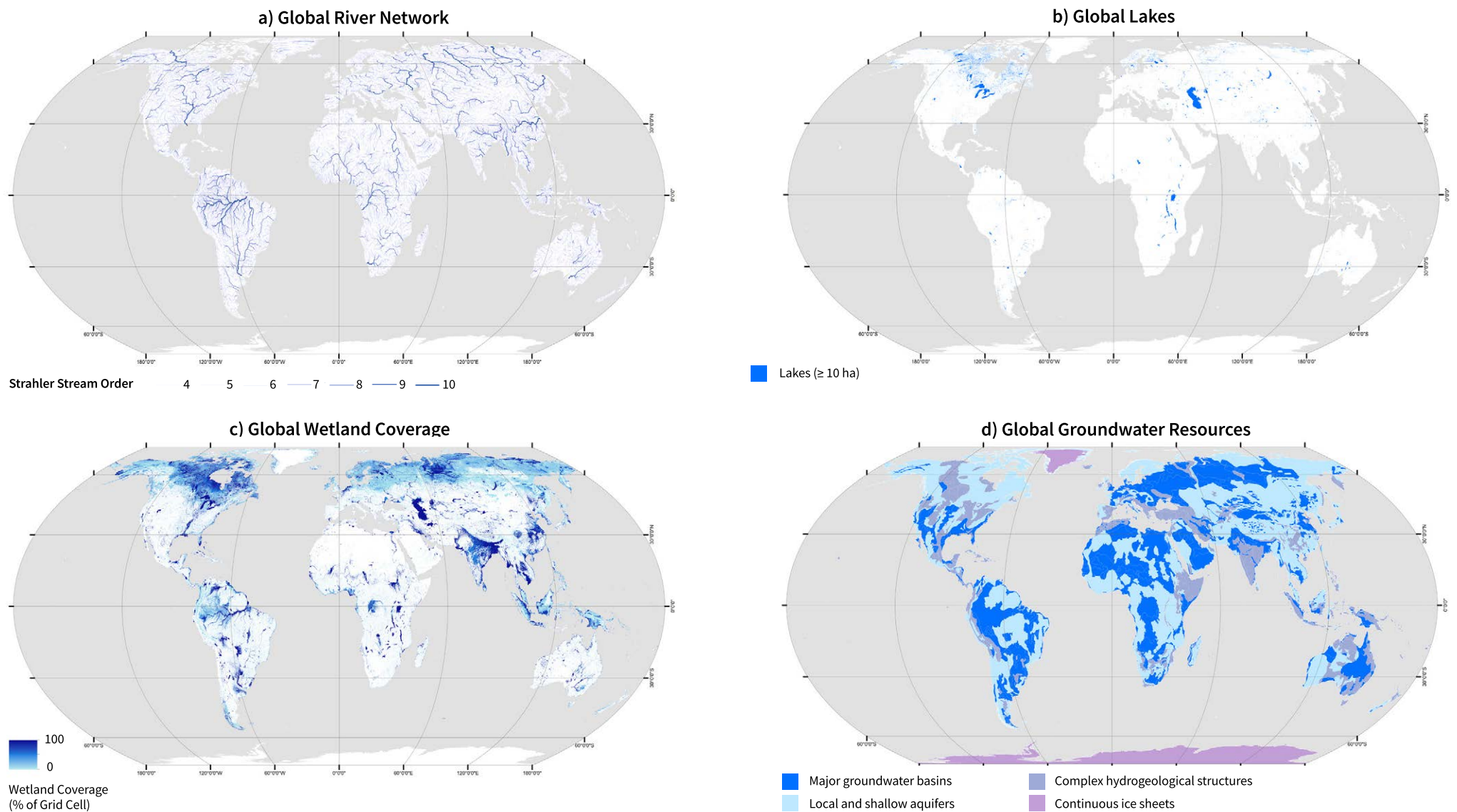


Fig. 1.1 Global spatial distribution of major freshwater ecosystems: (a) rivers, (b) lakes, (c) wetlands, and (d) potential groundwater stores. The maps highlight broad geographic patterns in the distribution of surface and subsurface freshwaters. References: a) Lehner & Grill, 2013; data available at <https://www.hydrosheds.org>; b) WHYMAP, 2010; c) Messenger et al., 2016; d) Lehner et al., 2025

recent global assessment identified potential aquatic and terrestrial groundwater-dependent ecosystems across 8.3 million km² of drylands, 53 % of which occur in regions experiencing declining groundwater storage (Rohde et al., 2024).

Together, these ecosystems form an expansive network in which surface and groundwater, freshwater and terrestrial habitats, and up-stream and downstream processes are inter-linked. Changes in one component can therefore propagate throughout the wider freshwater system.

1.3 Water as a threat

Water is essential for life. Yet too much, too little, or polluted water poses major risks to people, economies and ecosystems. Climate change, together with direct human modification of the water cycle, is intensifying these risks in many regions. Human-induced climate change has already altered the global water cycle, increased the intensity of heavy precipitation and exacerbated drought conditions in many regions, while land-use change, water abstraction and other human interventions further modify water flows and storage (IPCC, 2021, 2022a).

Between 1970 and 2021, 11,778 reported weather-, climate- and water-related disasters caused just over two million deaths and US\$ 4.3 trillion in reported economic losses worldwide. More than 90 % of reported deaths occurred in developing countries (WMO, 2023). Floods are among the most widespread and damaging water-related hazards. Droughts typically develop more slowly but can persist for months or years and affect very large areas, with cascading repercussions for water supply, agriculture, food security, energy production and ecosystems. Recent estimates put the global economic costs of drought at approximately US\$ 307 billion per year, although the full costs remain

difficult to quantify because many impacts are indirect, cumulative and long-term (Thomas et al., 2024; UNDRR, 2025).

Water-related diseases add substantially to these risks. Inadequate drinking water, sanitation and hygiene (WASH) were responsible for an estimated 1.4 million preventable deaths in 2019 (WHO, 2023; Wolf et al., 2023). Vector-borne diseases are also closely linked to water and climate. In 2023, malaria alone caused an estimated 263 million cases and 597,000 deaths worldwide (WHO, 2024). Moreover, dengue, chikungunya and other mosquito-borne diseases are expanding geographically as climate change alters the suitability and seasonality of vector habitats, while global trade, travel, urbanisation and population movements further facilitate their spread (Colón-González et al., 2021; WHO, 2024).

These risks are strongly interconnected. Drought reduces water availability and can deteriorate water quality, including by concentrating pollutants, while floods can contaminate drinking-water supplies, mobilise pollutants and pathogens, and overwhelm sanitation systems. For example, during the severe drought in Somalia in

2022, 7,796 cholera cases were reported within a period of just over six months across 25 drought-affected districts, where access to safe water and sanitation was severely constrained. Degraded freshwater ecosystems further reduce natural water purification, retention and buffering capacities. Water risk is therefore not simply a matter of too much or too little water, but also concerns its quality, timing and accessibility – and of the resilience of the ecosystems and societies that depend on it (see also [Box 1](#)).

1.4 Integrating freshwater resources, biodiversity and ecosystem management

Freshwater resources, biodiversity and ecosystems have traditionally been addressed through largely separate domains of science, engineering, policy and public administration. Water of suitable quality is primarily regarded as an essential and irreplaceable resource whose availability must be secured for domestic supply, agriculture, energy generation, transport and industry. Freshwater ecosystems, by contrast, are primarily considered through environmental policies aimed at conserving and restoring biodiversity, habitats and ecological processes. This sectoral separation fails to adequately recognise the fundamental interdependencies between human water security and ecosystem integrity and constrains opportunities for integrated solutions that reconcile competing demands and generate benefits for both people and nature (CBD, 2020; UNEP, 2024b).

This separation, reinforced by fragmented institutional responsibilities, is particularly problematic because freshwaters are both exceptionally rich in biodiversity and among the most threatened ecosystems worldwide. Despite their limited spatial extent, rivers, lakes and wetlands support a disproportionately large share of global biodiversity (Reid et al., 2019).

Moreover, water abstraction, flow regulation, pollution, fragmentation, invasive species, land-use change and climate-related extremes are profoundly altering freshwater ecosystems. Monitored freshwater vertebrate populations have declined particularly fast compared with their terrestrial and marine counterparts (WWF, 2024). Therefore, effective water management needs to account simultaneously for human demands and the ecological processes, connectivity and biodiversity required to sustain intact ecosystems (Haubrock et al., 2025; Tickner et al., 2020).

An important step is to recognise that biodiversity loss and ecosystem degradation can undermine the multiple benefits that intact freshwater ecosystems provide to societies and economies. Ecosystem services are the benefits that people and economies derive from functioning ecosystems, including water supply and purification, flood and drought regulation, carbon storage, food production, habitat provision and cultural benefits. The economic value of freshwater ecosystems can be exceptionally high. For example, global ecosystem-service valuations estimate swamps and floodplains at around US\$ 26,000 per hectare per year, compared with

approximately US\$ 3,000–4,000 per hectare per year for forests, reflecting the multiple benefits of flood regulation, water storage and purification, nutrient retention, food production and habitat provision, alongside recreation and cultural values (Costanza et al., 2014; Griebler & Avramov, 2015; Haubrock et al., 2025).

The degradation of these functions can increase vulnerability to extreme events, compromise water quality and availability, and intensify competition over water resources, with consequences for human health and inequalities in access to safe and reliable water supplies. The resulting economic and social costs include production losses, higher expenditures for water treatment and flood protection, the technical substitution of lost ecosystem functions, and costly ecosystem restoration. Consequently, economic gains from intensive water use that disregard the requirements of functioning and multifunctional ecosystems can externalise substantial costs to other sectors, regions and future generations.

The central lesson is that safeguarding water quantity and quality, maintaining the spatial extent and connectivity of freshwater

ecosystems, and preserving ecologically appropriate flow regimes can simultaneously strengthen human water security and freshwater ecosystem integrity. Therefore, water security and ecosystem integrity should not be treated as competing objectives by default. The protection, sustainable management and restoration of freshwater ecosystems need to become integral components of long-term water security and sustainable water-resource management (CBD, 2020; Haubrock et al., 2025; UNEP, 2024b).

The present study analyses the interconnected pressures on freshwater resources, ecosystems and biodiversity and advocates integrated approaches that reduce conflicts and promote synergies in water management. [Chapter 2](#) and [Chapter 3](#) assess the current state and trends of freshwater resources and biodiversity. [Chapter 4](#) examines the interdependencies between water-resource use and freshwater ecosystem integrity, followed by an economic assessment of water-resource and biodiversity challenges in [Chapter 5](#). Finally, [Chapter 6](#) identifies nine priority areas for overcoming the limitations of sectoral management and advancing the integrated sustainable management of freshwater resources, biodiversity and ecosystems.

2 Water resources worldwide – unevenly distributed water stress, irreversible alterations, governance challenges, and rising conflicts

Petra Döll and Robert Luetkemeier

Human societies manage freshwaters in various ways, either to benefit from it or to reduce water-related hazards. This alteration of natural water flows and storage is exacerbated by anthropogenic climate change, causing water stress for both humans and nature (including biodiversity). Water stress for humans and freshwater biodiversity is also caused by anthropogenic degradation of water quality, while a large fraction of the global human population has no or insufficient access to safely managed water services. That said, water stress and its negative consequences are unevenly distributed across scales from the regional to the continental level; they are modulated by long-distance interconnections, so-called telecoupling, separating causes and effects across the globe (Liu et al., 2013).

Overall, humans have caused many irreversible changes to freshwater systems in terms of both

quantity and quality. Governance schemes that seek to protect and restore water resources and freshwater ecosystems to ensure a reliable stream of ecosystem services should thus strive to reduce water stress and to limit or reverse anthropogenic alterations, wherever possible.

2.1 Human appropriation drives the competition for water between society and nature

Within the Earth system, the hydrosphere interacts closely with the lithosphere, the atmosphere, and the biosphere. The global water cycle provides water resources for humans and biodiversity on all continents. Humans abstract so-called “blue” water, the liquid water available as surface water in rivers, lakes, wetlands, reservoirs and the groundwater, to supply water for domestic purposes, cooling of thermal power plants, manufacturing, mining, irrigation and livestock production (Fig. 2.1). Surface water is also used in-situ for transportation and hydroelectricity production. In addition, humans rely on so-called “green” water, the water in the soil derived from precipitation, for crop and livestock production as well as forestry (Falkenmark & Rockström, 2006). We hereafter focus on water flows and storage in surface water bodies and groundwater, which serve as resources for humans and, at the same time, as habitats for freshwater biodiversity. While both humans and biodiversity require a certain quantity of water and appropriate water quality to fulfil their needs, it is the society and the economy who decide how much water to appropriate for societal and economic purposes and how much to leave for biodiversity. Thus, competition for water is strongest between humans and

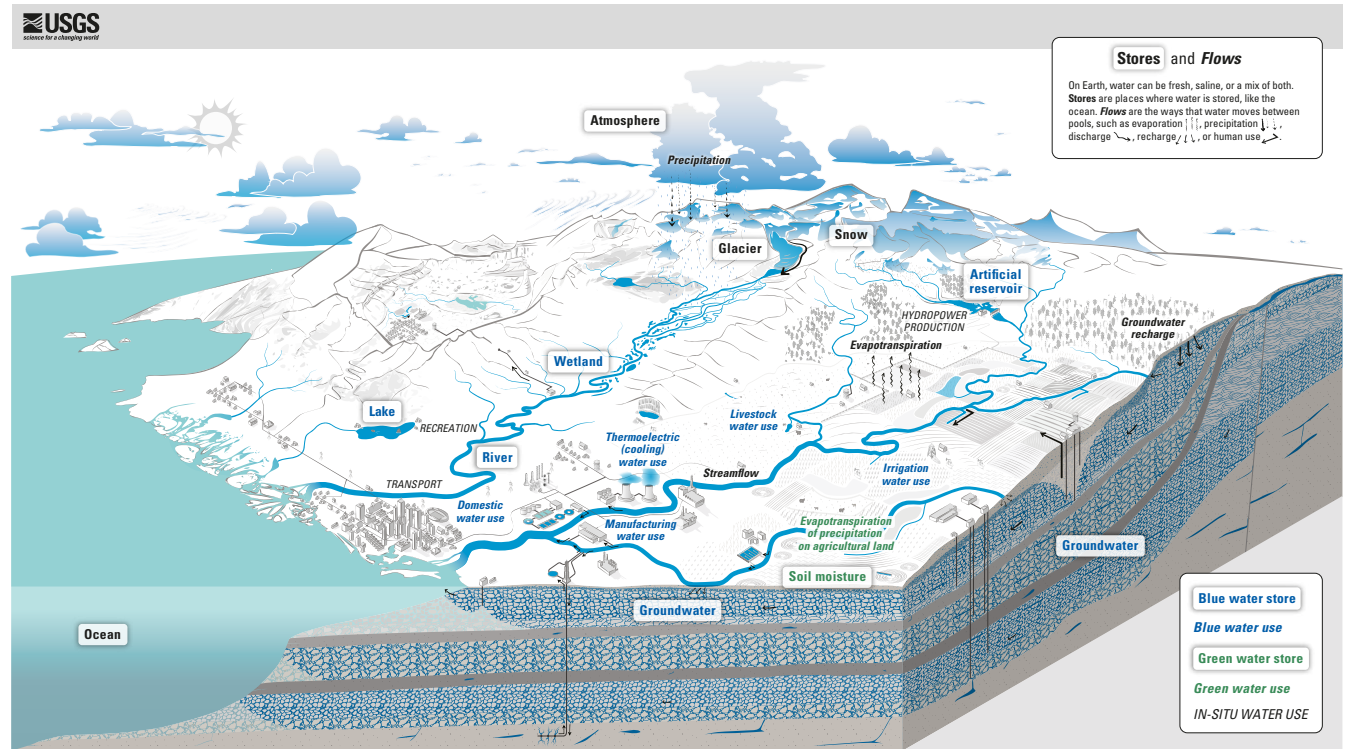
biodiversity, but also exists between water use sectors, e. g., between irrigation and industrial water use, or between upstream and downstream water users, in particular across country boundaries. Water quality of surface water and groundwater is directly affected by human water use through polluted return flows, in particular from domestic and industrial wastewater. More importantly, it is altered by a multitude of other human activities, such as industrial pollutant emissions into the atmosphere or the application of nutrients and pesticides in agriculture.

Globally, irrigation is the most important water use sector, with about 65–70 % of all water abstractions and 85–90 % of consumptive water use, i. e., the part of the abstracted water that evapotranspires (evaporates or transpires) during use (Müller Schmied et al., 2024). Different from other human water uses, the purpose of irrigation is to enhance plant growth; thus, the consumptive water use as a fraction of water abstractions is much higher in the irrigation sector than in other sectors. Consequently, only a relatively small fraction of the abstracted water remains in liquid form after human water use; the more technologically advanced the irrigation technique, the less water is returned

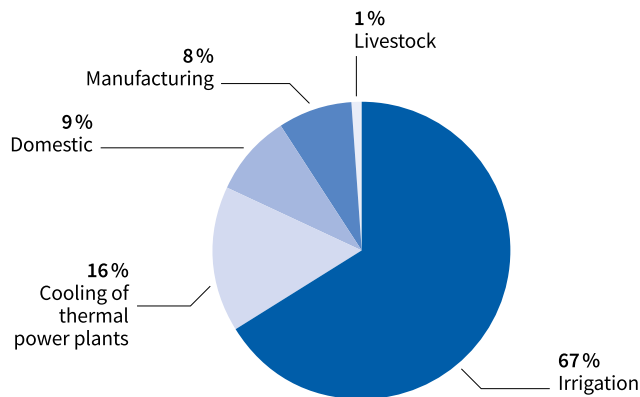
to the nearby environment in liquid form. In the case of all other sectoral uses, more than 80 % of the abstracted water remains liquid. Globally, irrigation is supported by more than 20,000 large artificial reservoirs behind dams globally (ICOLD, 2025), which cause pronounced changes in water flows and storages.

Water use for electricity production includes both the abstraction of cooling water for thermal power plants, which is the second largest water-use sector globally regarding water abstractions (Müller Schmied et al., 2024), and the in-situ water use for hydroelectricity production, which is dominantly related to more than 10,000 artificial reservoirs (ICOLD, 2025). The global values for water abstractions and consumptive water use for manufacturing and domestic purposes are rather similar, while livestock water use, i. e. the amount of water that livestock requires for drinking and washing, is important only locally (Fig. 2.1). Indirect livestock water use due to irrigation of livestock feed is much more significant. According to (Mekonnen & Hoekstra, 2020), crops produced for animal feed constitute 13 % of the global unsustainable blue water use in the sector, predominantly related to the production of

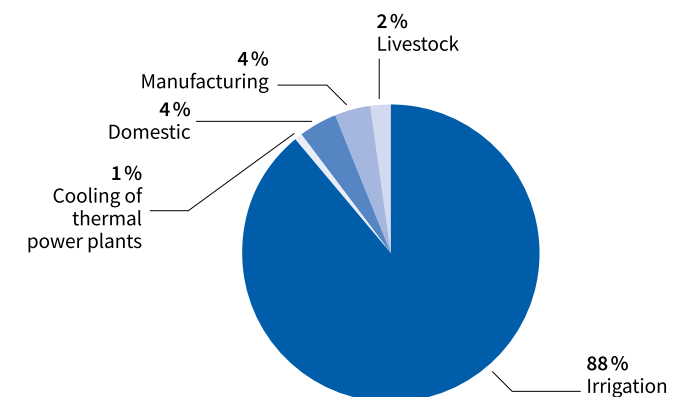
Fig. 2.1: The water cycle is impacted by human activities, together with sectoral shares of human water use. The water cycle is conceptualized as water flows between water stores in which the inflowing water is temporarily stored. Artificial reservoirs store streamflow, increase evaporation, and alter downstream streamflow dynamics. “Blue” water, i.e., the groundwater and the water in the surface water bodies, rivers, wetlands, lakes, and artificial reservoirs, is abstracted for irrigation, livestock, manufacturing, cooling of thermal power plants, and domestic purposes. Blue water use causes additional evapotranspiration, the so-called consumptive water use, a decline in groundwater levels and a decrease in surface water volume and flows as well as degradation of the water quality, altering habitat conditions in all connected blue water stores. Humans use “green” water when a part of the precipitation evapotranspires from the soil and enables crop growth. In-situ water use, including river transport, hydropower production and recreation, alters habitat conditions in surface water bodies. Figure modified from USGS Water Cycle Diagram (Corson-Dosch 2022). The pie charts show the sectoral shares of total human water abstractions and consumptive water use for the period 1991–2019 (Corson-Dosch 2022), in percent.



Global water abstraction
 $\approx 3800 \text{ km}^3/\text{year}$



Global consumptive water use
 $\approx 1300 \text{ km}^3/\text{year}$



maize, oats, wheat and fodder crops; the USA and China each account for 17 % of the global unsustainable blue water use related to animal feed, followed by Egypt (7 %), Pakistan (6 %) and Spain (5 %).

Biodiversity and ecosystem services. Alongside societal water demands, freshwater ecosystems require water to enable biodiversity to thrive and provide ecosystem services to society (see [Chapter 3](#) and [Chapter 4](#)). To estimate the degree to which streamflow can be altered by humans without excessively damaging the well-being of biodiversity, numerous approaches for quantifying so-called environmental flow requirements have been developed (Baghel et al., 2019). Limited ecological knowledge about the effects of streamflow alterations on biodiversity constrains the reliability of these approaches, which also entail human valuation of the admissible degree of alteration. Importantly, water stress does not only affect human societies but also biodiversity and ecosystem functioning. Reductions in water quantity, degradation of water quality, and limited environmental flows directly translate into altered ecological processes, and biodiversity loss. These ecological consequences are examined in detail in [Chapter 3](#) and [Chapter 4](#).

2.2 Water stress is distributed very unevenly around the globe

2.2.1 A broader definition: water stress encompasses quantity, quality and accessibility

The term “water stress” is not a strictly defined concept in the literature. Often, authors make use of volumetric quantification of water withdrawals over renewable water resources and consider a threshold of about 20 % as an adequate metric to differentiate between sustainable and unsustainable water abstraction; threshold exceedance would then indicate a water stress situation (Damkjaer & Taylor, 2017). However, this definition falls short in many ways e. g., Söller et al. (2024), so this study employs a broader definition. Following Schulte & Morrison (2014), we define water stress as the inability to meet the water demands of humans and ecosystems. Water stress encompasses three aspects: water quantity, water quality, and accessibility of water (i. e., whether people can make use of physically available water, which is a function of the sufficiency of infrastructure and the affordability of water, among other things) (Schulte & Morrison, 2014). The term “water scarcity” is best employed to exclusively denote the quantitative aspect of water stress, in the economic sense of relating the demand for water to its

supply. While floods are certainly water-related events that cause stress to humans, they are generally not included in the term ‘water stress’ (Schulte & Morrison, 2014).

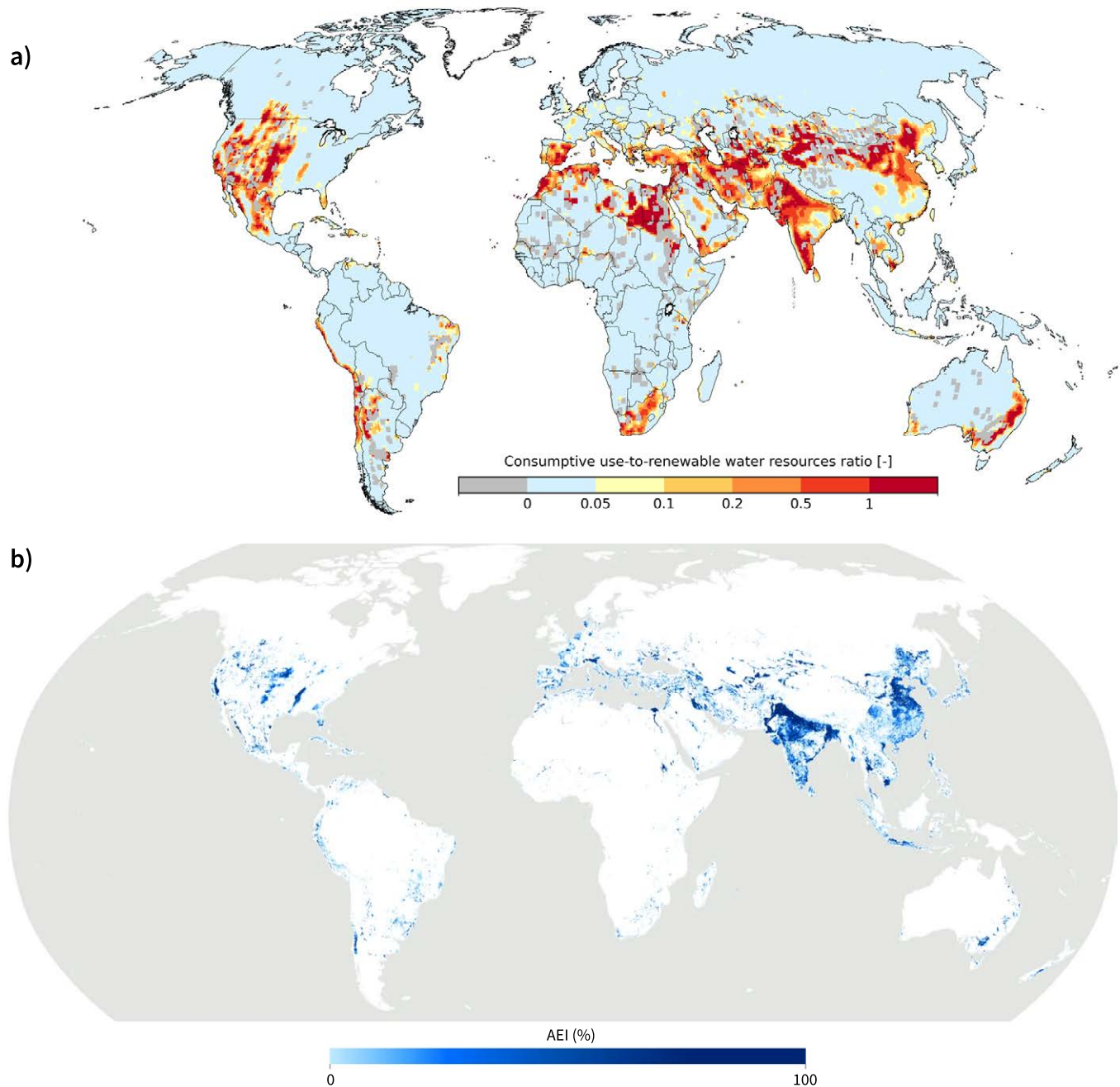
The three components of water stress are very unevenly distributed around the globe. Water scarcity, the water stress related to water volume, is usually quantified as an economic concept by a metric that relates the demand for water to its supply. Most often, the water abstraction-to-renewable water resources ratio is used, either aggregated over countries or river basins. However, these large spatial units implicitly assume that water can be evenly distributed over the whole spatial unit and therefore underestimate water scarcity in relatively dry sub-regions and possibly at locations away from the main rivers. In addition, water abstractions may include large return flows and thus, different from consumptive water use, do not represent the volumes of naturally occurring water that are lost by human water use.

2.2.2 Irrigated agriculture in dry regions is the primary driver of physical water scarcity

Fig. 2.2a provides a global map of physical water scarcity (WS_{phys}) at high spatial resolution of 0.5 x 0.5 arc-deg (about 55 x 55 km at the equator and 55 x 35 km at 50° latitude), where consumptive water use in the year 2015 is shown as a fraction of the renewable water resources (mean annual naturalised runoff in the period 1990–2019 (see **Text Supplement**). It is this fraction of liquid water that is lost due to human water use. This map visualises that physical water scarcity and the impact of human water use on mean annual water flows are distributed very heterogeneously around the globe. They are heavily concentrated in the dry regions of the globe with significant irrigation. Due to the predominance of irrigation in total consumptive water use, high values of physical water stress occur in regions with a high fraction of areas equipped for irrigation (**Fig. 2.2b**). Irrigated agriculture is the major driver of decreases in water flows and storage globally and more so in densely populated dry regions of the globe. Regions with the highest physical water scarcity are located in Mexico, the Western US,

Fig. 2.2: a) Physical water scarcity (WSphys) for humans and freshwater ecosystems as quantified by the consumptive use-to-renewable water resources ratio. Consumptive water use in 2015 and renewable water resources in 1990–2019 as computed by WaterGAP 2.3. Renewable water resources are defined as the long-term mean runoff, the difference between long-term mean precipitation and evapotranspiration. Values larger than 1 indicate 0.5 arc-deg grid cells where groundwater is depleted, and values less than 0 indicate that water inflow from upstream is the source of water use.

b) Global map of area equipped with irrigation (AEI) in 2015 (Mehta et al. 2024). Map of global AEI showing areas of high potential irrigation activity. Each 5 arc-min pixel shows the percentage of AEI. Only pixels having 1 % AEI or higher are shown.



Alberta (Canada), Peru, Chile, Argentina, and Brazil, in North and South Africa, along the Nile, around the Mediterranean basin, in the Middle East, Afghanistan, Pakistan, India, Central Asia, Northern China, and Eastern Australia. In these regions, where food, feed, and fibre production require irrigation, farmers are stressed by restricted water availability, which may be due to limited access to water driven by regulation and competition for water, increasing pumping costs, or dry wells and streams. As there is very little irrigation in Germany, WSphys remains below 0.1 except in the populated Rhine-Ruhr and Berlin regions. Regarding recent developments in irrigation, Mehta et al. (2024) found that half of the irrigation expansion between 2000 and 2015 occurred in water-stressed regions.

A global map of water scarcity, where the demand for water is determined by water abstraction instead of consumptive water use (see [Fig. Supplement 1](#)), shows a rather similar spatial scarcity pattern. The spatial patterns merely differ for the eastern US and large parts of Europe, including Germany, where the water abstraction-to-renewable water resources ratio may even exceed 1 in densely populated grid cells due to the re-use of (treated) wastewater by downstream water users (see [Fig. Supplement 1](#)).

2.2.3 High temporal runoff variability heightens water quantity-related water stress

The physical water scarcity indicator WSphys, however, does not show a complete picture of water scarcity. It is based on long-term mean annual runoff and disregards the fact that runoff varies from month to month and from year to year. Even in regions with low mean water scarcity as indicated by WSphys, water security may not exist due to high temporal variability of runoff and thus the water available for humans and livestock from streams, ponds, and shallow groundwater wells. It can be assumed that for the same values of WSphys, a higher runoff variability makes the provision of water for human use more difficult, unless all water is sourced from deep groundwater; it increases the probability of suffering from water scarcity in relatively dry years. Runoff variability can be quantified by the indicator Ivar (see [Text Supplement](#)). Examples for countries with low mean water scarcity WSphys ([Fig. 2.2](#)) but high runoff variability and thus stress are Somalia, Botswana, and Namibia ([Fig. 2.3](#)). An alternative variability indicator that takes into account the vulnerability of water supply to runoff variability shows smaller values in the case of groundwater instead of surface water use, shows a similar spatial pattern as [Fig. 2.3](#), except for some countries, including Germany, that have

a very high fraction of groundwater use and are therefore less vulnerable to temporal runoff variability ([Fig. Supplement 2](#)). Overall, water scarcity affects dry regions much more strongly than humid regions, both due to the need for irrigation for crop production and the higher seasonal and interannual variability of runoff and streamflow.

2.2.4 Poor water quality causes water stress

Unsuitable water quality forms the second component of water stress. It is much more difficult to quantify water stress caused by unsuitable water quality, due to the huge variety of chemical constituents that affect water quality and its suitability for humans and freshwater biota. Water quality measurements are costly and therefore rare, in particular for groundwater and lakes (UNEP, 2024b). Global water quality characterisations only exist for a few chemical constituents and are highly uncertain. Monitoring the achievement of the Sustainability Developing Goal (SDG) 6.3 goal on improving water quality considers only five parameters: oxygen, electrical conductivity, pH, and the nutrients nitrogen and phosphorus (UNEP, 2024b). Unless individual countries choose to do so, toxic substances such as heavy metals and anthropogenic organic chemicals

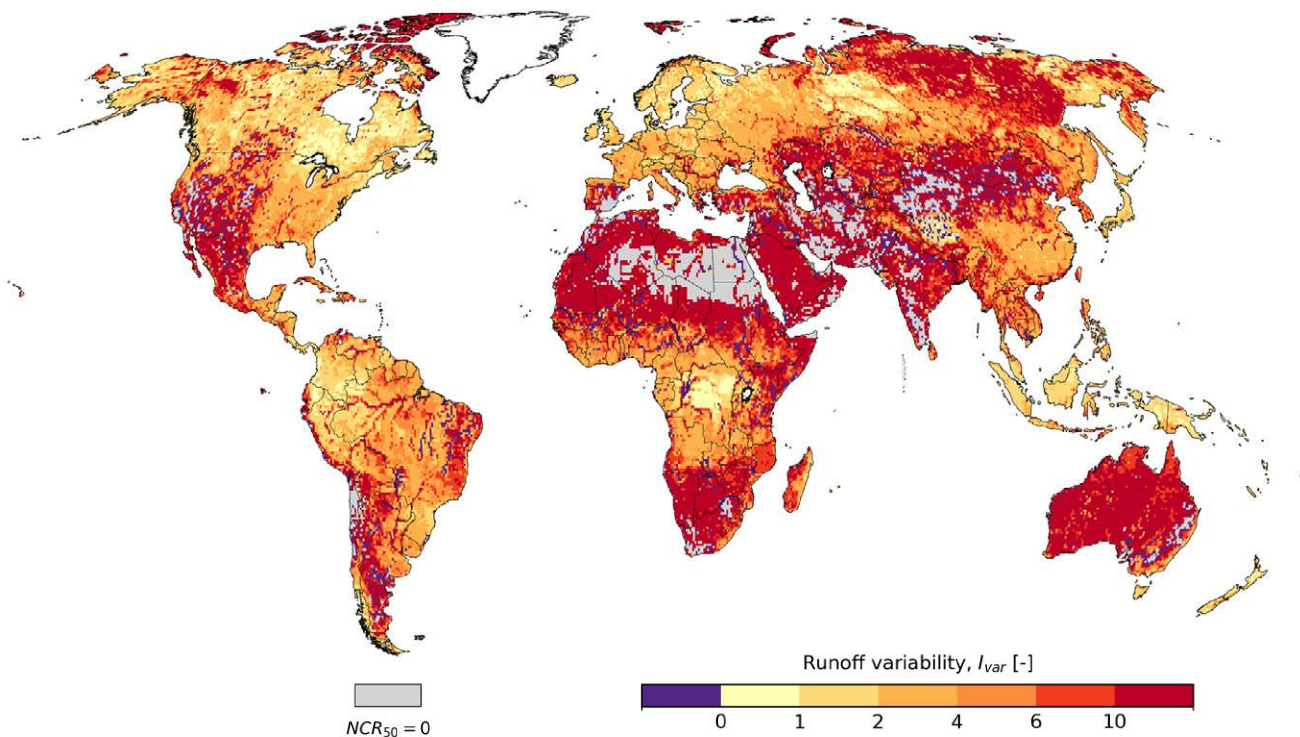
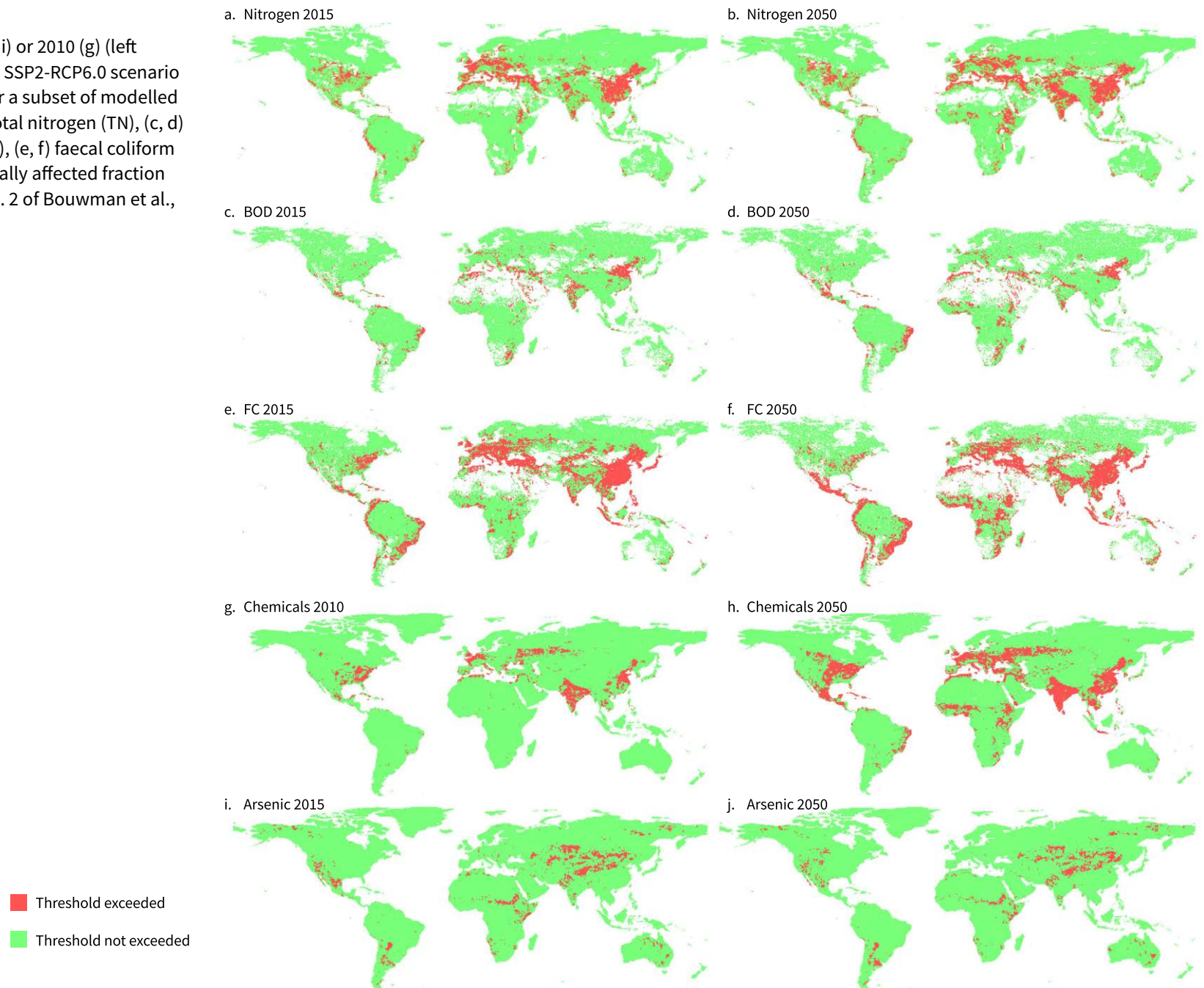


Fig. 2.3: Vulnerability of water supply due to the temporal variability of monthly runoff. The larger the vulnerability indicator I_{var} , the higher the vulnerability. If I_{var} is negative, there is a particularly high vulnerability, as people and freshwater ecosystems depend on water inflow from upstream. (Computed for the time period 1990–2019 by WaterGAP, 2026)

are not included due to insufficient data availability. The most reliable water quality data are available for the wealthiest countries group, while national authorities tasked with monitoring the quality of freshwater bodies in low-income countries require greater capacity to do so adequately (UNEP, 2024b). Electric conductivity or total dissolved solids are a measure of water salinity, which humans increase mainly by irrigation but also other activities such as de-icing; still, salinity is, with exceptions, a problem of dry regions, in particular in Asia (GlobeWQ, 2023). In 2023, about half of the water bodies in Europe were estimated to have a good ambient water quality, as compared to about 80 % in Africa and 30 % in Central and Southern Asia (UNEP, 2024b). Germany reported that 77 % of its water bodies have a good water quality (UNEP, 2024a).

The study of Bouwman et al. (2024) combined various global water quality models that estimate the concentrations of different chemical constituents, mostly in rivers, to derive consistent global maps that indicate where concentrations of various water quality components exceed thresholds below which harmful effects are considered tolerable (Fig. 2.4). Concentrations of the nutrient nitrogen around the year 2015 exceeded the threshold in large parts of Europe, including Germany, with 15 % of the area of industrialised countries (IND) exceeding the threshold (Fig. 2.4a). The corresponding

Fig. 2.4 Exceedance for 2015 (a–e, i) or 2010 (g) (left column) and 2050 according to the SSP2-RCP6.0 scenario (right column) of the thresholds for a subset of modelled water quality constituents: (a, b) total nitrogen (TN), (c, d) biochemical oxygen demand (BOD), (e, f) faecal coliform (FC), (g, h) ‘multisubstance potentially affected fraction of species’ (msPAF), and (i, j) As Fig. 2 of Bouwman et al., 2024.



values for Brazil, India, and China (BRIC) and the rest of the world (ROW) are 17 % and 11 %, respectively. The ecologically similarly relevant biological oxygen demand (BOD) is exceeded in smaller areas, with the lowest areal fractions in IND (**Fig. 2.4c**). The threshold for anthropogenic organic contaminants in surface waters (a summary of more than 1,000 different chemicals) is exceeded in the highly industrialised parts of Europe, including Germany, in all of South Asia, Eastern China, and parts of the USA (**Fig. 2.4g**). Excessive faecal coliform concentrations (FC) affect almost half of the globe (**Fig. 2.4**) but are problematic only if drinking water is not treated or water is used for swimming. Arsenic is a natural groundwater constituent with negative health effects that occur under specific geological conditions and affect people in BRIC and ROW countries who need to rely on groundwater wells for their drinking water supply. Comparing quantity- and quality-related water stress from **Fig. 2.2** and **Fig. 2.4** shows that physical water scarcity is exacerbated by poor water quality, particularly in South Asia and China.

2.2.5 Access to safe water services is improving but global inequalities remain

The third component of water stress, access to safely managed water services as defined

by (WHO/UNICEF, 2025), has improved over recent decades due to efforts also related to the Millennium Development Goals and the Sustainable Development Goals. However, while 100 % of the German population and 94 % of the population in Europe and North America used a safely managed drinking water service in 2024, this was still the case for only 32 % in Sub-Saharan Africa and for only 6 % in Chad (UNEP, 2024c).

Overall, the three components of water stress are distributed unevenly across the globe. Physical water scarcity is primarily fuelled by the demands of irrigated agriculture in dry regions and is further compounded by high temporal runoff variability. Water stress is also caused by poor water quality, as agricultural and industrial pollutants limit the amount of usable freshwater, though this remains difficult to monitor comprehensively on a global scale. Finally, while overall access to safely managed water services has improved in recent decades, regional inequalities persist, leaving populations in regions like Sub-Saharan Africa vulnerable compared to the universal access in wealthier regions of the world. In many dry regions of the globe, water quantity-related stress is exacerbated by water quality-related stress, strongly degrading aquatic habitats. However, poor water quality places a high stress on aquatic fauna and flora in many water-rich regions. In wealthy countries,

water quality-related stress for humans is mitigated by drinking-water treatment.

2.2.6 Germany's water abstractions are declining but climate change amplifies water stress

In Germany, water abstractions significantly dropped from 46 billion m³ in 1991 to 18 billion m³ in 2022 (UBA, 2025). While the largest share continues to be used as non-consumptive water in the energy sector for cooling power plants, this purpose's share declined from 62 % to 39 % in the same time period. This was driven by technological advancements in cooling systems and structural changes in energy production (e. g., the phase-out of nuclear power and a shift toward renewable energy sources such as wind and solar power). Major industrial sectors that require water are the chemical sector and the paper industry. Water abstractions for domestic supply also declined despite a growth in population and in 2022 accounted for 30 % of water abstractions, while 29 % of the abstracted water was utilised for manufacturing and mining, and only about 2.5 % for agricultural irrigation. Due to high rainfall and thus high productivity of rainfed agriculture, irrigation water use in Germany is very low as compared to most other countries. However, irrigation water use is growing in Germany due to climate change

(Bernhardt et al., 2025; Heilemann et al., 2026). Apart from the energy sector, which heavily relies on surface water resources for cooling purposes, the remaining sectors predominantly use groundwater. For the public water supply, about 71 % of water is withdrawn from groundwater and springs. In Germany, water abstractions add up to 10 % of the renewable water resources, i. e. the long-term difference between precipitation and evapotranspiration (UBA, 2025). Nevertheless, the recent droughts between 2018 and 2022 and in summer 2026 led to water stress situations with declining groundwater levels and low-flow situations (Wieland & Martinis, 2020) that impacted the water supply for society and ecosystems alike (Knutzen et al., 2025).

2.2.7 Germany secures universal access but fails to meet water quality targets

In Germany, surface and groundwater bodies are extensively monitored, particularly under the European Water Framework Directive (WFD) requirements. As of the latest assessment from 2021, only 9 % of surface water bodies achieve a “good ecological status” or “potential” (BMUV/UBA, 2022). The primary pressures on surface water quality stem from nutrient pollution, particularly nitrates and phosphates from agricultural runoff and chemical contamination from industrial and urban discharges. These

factors contribute to eutrophication, habitat degradation, and biodiversity loss, with lakes performing slightly better (20 % in “good status”) but still facing issues like harmful algal blooms due to high nutrient loads. Chemical status is particularly concerning, as none of Germany’s surface water bodies meets the WFD’s ‘good chemical status’ criteria, largely due to persistent legacy pollutants (e. g., mercury) and ongoing input of pesticides and heavy metals. Germany’s groundwater bodies show elevated levels of nitrate pollution from agricultural activities as the most widespread issue. Many regions exceed the 50 mg / l nitrate threshold, posing risks to drinking water quality and ecosystem health. Efforts to address these issues are underway, but progress is hindered by diffuse pollution sources, historical contamination, and the slow response of subsurface systems to remediation measures.

2.3 Human activities have changed the global water system in a significant and largely irreversible manner

2.3.1 Human water control increased sharply with the “Great Acceleration”

The development of human civilisations has been accompanied by the control of water for about the last 5,000 years, aiming at urban water supply, enhanced crop production by irrigation, flood and disease control, navigation, and the use of water as a source of power (Tvedt & Jacobson, 2006). Dams have stored and regulated water. Water has been diverted through canals to supply areas that need it. Wetlands have been drained and rivers have been straightened and deepened. More recently, advanced drilling and pumping technologies have led to widespread groundwater abstractions. Land cover changes, in particular for crop production, have led to changes in runoff and water quality. The latter was initially degraded first by the concentration of human waste in urban areas. Degradation was intensified by industrialisation in the 18th century, particularly due to heavy metals, and increasingly after World War II, particularly due to nutrients, artificial organic chemicals, and plastic particles. Acceleration of Earth system change after 1950 has been quantified by numerous indicators specifically relevant

to freshwaters, including water abstractions, numerous large dams, fertiliser consumption, and nitrogen flows into oceans, and has been termed the “Great Acceleration” within the Anthropocene (Steffen et al., 2015).

2.3.2 Groundwater and surface water depletion are mostly irreversible

Most alterations of the global water system are irreversible. Regarding water quantity, irreversibility is due to the irreversibility of (1) climate change and its impacts on water and (2) the depletion of groundwater and surface water (lakes and wetlands) by abstracting more water than can be replenished, even depleting groundwater that was recharged under different climatic conditions thousands of years ago. Jasechko (2026) could identify 67 cases around the world where groundwater levels rose after prolonged periods of decline due to certain management measures by humans. Most cases relate to the decline of local water tables, and in more than 80 % of the cases, changing the water supply to surface water sources was involved, and only in 6 % cases policy measures

to decrease groundwater use alone were successful. Under plausible socio-economic and demographic scenarios, it is very difficult to imagine that overexploitation, which is mainly caused by the need for irrigation to produce crops, will stop except under very beneficial circumstances, such as in wealthy California, which implemented laws and procedures to stop overexploitation of its groundwaters. For strongly depleted groundwaters, it could take decades to centuries to replenish them even if groundwater abstractions were stopped. Equally important, depletion of groundwater, lakes, and wetlands has already led to irreversible impacts such as land subsidence, soil degradation, and biodiversity loss (see [Chapter 2.2](#)). Even in cases of recovering groundwater tables after depletion, the land subsidence is not reversed, but merely stopped, at least within years to decades after stopping groundwater depletion (Jasechko, 2026).

There is also evidence that streamflow and other components of the water cycle may not recover completely from drought even after precipitation has returned to normal, due to persistent changes in transpiration by the

vegetation. During, and even seven years after, the end of the 10-years long Millennium Drought in southeastern Australia, streamflow as a fraction of precipitation was lower than before the drought in about one third of 161 near-natural catchments, likely due to a vegetation response to the drought, with increased evapotranspiration per unit of precipitation, and not only due to delayed filling of subsurface water stores; 80 % of the unrecovered catchments were found unlikely to recover anytime soon (Peterson et al., 2021).

2.3.3 Satellites reveal largely irreversible long-term water storage losses but also smaller gains

Satellites measuring the dynamics of the global gravity field can infer water storage changes from gravity anomalies, as water is the only medium on Earth capable of producing the observed temporal variations in gravity patterns. The so-called GRACE satellite missions have now been in space long enough to have observed long-term trends in freshwater storage that go beyond short-term variations due to stochastic climate variability. Climate variability causes freshwater flows, such as precipitation and streamflow, and the amount of freshwater stored to vary at sub-daily, daily, monthly, and annual scales. Therefore, 30-year averages are

usually computed as a baseline for climatological and hydrological assessment. Trends, i. e., changes in water storage, over 30 years or more can be assumed to be close to zero if there was no anthropogenic climate change or water depletion due to human water use. The GRACE satellites have now measured freshwater storage variations of the Earth for 24 years, since 2002. **Fig. 2.5** shows that most of the continents have been subject to significant trends in terrestrial water storage, based on satellite observations, indicating mostly irreversible changes due to climate change and human water use. Around the world, both increasing and decreasing water storage have been observed during the last 24 years. A trend of -1 mm/yr equates to a loss of 24,000 m³ of water per km² over the 24-year period, i. e., this amount of water is no longer stored as snow and ice, as soil moisture, as groundwater, or in surface water bodies. Increases in water storage are likely caused by anthropogenic climate change, while decreases are due to either climate change or large water abstractions. The highest decreases are due to glacier mass loss, e. g., in high northern latitudes, the Himalayas, and Patagonia, as well as to human overexploitation of the renewable water resources. Hotspots of groundwater depletion due to groundwater abstraction are the Central Valley (California) and the High Plains Aquifer in the USA, Mexico, northern Africa, the Middle East, northwestern India, and

northeastern China (Döll et al., 2014). India, the United States, Iran, Saudi Arabia, and China had the highest groundwater depletion rates in the first decade of the 21st century, and the rate of global groundwater depletion likely doubled in this decade compared to 1960–2000 (Döll et al., 2014). Overexploitation has also strongly depleted some lakes in dry regions in the 21st century, such as Lake Urmia in western Iran (DAHITI, 2026), following the earlier depletion of Lake Aral.

2.3.4 Climate change creates irreversible water-related changes on all land areas of the globe

Climate change leads to ubiquitous irreversible changes in water flows and storages worldwide which are different from local to regional impacts of human water use (**comp. Fig. 2.2**). Increasing climate variability leads to increased water stress. Heavy precipitation events and the resulting floods are projected to become more frequent in almost all regions, while droughts are projected to intensify on about half of the global land area (IPCC, 2021). Recent studies on future changes in flood and drought risk have not only considered changes in the physical occurrence of floods and drought, i. e. the changing flood and drought hazards, under different greenhouse gas emissions scenarios,

GSFC annual trend (2002–2025)

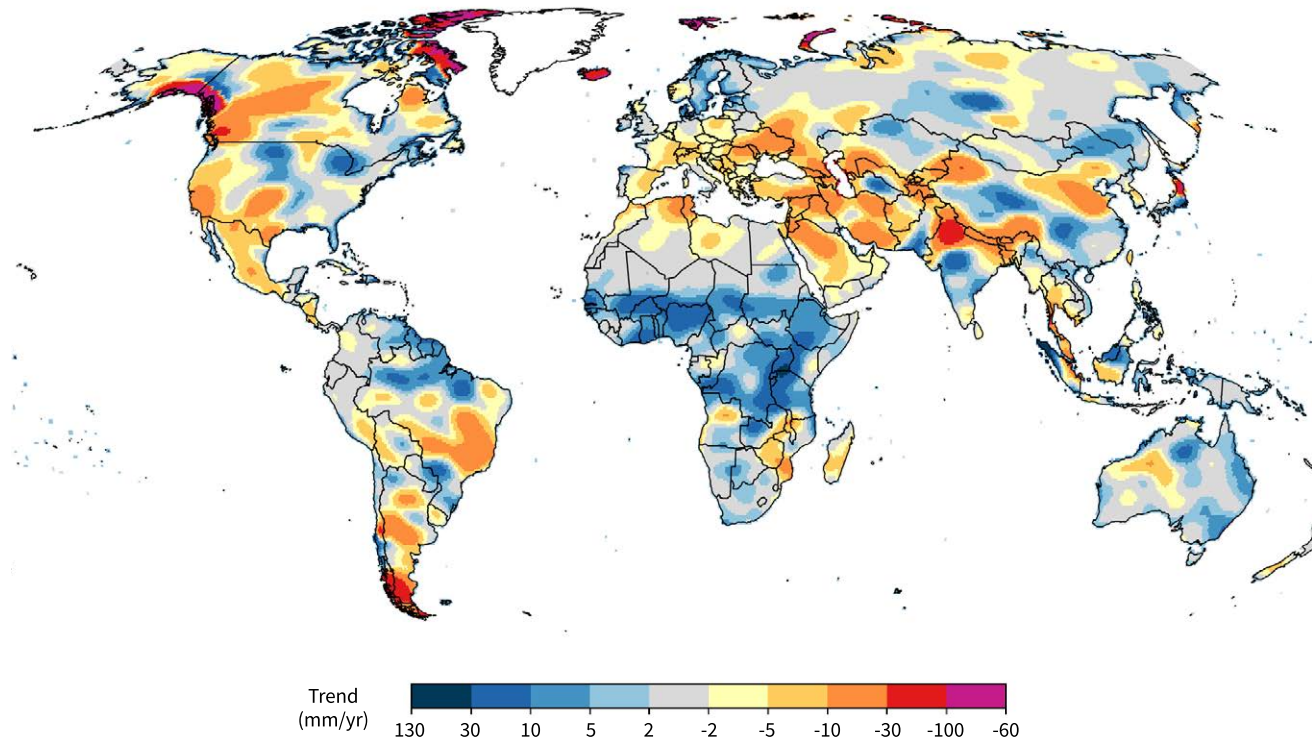


Fig. 2.5 Average annual change in terrestrial water storage between 2002 and 2025 as derived from GRACE satellite observations. Trends were computed using the GSFC mascon product (Loomis et al., 2019). Please note that due to the large footprint of the satellite observations, the trends are spatially smoothed; actual trends are more localised and can be larger. Without climate change and human water depletion, the trends would be close to zero.

but also scenarios of future socio-economic conditions affecting exposure and vulnerability. The study of Oh et al. (2023) identified parts of Africa, southern Asia and South America to face the most strongly increasing drought risks but points out that modelling uncertainties dominate the variations due to different scenarios. Coastal and riverine flood risk is projected to increase by 13 % even under moderate climate change scenarios until the mid 21st century (Duan et al., 2026).

With every increment of global warming, the changes in the water system, such as changes in mean and extreme precipitation, evapotranspiration, runoff, soil moisture, and groundwater levels, as well as sea level rise affecting coastal water resources, increase in a roughly linear manner (IPCC, 2021). If net zero greenhouse gas emissions were reached at some point, temperatures and probably also precipitation would remain approximately at the level reached when net zero greenhouse gas emissions occur (IPCC, 2021). Global negative net CO₂ emissions, which are very difficult to achieve, would gradually reverse changes in surface temperatures. However, many changes due to past and future greenhouse gas emissions will be irreversible for centuries to millennia (IPCC, 2021). Mountain and polar glaciers will continue melting for decades or centuries, loss of permafrost carbon following permafrost thaw is irreversible

at centennial time scales, and sea levels are bound to rise for centuries to millennia due to continuing deep-ocean warming and ice-sheet melt and will remain elevated for thousands of years (IPCC, 2021). Thus, saltwater will likely continue to encroach into coastal freshwater groundwater for the next centuries, threatening the water supply in low-lying coastal regions, including islands, and water storage in glaciers will no longer be available for water supply and ecosystems.

The largest relative decreases of mean annual runoff and streamflow is projected for the Mediterranean basin and Turkey, the Amazon basin and southwestern South America, Central America and the central USA, and southern Africa (Arias et al., 2021). **Fig. 2.6** of Gudmundsson et al. (2026) shows changes in seasonal mean river flow between the pre-industrial (1850–1900) era and the period when a 3 °C global warming with respect to the pre-industrial era will have been reached (Gudmundsson et al., 2026). Even though mean annual streamflow is projected to remain relatively stable in Central Europe and increase in boreal regions (Siberia, northern Canada, and Alaska), decreases are projected for the summer months. These projected changes are likely to remain for a long time if and when global warming can be limited to 3 °C, which is approximately the maximum global warming level that will be reached

if policies implemented by the end of 2020 will continue to be implemented (IPCC, 2022b).

2.3.5 Water quality degradation is largely irreversible due to natural processes and persistent emissions

Even if the emissions of compounds that degrade water quality decreases or stops altogether, degradation is (largely) irreversible for a large number of contaminants. Organic microcontaminants and heavy metals tend to be strongly adsorbed to the solid particles in the groundwater or the streambed and will re-enter the water phase slowly once the concentration in water decreases. Groundwater, including dissolved contaminants, usually moves very slowly, e. g., only 1 m per month, year, or even decade. Plastic particles degrade slowly to ever smaller but no less problematic grain sizes. Groundwater and river water salinisation due to irrigation or climate change is also expected to be largely irreversible (Kaushal et al., 2023).

In any case, emissions are not likely to decrease anytime soon for most problematic compounds. The UNEP 2024 report on achieving the SDG goal of improved ambient water states that in areas where good data were available for 2017, 2020, and 2023, water quality has degraded over time (UNEP, 2024d). Scenarios for five water

constituents for 2050 developed in (Bouwman et al., 2024) indicate a rather stable global pattern of water stress due to water quality, with increasing water quality problems associated with nitrogen, biological oxygen demand, and anthropogenic chemicals in nearly all parts of the world compared to 2015 (**Fig. 2.4**). Nitrogen pollution is projected to stabilize in industrialised countries but to expand in BRIC and ROW countries, and exceedance of the BOD threshold is increasingly being exceeded in all parts of the world. The scenario includes a limited increase in the area with exceedances of anthropogenic chemicals in industrialised countries, and an approximate doubling in BRIC and ROW countries. Faecal coliforms will decrease in many industrialised countries, stabilise in BRIC countries, and increase in many ROW countries. The number of people affected by excessive As concentrations is expected to increase, while the affected area is expected to remain constant. Although climate-induced water quality degradation due to increases in water and surface temperatures or melting of the cryosphere has been observed (medium confidence), evidence of global-scale changes in water quality is limited because many studies are isolated and have limited regional coverage (Caretta et al., 2022). Climate change is projected to increase incidents of water pollution, salinisation, and eutrophication in some local rivers and lakes due to increasing drought and flood events and

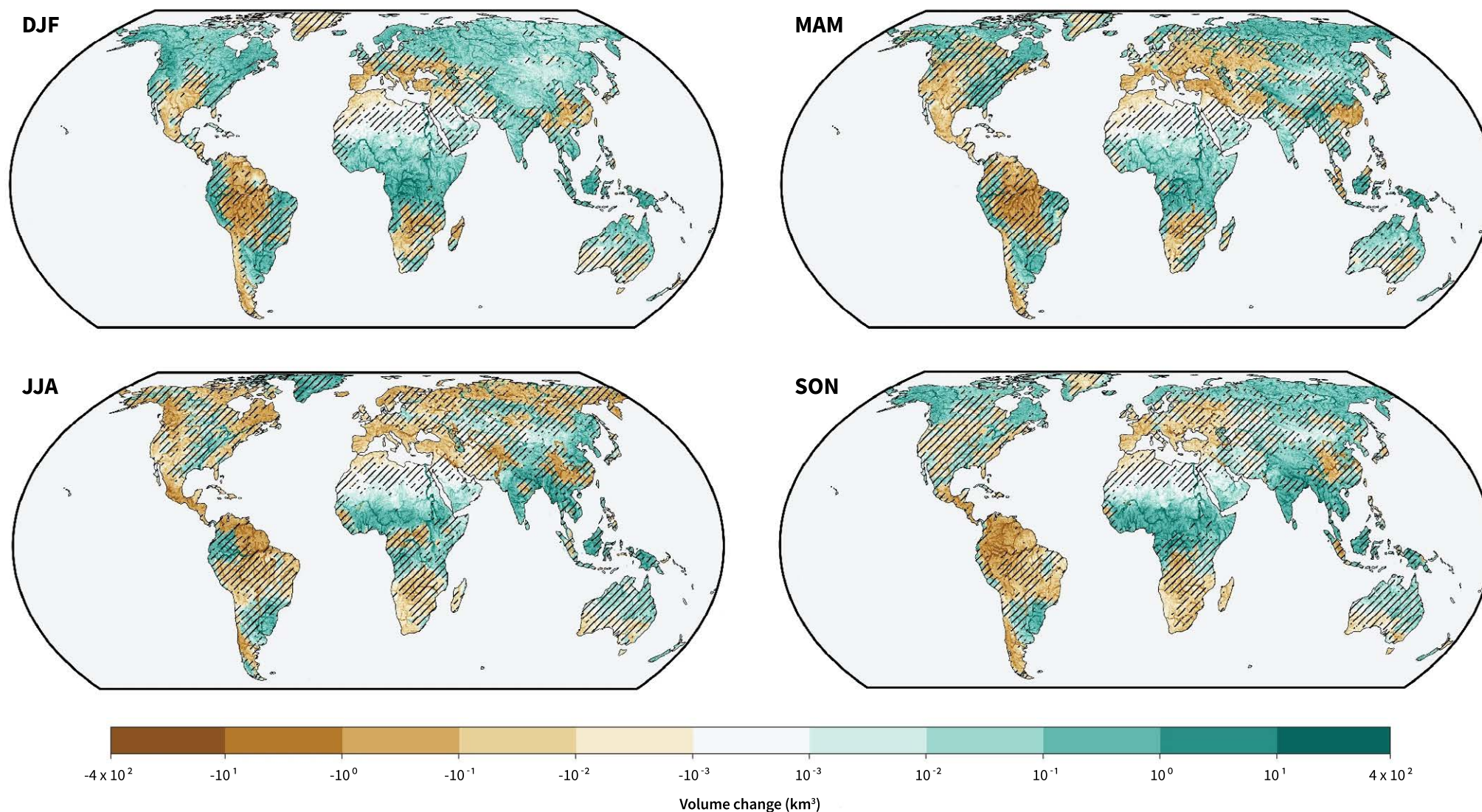


Fig. 2.6 Future change in seasonal mean river flow between the pre-industrial (1850–1900) era and the period when a 3°C global warming with respect to the pre-industrial era will have been reached, over December, January and February (DJF); March, April and May (MAM); June, July and August (JJA); and September, October and November (SON). Hatching indicates grid cells and regions where fewer than six out of eight ensemble members agree on the direction of change. (Figure 6 of Gudmundsson et al., 2026)

rising sea levels and water temperatures, but exact quantification is lacking at a global scale (Caretta et al., 2022). However, erosion is projected to increase globally, as well as in Germany in the future, predominantly due to climate change rather than changes in land use (Caretta et al., 2022), with negative consequences for water quality and human water supply.

Beyond their implications for human water security, these transformations of the water cycle have profound ecological consequences. As discussed in [Chapter 2.2](#), freshwater biodiversity loss and the degradation of ecosystem services are driven by the same underlying processes – water abstraction, pollution, hydro-morphological alteration, climate change, and increasingly also telecoupled resource use. Consequently, effective water governance must address human water security and ecological water requirements simultaneously rather than treating them as separate management objectives.

2.4 Water governance as a response to water stress

The anthropogenic alterations to the global water system and the resulting water stress become evident at local and regional levels, where conflicts over scarce water resources emerge. Yet, the structural drivers of these tensions extend beyond localised boundaries, operating across divergent social-ecological scales. At the micro level, water use is shaped by individual and household consumption routines, available technologies, and localised cultural practices (Inman & Jeffrey, 2006). At the meso-level, domestic, industrial and agricultural water use is structured by settlement patterns, sector-specific technologies, environmental conditions and regulatory frameworks. At the macro level, these patterns are sustained by broader economic systems and international supply chains, embedded in telecoupling flows (Moore et al., 2024).

2.4.1 Large-scale water transfers: physical and virtual mechanisms of stress relocation

These large-scale interconnections are operationalised through physical inter-basin water transfers (IBWT) and virtual water trade. IBWTs

represent a global engineering response to regional water scarcity, wherein water is diverted from low-consumption to high-consumption areas. This often involves extracting water from rural regions and transporting it to urban centres, industrial hubs, or agricultural regions. Such transfers frequently lead to environmental degradation in donor basins, including salinisation and aridification, and the spread of waterborne diseases (Zhuang, 2016), as well as stakeholder conflicts (Gupta & Van Der Zaag, 2008). Moreover, these long-distance water transfer schemes tend to create path dependencies and lock-in effects, limiting the adoption of alternative, more sustainable solutions (Kuhn et al., 2024). A study by Shumilova et al. (2018) revealed that if all planned IBWT mega-projects were to be realised, 1,910 km³ of water would be transported annually through “artificial rivers”. This volume would be equivalent to 26 times the mean annual discharge of the River Rhine. One of the largest schemes globally is the South-to-North Water Diversion (SNWD) project in China, intended to transport about 44.8 km³ of water from the Yangtze River to the northwest. This project supports the larger Beijing region but has adverse effects on aquatic biodiversity in the donor regions (Yan et al., 2023).

In parallel, virtual water trade serves as a second, equally significant mechanism for redistributing water stress across distances. It is defined as the total volume of freshwater consumed (green and blue water) or polluted (grey water) across the life cycle of a product (Allan, 1998). Embedded in globally traded commodities, virtual water links hydrologically disconnected regions through patterns of consumption and production. The international virtual water network enables water-scarce, or high-consumption countries to externalize their water footprints to distant exporting regions (Hoekstra & Mekonnen, 2012). As a result, many localized water conflicts cannot be understood without considering the global trade system. Empirical analyses confirm that virtual water flows have intensified in recent decades (Dalin et al., 2012), driven in part by multilateral trade agreements (Falsetti et al., 2022). Virtual water export can generate ecological externalities in regions, which are often already characterised by acute water scarcity (Mekonnen et al., 2024; Vallino et al., 2021). A recent review by Mekonnen et al. (2024) synthesised global estimates of virtual water trade, revealing that between 1980 to 2014, approximately 66 % of these flows were tied to agricultural products, followed by wood

products (9 %) and industrial goods, energy, and mining products (24 %) (Mekonnen et al., 2024).

2.4.2 The blue virtual water footprint threatens freshwater biodiversity

Looking specifically at the impacts of virtual water trade on freshwater biodiversity, one relevant metric is the degree to which environmental flow requirements are infringed. Mekonnen and Hoekstra apply the widely accepted benchmark that 80 % of natural runoff must be reserved for ecosystems to avoid adverse ecological consequences (Mekonnen & Hoekstra, 2020; Richter et al., 2012).

By analysing the blue virtual water trade volumes, the authors assessed whether countries and regions are overstraining their water resources at the expense of environmental flows and, consequently, freshwater biodiversity. Their findings reveal that more than half of the global blue water footprint originates from regions where the sustainable threshold of 20 % is exceeded. While not all blue virtual water is traded globally, approximately 43 % of global blue virtual water trade can be traced to regions where environmental flow requirements are infringed (Mekonnen & Hoekstra, 2020).

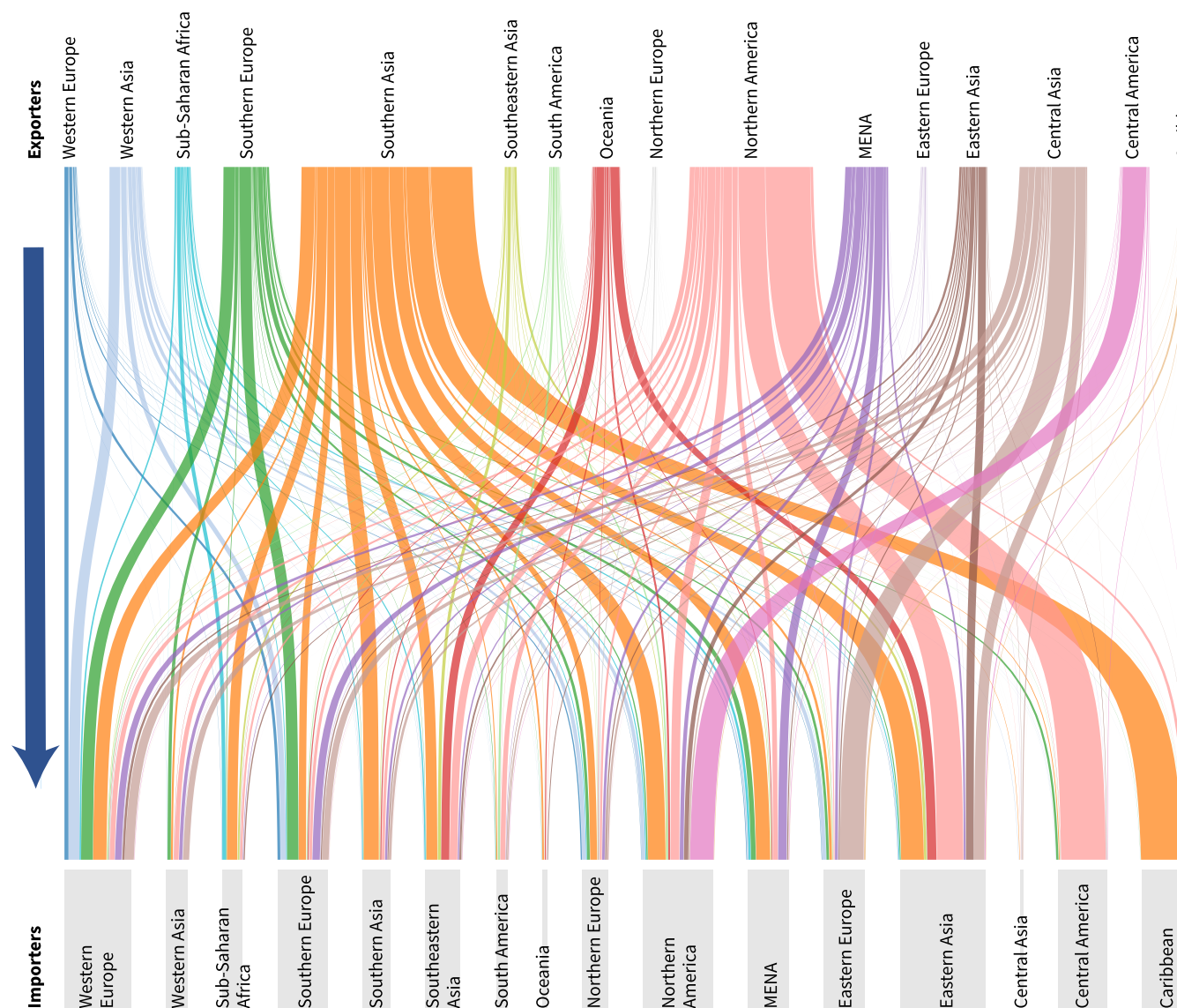


Fig. 2.7 Blue virtual water trade that infringes environmental flow requirements in the exporting region. Ribbons show the relative blue virtual water volumes exported from world regions shown at the top to the world regions at the bottom for the period 1996–2005. (Source Mekonnen & Hoekstra, 2020)

Fig. 2.7 provides an illustration of the blue virtual water trade component. It highlights that the majority of blue virtual water volumes infringing local environmental flow requirements originate from Southern Asia and North America, with North America, Eastern Asia, and Western Europe being the strongest importers of these volumes. In terms of specific products, the study shows that 29 % of the blue water footprint that infringes environmental flow requirements comes from wheat, 17 % from rice, and 8 % from cotton. Overall, crops produced for animal feed make up 13 % of the blue virtual water footprint linked to infringed environmental flow requirements (Mekonnen & Hoekstra, 2020).

Another study by Rosa et al. (2019) more closely examined the unsustainable irrigation water consumption (UWC), defined as agricultural water use that directly exceeds local renewable blue water availability, thus leading to depletion of surface and groundwater bodies. **Fig. 2.8** shows the volumes of UWC traded globally and the respective importing and exporting regions. Herein, USA was identified as the largest exporter of UWC, accounting for about 22 % of global UWC transfers. China, on the other hand, is the largest net importer of crops connected to unsustainable irrigation water consumption. This study also shows that the recent increase in

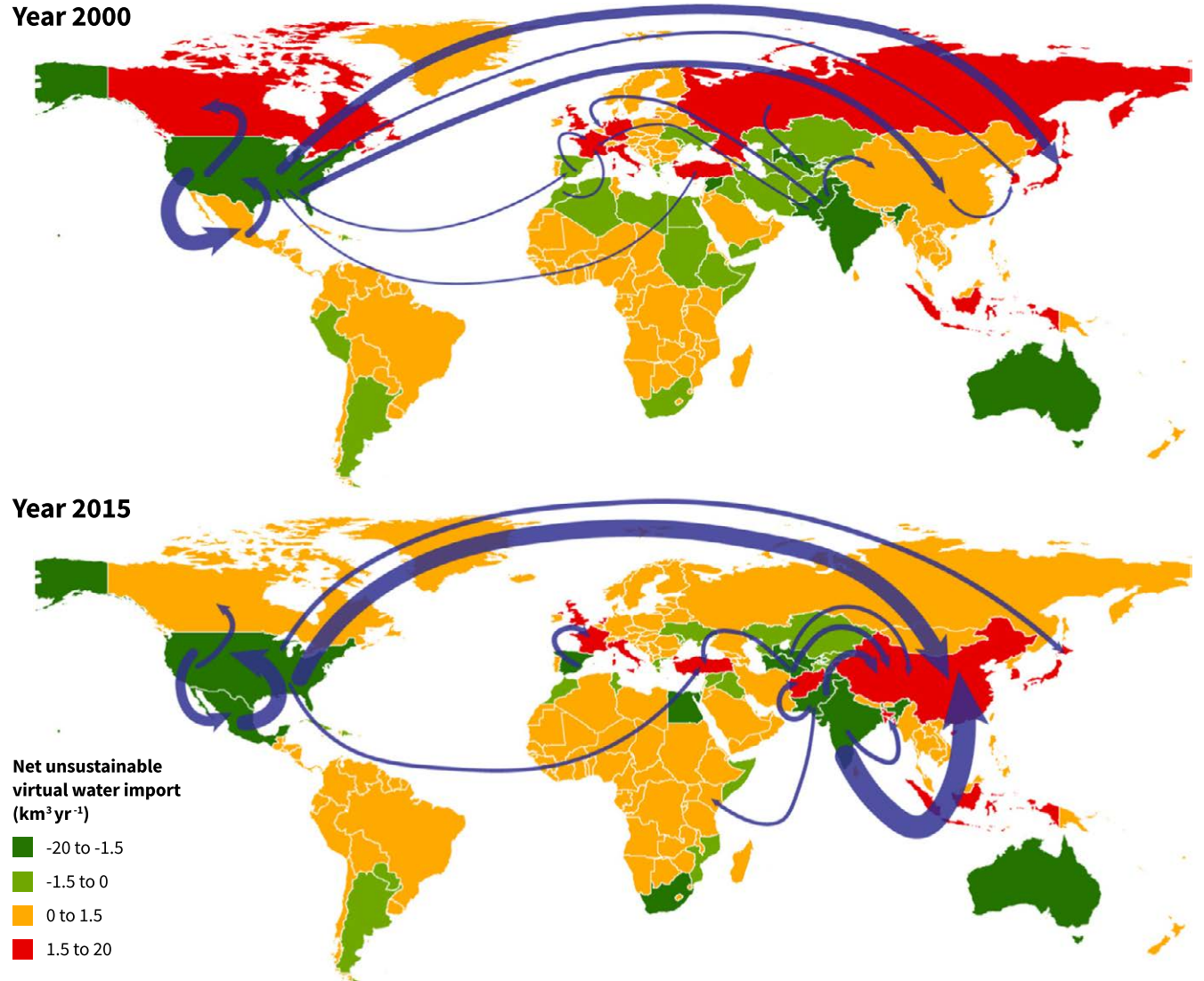


Fig. 2.8 Volumes of unsustainable irrigation water consumptions embedded in the trade food commodities for the years 2000 and 2015. Green countries represent net exporters of UWC, while net importers of UWC are shown in orange and red colours. The arrows illustrate the fifteen largest UWC flows globally. (Source: Rosa et al., 2019)

UWC was due to an expansion of maize, wheat and cotton production.

Overall, global value chains and transboundary trade policies operate largely independently of catchment realities and thus, commodity consumption remains structurally decoupled from local hydrological limits (Hoekstra & Mekonnen, 2012). By externalizing the water footprint, high-consumption countries avoid immediate domestic resource depletion (Demeke et al., 2024; Mekonnen & Hoekstra, 2020; Rosa et al., 2019).

2.4.3 Germany sustains a strong blue virtual water footprint abroad

Considering Germany's role in the global virtual water system in greater detail, it becomes evident that the country maintains a substantial externalised water footprint. According to Mekonnen and Hoekstra (2020), Germany imports blue virtual water volumes of about 9.1 km³, annually. Around 40 % of this volume is connected to infringed environmental flow requirements in the originating countries (Mekonnen & Hoekstra, 2020).

A closer examination of Germany's virtual water imports reveals that cotton, citrus fruits, and sugar cane are among the commodities with

the highest volumes of unsustainable irrigation water used in their production (Rosa et al., 2019). The case of cotton is particularly illustrative, as it underscores that not only food consumption but also textile production and consumption significantly contribute to water scarcity in exporting regions. In Germany, most cotton is imported from India and Pakistan. Overall, this import pattern can be confirmed for many European Union countries. In total, the EU imports 18.7 km³ of UWC volume, of which cotton accounts for about 36 %. It demonstrates how the demand of industrialised regions like the European Union for non-food agricultural products such as the textile industry drives water-intensive practices in already water-stressed regions, worldwide.

2.4.4 Competing water demands and climate pressure create conflicts

The aforementioned pressures caused by climate change, changing water use patterns and telecoupling flows create acute conflicts over water allocation and desired water quality levels on a local level. Traditional institutional frameworks regularly fail to balance competing sectoral requirements and responsibilities (Pahl-Wostl et al., 2012) and often lack an adequate consideration of environmental flow requirements (Arthington et al., 2024). Often, diverse stakeholders such as farmers, companies,

municipal water suppliers, and environmental conservation groups compete directly over water resources (Kosow et al., 2024; Sivakumar, 2011). In such conflicts, water can be considered as a trigger of conflicts, it can be used as a weapon or it may be a casualty as a result of conflicts (Gleick & Shimabuku, 2023). Disputes can even escalate to armed conflicts when arid climate patterns manifest locally (Bagwell et al., 2026).

These local frictions are reinforced and locked in by historical water rights and institutional path dependencies that maintain misallocations (Madani, 2026). This legal rigidity marginalises ecological water requirements (Arthington et al., 2024) and maintains power asymmetries, thereby excluding vulnerable or small-scale users such as Indigenous communities, subsistence farmers, or marginalised local populations from formal decision-making processes (Gupta et al., 2013). Furthermore, where formal allocation mechanisms fail to adapt or enforce limits, informal and unregulated practices emerge, such as unauthorized well-drilling and ground-water abstraction. These informal regimes establish allocation systems that actively undermine centralized management institutions and accelerate localised resource abstraction (Pahl-Wostl et al., 2012).

2.4.5 Existing water governance models struggle with complexities

Water governance describes formal and informal mechanisms that establish and sustain certain water-use patterns. Over recent decades, approaches have emerged to manage freshwater resources and ecosystems, each with specific strengths and weaknesses. Integrated Water Resources Management (IWRM) was established to overcome fragmented, sector-by-sector models and can be considered the dominant paradigm in water management today. It explicitly acknowledges the vulnerability of water resources and ecosystems, the necessity of stakeholder participation, the importance of gender equality and the social and economic value of water. In theory, IWRM's strength lies in aligning management boundaries with natural river basins and establishing cross-sectoral coordination, framing environmental water requirements as essential for water security. However, IWRM often remains theoretical in practice, struggling to address structural power asymmetries that marginalise environmental requirements. This misalignment hinders enforcement and thus weakens its ambitions to prevent overdraft, habitat fragmentation, and the degradation of freshwater ecosystems (Bilalova et al., 2026). [Fig. 2.9](#) shows the level of adoption of IWRM frameworks, globally, highlighting limited implementation in Central Africa, the Middle

East and South America. In contrast, Northern America, Europe, Australia and parts of Asia, including Russia and China, already adopted legal frameworks that incorporate IWRM principles. Among countries with implemented IWRM frameworks, differences can be found in the centrality of decision-making processes and the degree of stakeholder engagement (Ravesteijn et al., 2009).

Managing water within clearly defined jurisdictions is one challenge. When water crosses borders, however, responsibilities and vulnerabilities must be identified and negotiated within a transboundary context. Around 60 % of global freshwater flows cross national boundaries, but only 43 out of 153 countries that share transboundary water courses have operational cooperation arrangements covering at least 90 % of their shared catchment areas (UNESCO & UNECE, 2024). [Fig. 2.9b](#) indicates the proportion of global transboundary basins that have already adopted some form of international agreement on managing common water resources. Though fewer data is available here than for national regulations, it becomes clear that all global regions require more formal international agreements to enhance the management of transboundary water systems.

In practice, governance models rarely align perfectly with specific theoretical frameworks.

While integrated approaches like IWRM remain highly influential, alternative and complementary paradigms also exist. For instance, polycentric governance systems can be considered practical ways to implement effective integrated water management. They combine decentralised decision-making with integrated coordination across multiple levels. The Emscher River in Germany exemplifies this, as a polycentric setting successfully enabled its transition from a heavily polluted industrial open sewer to a revitalised river network with enhanced ecosystem quality (Pahl-Wostl & Knieper, 2023). Similarly, California's Sustainable Groundwater Management Act (SGMA) illustrates this approach in response to severe groundwater depletion. It mandates the establishment of regional Groundwater Sustainability Agencies (GSAs), ensuring strong stakeholder participation to develop locally adapted solution strategies within a state-wide regulatory framework (Liu et al. 2022). In contrast to integrated approaches, however, top-down command-and-control frameworks, such as mandatory abstraction caps or industrial discharge prohibitions, establish clear, enforceable ecological thresholds but suffer from a lack of localized flexibility. Their rigidity frequently drives informal, unmonitored abstraction routines among local users, which effectively bypass state environmental protections and accelerate biodiversity decline. More

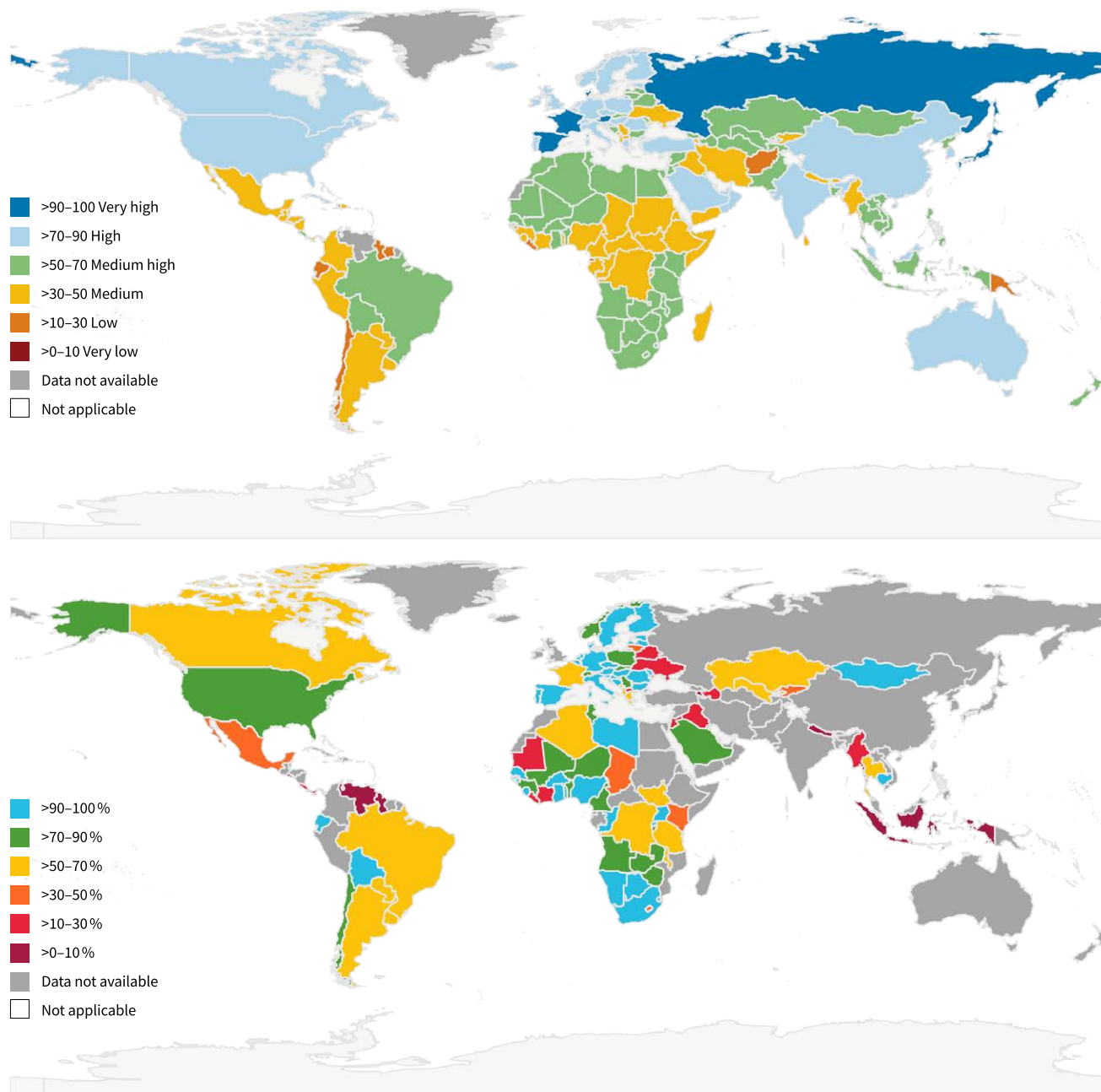


Fig. 2.9: Top: SDG-indicator 6.5.1 on the global status of the 'degree of integrated water resources management implementation', based on data available up to 2023. Bottom: SDG-indicator 6.5.2 on the global status of the 'proportion of transboundary basin area with an operational arrangement for water cooperation', based on data available up to 2023. (Sources: UNEP, 2024b; UNESCO & UNECE, 2024)

market-based instruments, such as water pricing and tradable entitlements, aim to maximise water allocation efficiency but often fail to internalise ecological externalities (Wheeler et al., 2025). Market mechanisms can, for instance, exacerbate trade-offs between global climate mitigation and local adaptation, as showcased by the recent endeavour to implement green hydrogen production sites (Rödl, 2025). In these cases, water-intensive hydrogen projects, driven by market incentives, often target regions with extensive renewable energy potential (e.g., PV) that are, however, already water-stressed.

To address the structural deficits of conventional approaches, however, recent ideas on adaptive governance argue for a paradigm shift from a static view of ‘institutional fit’ to a dynamic framework of ‘institutional fitness’. This view demonstrates that as hydrological cycles become tightly coupled across larger spatial scales in the Anthropocene, effective governance requires enhanced information-processing capacity, response diversity, and long-term anticipatory flexibility rather than relying on closed, stationary biophysical boundaries (Moore et al., 2024). Thus, in an era of telecoupling and virtual water transfers, water governance must transcend traditional territorial and sectoral boundaries to address the spatial, institutional, and scalar mismatches that characterize globalized water systems (Coenen et

al., 2023; Newig et al., 2019). In addition, investments in water across sectors are chronically underfunded due to a systemic undervaluation of its economic, social, and environmental benefits (OECD, 2026b; see [Chapter 2.5](#) for more details).

2.4.6 The EU Water Framework Directive misses its objectives

Within the European Union, the primary legal mechanism for water governance is the Water Framework Directive (WFD), which follows the IWRM principles and requires member states to achieve “good ecological and chemical status” across all river basins and their water bodies by 2027 (now postponed from 2015 originally) (Directive 2000/60/EC). Despite its progressive conceptual integration of ecological flows, the WFD’s implementation faces profound institutional deficits (European Commission, 2025; Linton & Krueger, 2020). Although, a positive trend in overall status indicators is visible for many water bodies in Europe, a substantial proportion of surface and groundwater bodies consistently fails to meet compliance targets (Elsner, 2023; European Commission, 2025). These shortfalls stem from persistent administrative barriers, including the misalignment of ecological boundaries with political jurisdictions (‘spatial fit’), weak enforcement of environmental

flows (i.e., the amount of river flows that are necessary for the health of the river ecosystem) against industrial and agricultural interests, and a systemic lack of binding transboundary coordination and regulatory fragmentation (Linton & Krueger, 2020; Moss et al., 2020). In Germany, the latter is visible in the enforcement of the Nitrates Directive and the Drinking Water Ordinance. Despite clear directives, agricultural nitrate pollution remains a challenge, with groundwater bodies in regions characterised by intensive livestock production (e.g., Lower Saxony and North Rhine-Westphalia) regularly exceeding maximum concentration limits (European Commission, 2025).

2.4.7 Germany’s National Water Strategy modernises governance but overlooks global impacts

In an attempt to unify these fragmented approaches, Germany adopted the National Water Strategy in 2023 (BMUV, 2023). The strategy represents a key instrument in the country’s water management by introducing participatory principles, modernizing infrastructure, and expanding digital monitoring frameworks, among other things. It seeks to move beyond purely structural adaptations to integrate proactive conflict management among competing user groups, such as public water

suppliers, agriculture, and industry. It intends to establish guidelines to coordinate regional water distribution and set clear prioritisation rules during periods of water scarcity. Furthermore, it places an emphasis on freshwater ecosystem protection, mandating a transition towards a semi-natural water regime and the large-scale restoration of degraded aquatic habitats (BMUV, 2023). Despite these advancements, the strategy exhibits a major scale mismatch. It remains domestically inward-looking. It contains no binding regulatory or accounting mechanisms to address Germany's extensive international virtual water imports, thereby omitting the country's profound role in driving water stress within catchments abroad.

2.5 Key messages

- **Humans make use of blue water from surface and groundwater for domestic, industrial, energy and agricultural purposes.**

Irrigation is the dominant water use globally. Because societal actors essentially decide how much water is abstracted and how much remains for ecosystems, competition is strongest between humans and freshwater biodiversity. This adversely affects biodiversity and ecosystem functioning by reduced water availability and degraded water quality.

- **Water stress encompasses water quantity, water quality, and access to safely managed water services, and all three components are unevenly distributed around the globe.**

Physical water scarcity is concentrated in dry, densely populated regions with intensive irrigated agriculture, and is compounded by high temporal runoff variability, poor water quality and, in many world regions, insufficient access to safe water services. Even in Germany, declining water abstractions do not prevent climate change induced water stress, and many water

bodies fail to meet ecological and chemical quality targets.

- **Human water control has grown strongly since the mid-twentieth century, leading to alterations that are, in many cases, irreversible.**

In some regions, depleted groundwater and surface water may take decades to centuries to recover and land subsidence cannot be undone. Satellite observations reveal long-term water storage losses across most continents, driven by climate change, and by groundwater overexploitation in dry regions, but also storage increases due to climate change in some regions. With every increment of global warming, changes in water flows and storages intensify.

- **Water stress and the conflicts it provokes are shaped not only locally but by telecoupled flows.**

Long-distance water transfers and virtual water trade relocate water stress to distant regions, and importing countries such as Germany externalize substantial water footprints abroad. Conventional governance approaches struggle with power

asymmetries, institutional path dependencies and weak enforcement, as illustrated by the EU Water Framework Directive's missed targets. Effective responses therefore require adaptive, polycentric governance that addresses both domestic allocation conflicts and the externalised water footprints embedded in global trade.

Box 1: Crisis management in the face of irreversible changes

Petra Döll

Are we currently experiencing a water and biodiversity crisis? The term ‘crisis’ is frequently used in academic, policy and everyday language, albeit often in a rather vague manner. There is no universally accepted definition for it. Generally, a crisis refers to a socially relevant, hazardous, sudden and temporary deviation from normality that inflicts a shock on the system (Steg, 2020). A crisis requires urgent decision-making and an end to ‘business as usual’. Usually, the term implies that the hazardous deviation from normality can be overcome by effective governance, allowing the system to return to its normal or pre-disturbance state. Therefore, it may be misleading to talk about the ‘climate crisis’, the ‘biodiversity crisis’, or the ‘water crisis’, as full recovery to the pre-disturbance state is not possible due to the largely irreversible environmental changes caused by human activities in the Anthropocene.

Climate change, for example, is effectively irreversible over the coming centuries. Even if we can stop global warming, the climate will not return to pre-industrial, i. e., pre-crisis conditions within this timeframe (Lee et

al., 2021). Moreover, many of its impacts are even more persistent than climate change itself. Sea level rise, glacier mass loss, and alterations to major biogeochemical cycles will continue long after global temperatures have stabilised (Lee et al., 2021). Thus, many climate-induced changes to water resources are likely to persist for millennia.

The consequences of climate change for biodiversity are expected to be even more pronounced because evolutionary processes operate over much longer timescales. However, anthropogenic changes in water resources and biodiversity are not only irreversible due to climate change but also due to other drivers. As argued by Madani (2026), many coupled human–water systems suffer from ‘water bankruptcy’, as water use persistently exceeds hydrological carrying capacity (insolvency), progressively eroding the water and biodiversity capital to the extent that some damages become irreparable (irreversibility) (Madani, 2026). One example of physical irreversibility is the permanent land subsidence due to groundwater abstractions. However, irreversibility may also result from

prohibitively high social costs of recovery to the non-degraded state. Water management in the face of irreversible changes requires the identification, communication, and costly implementation of realistic management targets. Using the term ‘water crisis’ for systems that can no longer recover their historical baseline conditions may encourage inappropriate management approaches that prioritise short-term emergency measures, such as those suitable for temporary water crises caused by droughts (Madani, 2026). The concept of water bankruptcy proposed by (Madani, 2026) can be extended to water systems that suffer from largely irreversible deterioration of water quality (e. g., caused by persistent organic substances produced by the chemical industry) and changes in aquatic species composition (e. g., due to climate change).

In summary, the state of global water resources and aquatic biodiversity may justifiably be called a ‘crisis’ in the sense that it constitutes an emergency and demands urgent and unconventional action. In this sense, we are facing an interconnected polycrisis

involving climate change, water scarcity, pollution, and aquatic biodiversity loss. **Fig. Box 1** illustrates that we can slow down (blue and green lines), stop (blue line), or even reverse (green line) the degradation that leads to a crisis and would otherwise continue in an unmitigated way (red line). However, due to irreversible changes before or during the crisis, we cannot hope to reach the state before the crisis, even with the strongest crisis management efforts. Now, we need to understand and communicate the irreversible changes to which we must adapt, even after overcoming the polycrisis. And we must find ways to prevent any further degradation beyond what is unavoidable due to irreversible changes that we have already triggered.

Crisis in the face of irreversible changes

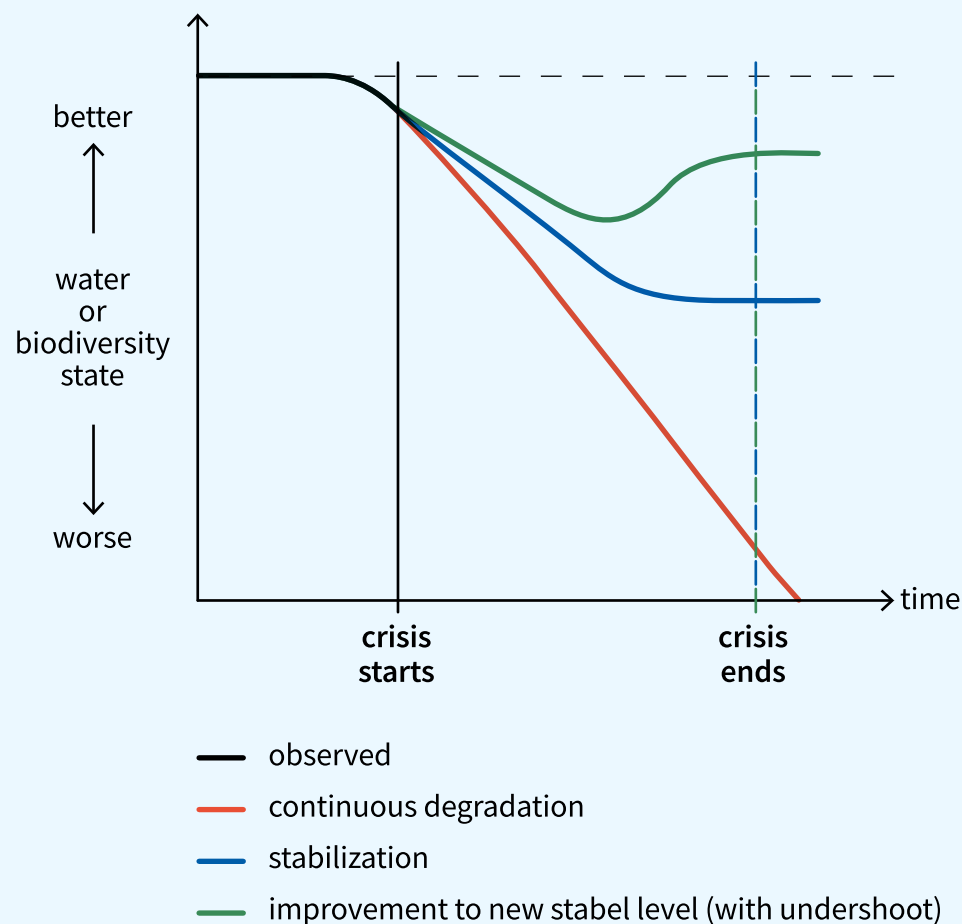


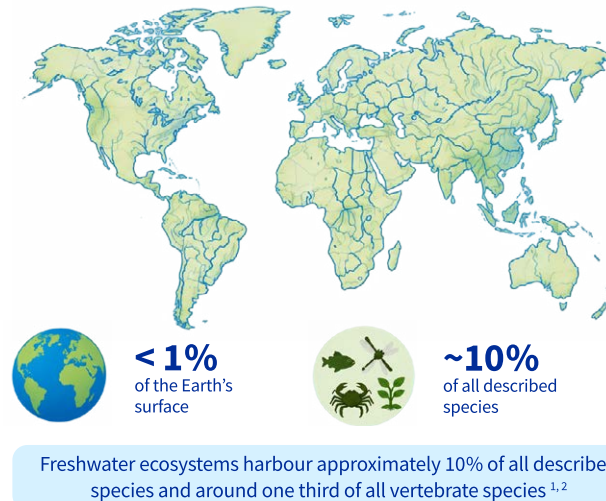
Fig. Box 1 Conceptualisation of crisis development in the face of irreversible changes in water and biodiversity systems. The state indicators shown on the y-axis may be, for example, groundwater level or species number. Please note that the crisis ends only in the case of stabilisation (blue) or improvement to a new stable level (green).

3 Freshwater ecosystems and biodiversity

Andrea Sundermann and Peter Haase

Although freshwater ecosystems cover less than 1 % of the Earth's surface, they support a disproportionately large share of global biodiversity, harbouring approximately 10 % of all described species and nearly one third of all vertebrate species (Strayer & Dudgeon, 2010; Tickner et al., 2020) (Fig. 3.1). This exceptional biodiversity makes freshwater ecosystems global biodiversity hotspots. Beyond this important role, they provide essential ecosystem services, including drinking water supply, food production, nutrient cycling, flood regulation, carbon sequestration, and multiple recreational opportunities (see also Chapter 4). Maintaining their ecological integrity is therefore fundamental not only for biodiversity conservation but also for water security, climate adaptation, and sustainable socio-economic development.

Global context



Freshwater diversity in selected taxonomic groups

Taxonomic group	Freshwater diversity (approximate number)	Key message
Fishes ³	18,898 species (of ~37,000 total)	51% of all fish species (species occurring in freshwater at any life stage)
Odonata ⁴	5,680 species (of 5,680 total)	Larvae of almost all species develop in freshwater
Decapoda ⁵	~3,000 species (of ~15,000 total)	Spend a significant part of their life cycle in freshwater
Mollusca ^{6,7}	~5,000 species (of unknown total)	Live in freshwater throughout their life cycle
Aquatic macrophytes ⁸	3,457 species (of unknown total)	Vascular plants in inland freshwater and brackish-water habitats

Essential for people and planet – key ecosystem services



Drinking water supply



Food provision



Nutrient cycling



Flood regulation



Carbon sequestration

Fig. 3.1 Global importance of freshwater biodiversity. Freshwater ecosystems occupy less than 1 % of the Earth's surface but support a disproportionately large share of global biodiversity and provide essential ecosystem services. ¹(Strayer & Dudgeon, 2010); ²(Albert et al., 2021); ³(Vardakas et al., 2025); ⁴(Kalkman et al., 2007); ⁵(Cumberlidge et al., 2015); ⁶(Bogan, 2008); ⁷(Strong et al., 2008); ⁸(Murphy et al., 2019).

3.1 Global freshwater ecosystem degradation

Despite their exceptional biodiversity and indispensable contribution to human well-being, freshwater ecosystems are undergoing widespread and accelerating degradation at the global scale. This deterioration is evident across multiple dimensions, including changes in ecosystem extent, water quantity, and water quality. Globally, 50 % of countries have at least one freshwater ecosystem type in a state of degradation (UNEP, 2024c). This deterioration affects all major freshwater ecosystem types, although its extent and underlying processes differ considerably among them.

Wetlands provide one of the clearest examples of this global decline. Since 1700, more than 50 % of the world's wetlands have been lost, and remaining wetlands continue to be degraded (Convention on Wetlands, 2025; Darwall et al., 2018; Tickner et al., 2020). The proportion of countries reporting deterioration of their wetlands increased from 31.8 % in 2011 to 41.5 % in 2021, while the proportion of those reporting improvements declined from 22.7 % to 14.4 % over the same period. These trends indicate that wetland degradation continues despite increasing international conservation efforts (Convention on Wetlands, 2025); **Fig. 3.2**).

Global wetland loss and condition

Wetlands provide one of the clearest examples of global decline

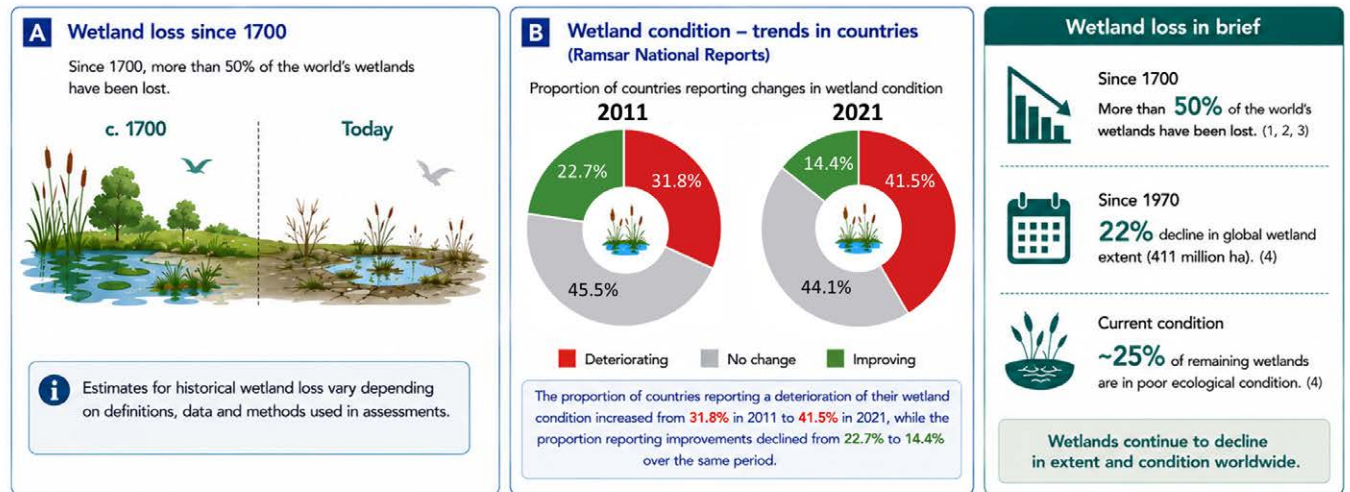


Fig. 3.2 Global wetland loss. More than half of the world's wetlands have been lost since 1700, and wetland degradation continues despite international conservation efforts. (Darwall et al., 2018; Tickner et al., 2020; Convention on Wetlands, 2025)

Groundwater ecosystems represent another major, but far less visible example of global freshwater ecosystem degradation. As the world's largest freshwater biome, groundwater ecosystems are of considerable ecological importance, yet remain among the least documented and least protected components of the freshwater realm (Saccò et al., 2024). Globally,

53 % of groundwater-dependent ecosystems occur in regions experiencing declining groundwater trends, whereas only 21 % are located within protected areas or regions subject to policies aimed at sustainable groundwater management (Rohde et al., 2024).

Similar freshwater ecosystem degradations are also reported worldwide for many lakes and rivers, including densely populated regions of Europe, East Asia, and North America (e. g., Han et al., 2024; Vörösmarty et al., 2010), further underscoring the far-reaching impacts on these ecosystems.

3.1.1 Freshwater ecosystem degradation in Europe and Germany

Europe provides a particularly well-documented example of the resulting long-term degradation of freshwater ecosystems. Despite extensive environmental legislation, monitoring, and restoration efforts, only approximately 37–39.5 % of European surface water bodies currently achieve a good ecological status under the Water Framework Directive. Thus, the majority of assessed surface waters remains below the ecological objectives established by European water policy.

Germany reflects this broader European pattern, but the ecological condition of its running waters is particularly poor. Only approximately 8 % of German rivers and streams achieve good ecological status or ecological potential (BMUV/UBA, 2022), while around one quarter of lakes meet this objective (UBA, 2025; Fig. 3.3). This pattern extends beyond the ecological

Ecological status of rivers and lakes in Germany

According to the third River Basin Management Plans under the European Water Framework Directive (status 2016–2021)

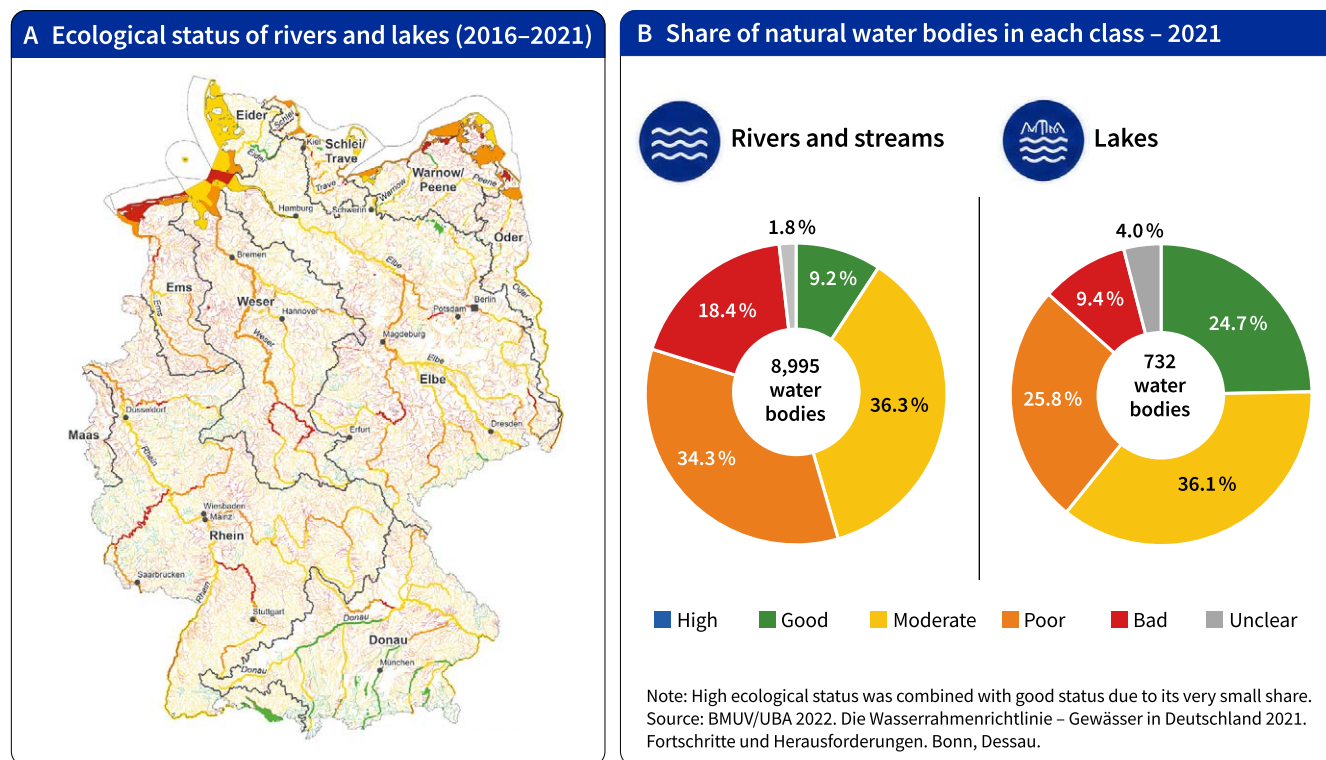


Fig. 3.3 Ecological status and ecological potential of German rivers and streams according to the third River Basin Management Plans under the European Water Framework Directive. (Copyright: German Environment Agency (UBA))

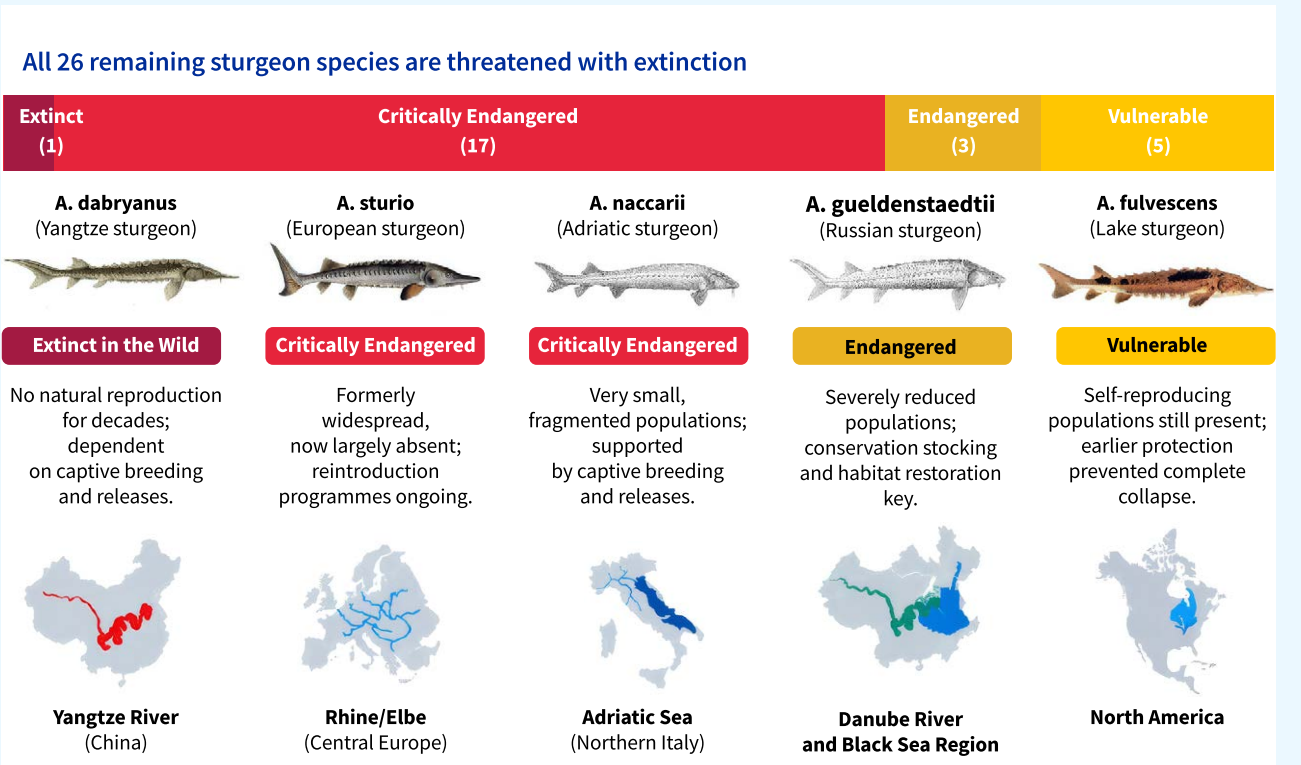
status of individual water bodies: between 60 and 75 % of freshwater and floodplain habitat types in Germany are considered threatened, indicating substantial degradation across freshwater and associated riparian ecosystems (Faktencheck Artenvielfalt, 2024).

3.2. Freshwater biodiversity loss

The widespread degradation and loss of freshwater ecosystems is mirrored by similar losses in species, community and genetic diversity, together reflecting an overall biodiversity loss. Freshwater biodiversity is declining more rapidly than biodiversity in terrestrial or marine ecosystems (Dudgeon & Strayer, 2025; Tickner et al., 2020). Approximately one quarter of assessed freshwater fishes, dragonflies, and decapod crustaceans are threatened with extinction (Sayer et al., 2025). Importantly, extinction risk varies among organism groups and regions, but the assessment demonstrates that high levels of threat are widespread across freshwater ecosystems globally. In Europe, 42 % of the 558 native freshwater fish species assessed in 2026 are classified as threatened, with a further 18 % classified as Near Threatened (European Commission. Directorate-General for Environment, 2026). European sturgeons provide a striking example of how far such declines can progress: once widespread across major European river systems, their natural populations have disappeared from most of their former range, primarily because of river fragmentation, habitat loss, and overexploitation (see [Box 2](#)).

Box 2: Global conservation status of sturgeons

Andrea Sundermann, Jörn Gessner, Peter Haase, Klement Tockner



Beyond threatened-species assessments, long-term population trends provide complementary evidence by revealing declines in abundance that may occur long before species become globally threatened or extinct. Monitored populations of migratory freshwater fishes, for example, have declined by approximately 81 % since 1970, illustrating the magnitude of population-level change in freshwater ecosystems (Piczak et al., 2024). Such population declines are ecologically important because substantial losses in abundance can reduce ecological resilience even where species remain locally present.

3.2.1 Knowledge gaps in freshwater biodiversity

Although freshwater monitoring has increased substantially over the past decades, large parts of freshwater biodiversity remain insufficiently documented, particularly microorganisms, meiofauna (e.g., water fleas), groundwater organisms, and numerous invertebrates. Groundwater ecosystems alone are estimated to harbour more than 100,000 species, many of which remain undescribed and highly endemic (Saccò et al., 2024). Even in comparatively well-studied regions such as Europe, biodiversity inventories continue to reveal previously unknown taxa (Deharveng et al., 2009; Koch et al., 2024; Lehmitz et al., 2025). Consequently,

current assessments likely capture only the visible fraction of an overall global freshwater biodiversity decline, while many species may disappear before they are even discovered or formally described (Engelisch et al., 2024).

In summary, current evidence demonstrates that freshwater ecosystems are simultaneously biodiversity hotspots, providers of essential ecosystem services, and among the most threatened ecosystems on Earth. Their biodiversity continues to decline, ecosystem conditions are deteriorating across all major freshwater ecosystem types, and the true magnitude of this decline is likely considerably greater than currently documented because of persistent monitoring and knowledge gaps. These findings identify freshwater biodiversity loss as one of the major environmental challenges of the twenty-first century and raise the question of which underlying drivers are responsible for the observed patterns of degradation and biodiversity decline.

3.3 Drivers of freshwater biodiversity loss

3.3.1 Direct and indirect drivers

The causes of freshwater biodiversity loss can be divided into indirect and direct drivers of biodiversity change. Indirect drivers comprise the underlying societal, economic, demographic, technological, institutional, and cultural factors that shape human interactions with nature. They include, for example, patterns of production and consumption, population growth, economic development, governance structures, and societal values. Although these drivers generally do not affect freshwater biodiversity directly, they determine the intensity and spatial distribution of human activities and thereby influencing biodiversity through direct drivers (Balvanera et al., 2019; Jaureguiberry et al., 2022). Direct drivers include anthropogenic or natural processes that act directly on freshwater biodiversity comprising (in descending order of importance): land-use change, pollution, direct exploitation of natural resources, invasive alien species, and climate change (Balvanera et al., 2019; Jaureguiberry et al., 2022). The importance of these drivers is also reflected in the recent global assessment of freshwater fishes, odonates and decapod crustaceans: among threatened species, “54 % are affected

by pollution, 39 % by dams and water extraction, 37 % by land-use change and associated effects from agriculture, and 28 % by invasive species and disease” (Sayer et al., 2025). Importantly, “84 % of threatened species are affected by more than one threat” (Sayer et al., 2025), highlighting that freshwater biodiversity loss commonly results from multiple, co-occurring drivers. The following paragraphs demonstrate the magnitude and ecological consequences of each of these five direct drivers using prominent, quantitatively documented examples from freshwater ecosystems.

Land-use change

Land-use change, including agricultural expansion, urbanization, river regulation, and dam construction, fundamentally alters freshwater habitats, hydrological regimes, and ecological connectivity (Reid et al., 2019; Vörösmarty et al., 2010). Streams and rivers are particularly sensitive because their ecological integrity depends on longitudinal (source to mouth), lateral (river to floodplain), and vertical (surface to subsurface and groundwater) connectivity. Globally, only 37 % of the world’s large rivers remain free-flowing (Grill et al., 2019), while, e. g., only 1 % of river floodplains in Germany are in

pristine condition (BFN, 2021) illustrating the extent of river degradation.

Pollution

Pollution is a pervasive driver of freshwater biodiversity loss. Nutrients, pesticides, pharmaceuticals, and additional chemicals enter freshwater systems from agriculture, urban areas, wastewater, and industry, affecting biodiversity directly through toxicity and indirectly through altered ecosystem processes (Sigmund et al., 2023; Sylvester et al., 2023). Agricultural pesticide contamination provides a prominent example: in more than 100 small agricultural streams in Germany, 81 % exceeded regulatory acceptable concentrations and 83 % failed to meet pesticide-related ecological targets, with pesticides identified as the dominant stressor for vulnerable freshwater insects (Liess et al., 2021). Industrial and other organic chemicals represent another major source of pollution. Across approximately 4,000 European monitoring sites, chemical contamination reached levels associated with potential chronic effects on sensitive freshwater organisms at 42 % of sites and with potential acute lethal effects at 14 %, with industrial chemicals such as tributyltin, polycyclic aromatic hydrocarbons and brominated

flame retardants among the major contributors to chemical risk (Malaj et al., 2014). Pollution affects all major freshwater ecosystems, including rivers, lakes, wetlands, and groundwater systems.

Direct exploitation of natural sources

Direct exploitation includes the removal of species, the abstraction of freshwater for agriculture, drinking-water supply, industry, and energy production as well as the exploitation of resources such as sand or gravel for constructions. The Colorado River provides a prominent example: around 70 % of catchment water supplies are used for agricultural irrigation, while the river supplies municipal and industrial water to nearly 22 million people. Intensive abstraction and river regulation have profoundly altered natural flow and habitat conditions. These changes are reflected in the native fish fauna of the lower Colorado River: of nine species historically occurring in the mainstem, five have been locally extirpated, while seven of the original nine were listed as endangered at the federal level (Mueller et al., 2005), illustrating the severe biodiversity changes associated with extensive river modification, alongside impacts from invasive species.

Invasive alien species

Invasive alien species can fundamentally alter community composition and ecosystem functioning. The zebra mussel (*Dreissena polymorpha*) provides a prominent example: following its invasion of the Hudson River, zooplankton and deep-water invertebrate biomass declined by approximately 50 % during the first decade, accompanied by major changes in fish abundance, distribution, and growth (Strayer et al., 2014). Long-term observations demonstrate that these effects are dynamic, with partial recovery of some ecosystem components over time. The Rhine River provides a well-documented European example of this restructuring. In recent decades, invertebrate and fish assemblages have increasingly included non-native species; the opening of the Rhine–Main–Danube Canal in 1992 established a major invasion corridor for Ponto-Caspian species (Black Sea area), while shipping, including ballast-water transport and hull fouling, provides an additional introduction pathway (Borcherding et al., 2011; Leuven et al., 2009). Invasive species can become highly dominant and thereby alter community structure and native biodiversity. For example, three invasive amphipod species accounted for 89 % of benthic biomass at a studied Rhine site, while long-term observations document substantial shifts in freshwater invertebrate community composition associated with

invasion (Hellmann et al., 2015, 2017; Le Hen et al., 2023).

Climate change

Climate change increasingly intensifies other drivers of freshwater ecosystems. Rising temperatures, prolonged droughts, declining discharge, and reduced dilution capacity can reinforce eutrophication and chemical pollution, while changing habitat conditions and reducing ecosystem resilience. A global analysis of more than 15,000 lakes provides a prominent example: progressive deoxygenation occurred in 83 % of investigated lakes, with approximately 55 % of the observed oxygen loss attributed to climate warming (Zhang et al., 2025). Warming strengthens thermal stratification, reduces oxygen solubility, and can increase biological oxygen demand, thereby further exacerbating oxygen depletion and other existing environmental drivers (Jane et al., 2021; Kaushal et al., 2025; Wang et al., 2025). Furthermore, climate change is predicted to lead to a severe losses of genetic diversity within freshwater species. For example, shifts in species' ranges driven by climate change can lead to a substantial losses of unique genetic lineages, as not all individuals of all species are able to track their climate niche. It is expected that the number of unique genetic lineages lost will greatly exceed those at the scale of conventionally defined species (Bálint et al., 2011).

3.3.2 Interaction among drivers

The five direct drivers presented above often interact across spatial and temporal scales, commonly resulting in even higher impacts on freshwater biodiversity. Land-use change, for example, can simultaneously alter habitats, increase water abstraction, and introduce nutrients and contaminants, while climate change can intensify the other drivers through higher temperatures, drought, and reduced discharge. Such interactions are particularly important in freshwater ecosystems because their hydrological connectivity allows drivers originating within the catchment to propagate through streams, rivers, lakes, wetlands, and groundwater systems (Jaureguiberry et al., 2022; Reid et al., 2019; Vörösmarty et al., 2010). Consequently, the ecological effects of multiple drivers may differ substantially from those expected from individual drivers considered in isolation.

A striking example of such interactions is the ecological collapse of the Oder River in 2022. A massive bloom of the brackish-water alga *Prymnesium parvum* was identified as the immediate cause of the mortality event, but its development was facilitated by a combination of elevated salinity, exceptionally low discharge, high water temperatures, and pre-existing anthropogenic drivers. The resulting toxic algal bloom caused the death of an estimated

River stress from drivers of biodiversity loss

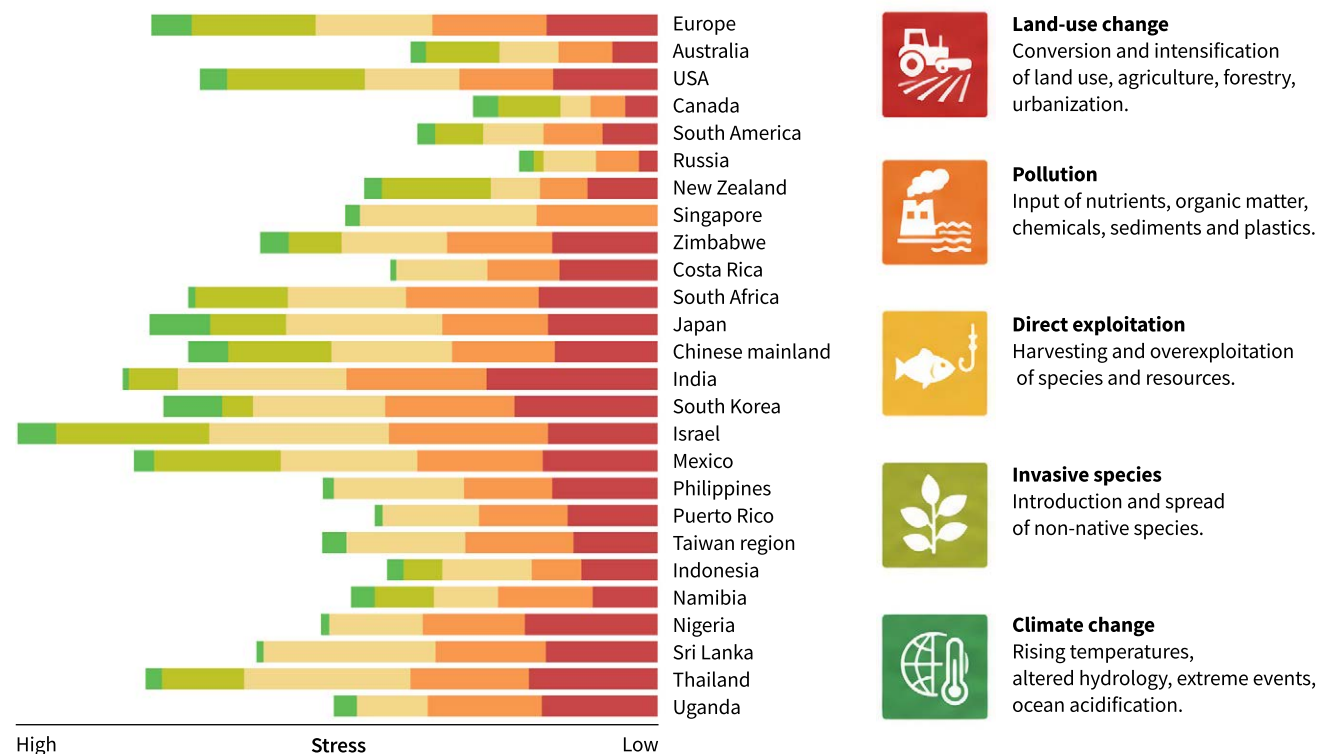


Fig. 3.4 Combined stress from major drivers of freshwater biodiversity loss across 26 global regions. Each bar represents a country or region. The total length of the bar indicates the overall cumulative stress level (high to low). Colours show the relative contribution of each driver to the total stress in that region. Freshwater ecosystems worldwide are simultaneously affected by multiple interacting stressors, although their relative importance varies among regions. (Copyright: Haase et al., 2025; modified)

3.3 million fish, 65 million bivalves, and 147 million snails, demonstrating how individually distinct drivers can combine to trigger abrupt and severe ecological consequences (Szlauer-Łukaszewska et al., 2024). Rather than representing the effect of a single driver, the Oder event illustrates how climatic, hydrological, and chemical drivers can interact and potentially push already stressed freshwater ecosystems towards critical ecological thresholds.

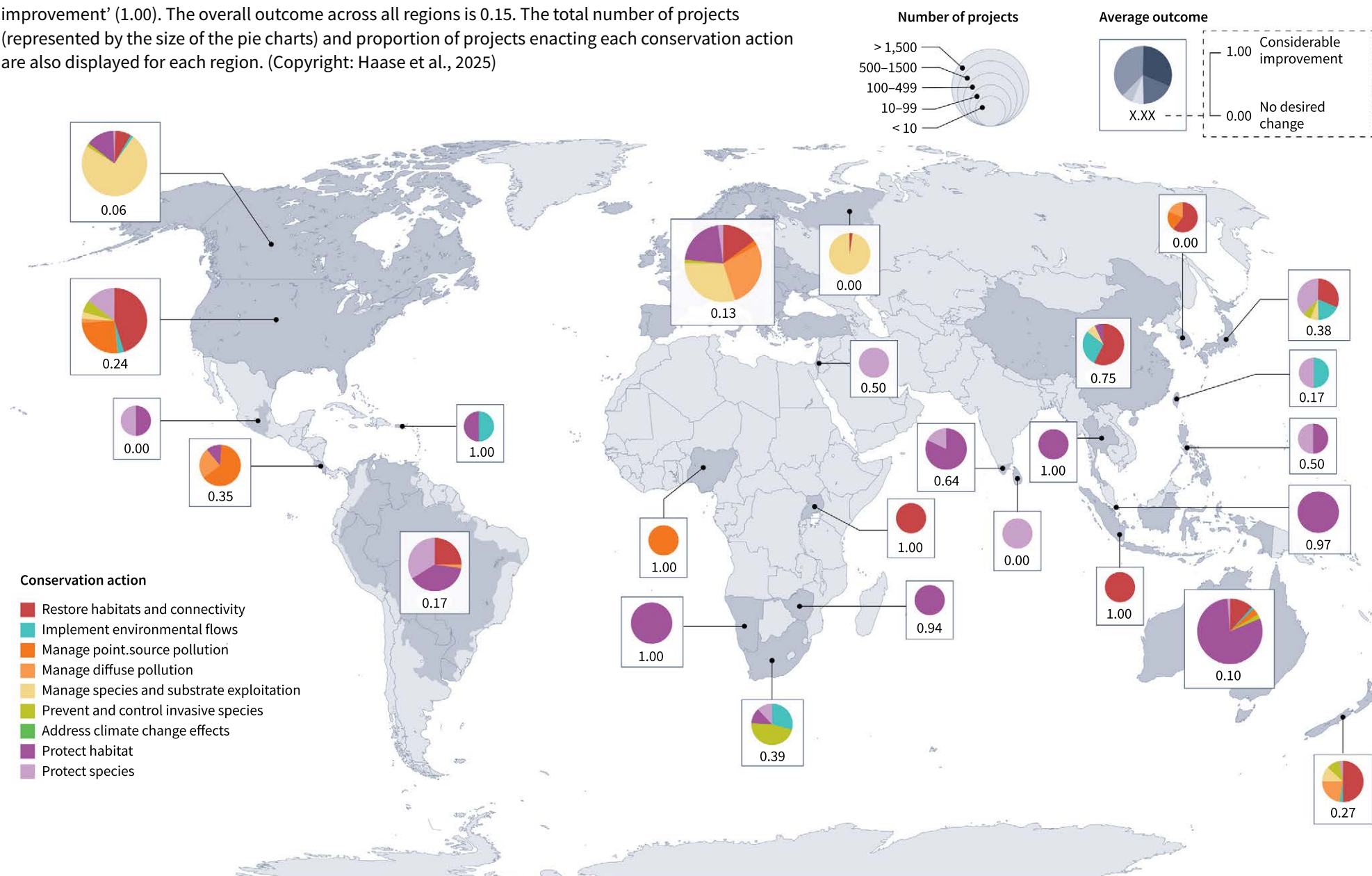
Understanding freshwater biodiversity loss therefore requires a multiple-driver perspective that links direct and indirect drivers to the environmental drivers they generate and considers their cumulative and interacting effects across entire catchments. The relative contribution of individual drivers varies among regions and ecosystem types, but their simultaneous occurrence is a global characteristic of freshwater degradation (Fig. 3.4). These interacting drivers explain why freshwater biodiversity is declining. However, the ecological significance of this decline ultimately depends on how the loss of biodiversity affects ecosystem functioning.

While, in reality, freshwater ecosystems are often affected by multiple drivers, the applied conservation actions commonly only address a single driver, which may not even be the most limiting one (Haase et al., 2025). For example, one common conservation action is habitat restoration,

which, however, rarely mitigates water pollution or the effects of climate change. Accordingly, the overall outcome of conservation actions is often comparatively low (Fig. 3.5). However, conservation actions can be significantly more successful, if all drivers are appropriately addressed.

In summary, mitigating the impacts of multiple drivers to significantly improve biodiversity in freshwater ecosystems requires a catchment-scale approach. This must combine a wide range of measures, including conventional civil-engineering methods (as a last resort for protecting buildings), nature-based solutions, and biodiversity conservation actions. Such an approach also relies on collaboration among local and regional stakeholders, including authorities, citizens, scientists, and practitioners (Haubrock et al., 2025).

Fig. 3.5 Distribution and proportion of the nine conservation action categories across 26 regions. For each region, averaged outcomes per region range from ‘no desired change’ (0.00) to ‘considerable improvement’ (1.00). The overall outcome across all regions is 0.15. The total number of projects (represented by the size of the pie charts) and proportion of projects enacting each conservation action are also displayed for each region. (Copyright: Haase et al., 2025)



3.4 Ecological consequences of biodiversity loss

Freshwater biodiversity is not only an indicator of ecosystem condition but also a fundamental determinant of ecosystem functioning. Species contribute to ecosystem functions including primary production (turning solar energy into biomass), organic matter decomposition, nutrient cycling, sediment stabilisation, and water purification (addressed in more detail in Chapter 4). Together, these functions maintain ecosystem productivity, stability, and resilience under both natural and anthropogenic disturbances (Cardinale et al., 2012; Hooper et al., 2012). Biodiversity loss can impair ecosystem functioning and, consequently, the capacity of freshwater ecosystems to provide essential services to society.

However, the relationship between biodiversity loss and declining ecosystem functioning is rarely linear. In ecosystems with substantial functional redundancy (different species contribute to a single function), initial species losses may have comparatively small effects on ecosystem functioning. Continued biodiversity loss, however, can progressively erode this redundancy and reduce ecological resilience, thereby diminishing the capacity of ecosystems to withstand disturbance and recover once

anthropogenic drivers are reduced (Fig. 3.6). Biodiversity loss may therefore affect not only how freshwater ecosystems function, but also their capacity for subsequent recovery.

How these changes in ecosystem functioning ultimately translate into altered ecosystem services and consequences for human well-being and society is addressed in Chapter 4. Here, however, the consequences for ecological resilience lead directly to another fundamental question: once freshwater ecosystems and their biodiversity have been degraded, to what extent can ecological recovery occur following driver reduction?

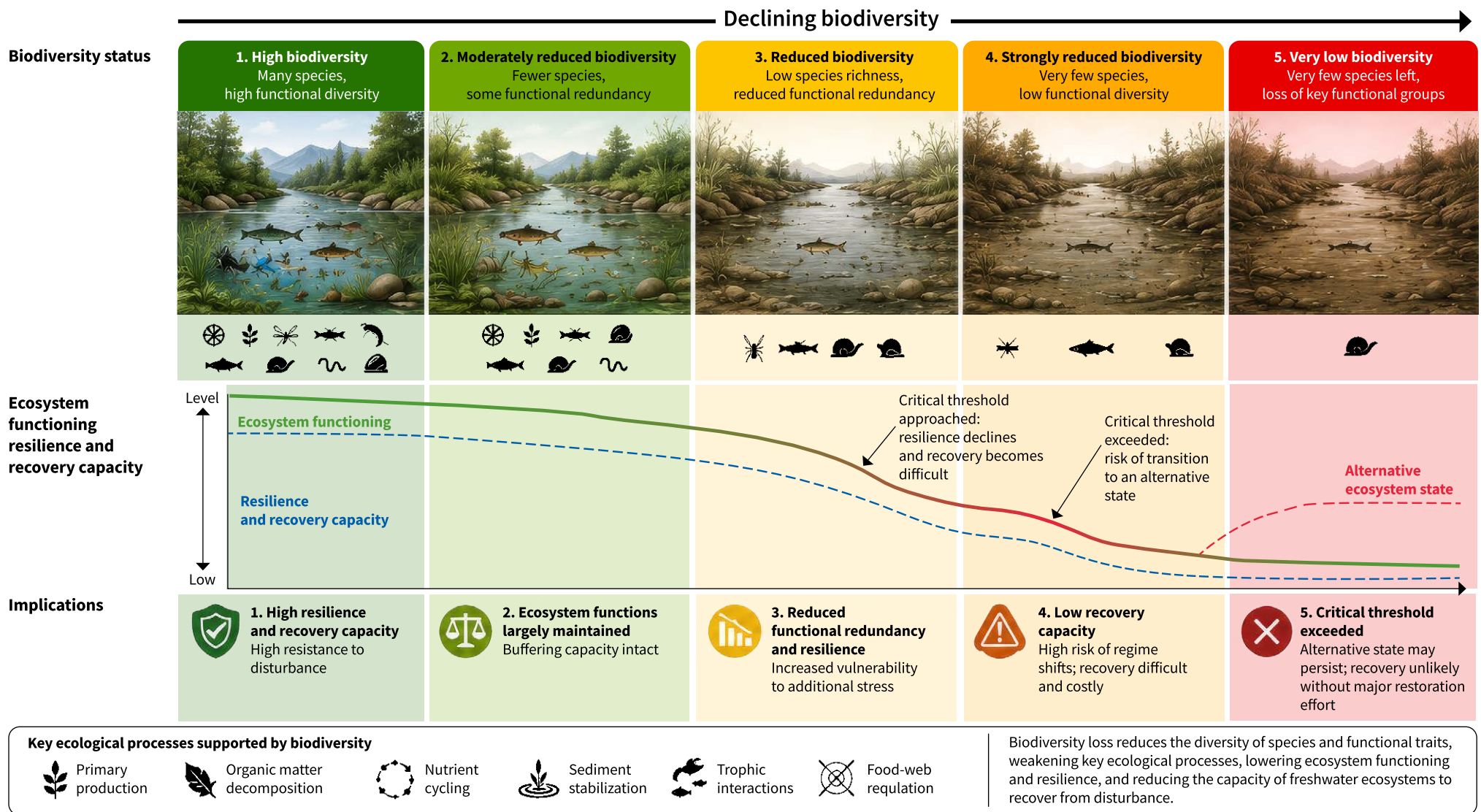


Fig. 3.6 Conceptual relationship between freshwater biodiversity, ecosystem functioning, resilience, and recovery potential. Biodiversity supports key ecosystem processes and ecological resilience, whereas continued biodiversity loss can progressively reduce ecosystem functioning and the capacity of freshwater ecosystems to withstand disturbance and recover following driver reduction.

3.5 Recovery and legacies of freshwater degradation

Although freshwater degradation has accumulated over centuries through river regulation, agricultural expansion, and industrialisation, anthropogenic drivers accelerated rapidly after the 1950s as part of the global ‘Great Acceleration’ (Steffen et al., 2015). Consequently, by the second half of the twentieth century, many freshwater ecosystems, particularly in densely populated and industrialized regions such as Europe, had reached even higher levels of degradation. In response, substantial efforts were undertaken to reduce major drivers through reductions in organic pollution (improved wastewater treatment) and acidification (e.g., reduced sulphur emissions), implementing conservation actions (e.g., species and habitat protection), and restoring ecological connectivity (e.g., removal of dams).

Such driver reductions can improve environmental conditions and facilitate biological recovery. Recovery, however, depends on habitat availability, connectivity, regional species pools, and species’ attributes (traits), while local extinctions, persistent habitat modification, invasive alien species, and changing climatic conditions may constrain or prevent the full recovery of former communities. Moreover,

degradation and recovery do not necessarily follow the same trajectory. If ecological thresholds are crossed, ecosystems may shift into alternative stable states that are maintained by internal feedback mechanisms even after the original driver has been reduced (Capon et al., 2015). For example, in shallow lakes nutrient enrichment can induce a transition from a clear-water, macrophyte-dominated state to a turbid, phytoplankton-dominated state, which may persist even after nutrient inputs are substantially reduced (Scheffer et al., 2001). This alternative state illustrates that removing or reducing the original driver does not necessarily restore the former ecological state. The reasons are manifold including insufficient nearby species pools or the occurrence of invasive alien species. Thus, while drivers can often be mitigated, complete recovery becomes increasingly unlikely.

3.5.1 Evidence for freshwater ecosystem recovery

European rivers provide strong evidence that conservation actions pay off and ecological recovery is possible. A Europe-wide analysis of 1,816 freshwater invertebrate time series from

22 countries between 1968 and 2020 found an average annual increase of 0.73 % in taxon richness, 1.17 % in abundance, and 2.4 % in functional richness, with taxon richness increasing in 70 % of the investigated time series (Haase et al., 2023). Long-term assessments similarly document gradual improvements in the ecological quality of European rivers (Sinclair et al., 2024). These trends demonstrate that reducing major anthropogenic drivers can facilitate biodiversity recovery at continental scales.

However, these improvements do not represent a complete return to former ecological conditions. Long-term trends in the ecological quality of 1,365 European rivers show that recovery has come to a halt in recent years and remains below the Water Framework Directive target of good ecological status (Sinclair et al., 2024); [Fig. 3.7](#)). The European experience therefore demonstrates both the potential and the limits of freshwater recovery: ecological conditions and biodiversity can improve substantially following driver reduction, but recovery may slow down or plateau before ecological integrity is fully restored. Achieving further recovery consequently requires sustained action to reduce all drivers to fully restore ecosystem functions – an

objective that has increasingly been incorporated into international, European, and national environmental policy.

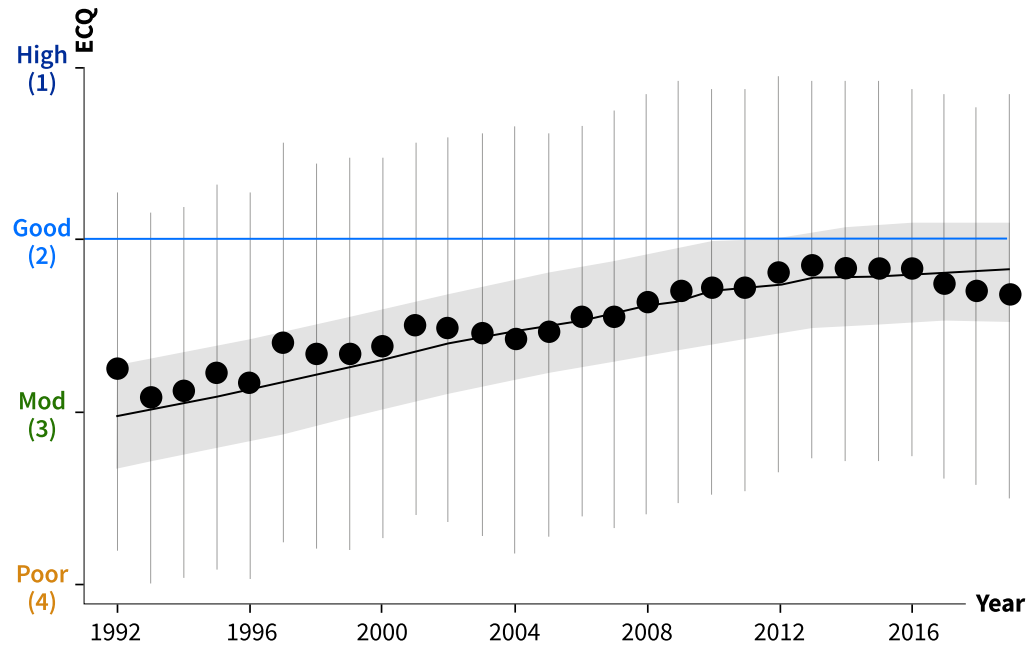


Fig. 3.7 Multidecadal trends in ecological quality classes (EQC) of 1,365 European rivers between 1992 and 2019. Ecological quality improved gradually but remained below the Water Framework Directive target of good ecological status. Black dots indicate the annual means of EQCs and grey vertical lines their standard deviations. The solid line and the grey background (95 % confidence intervals) are based on model outputs (Copyright Sinclair et al., 2024)

3.6 Freshwater policy and implementation

3.6.1 Global policy frameworks

The growing recognition of freshwater biodiversity loss and the need for ecological restoration have resulted in an increasingly comprehensive policy framework from the global to the national level (Fig. 3.8). Together, these initiatives acknowledge that safeguarding freshwater ecosystems is fundamental for biodiversity conservation, climate resilience, sustainable development, and human well-being. Although they differ in their scope and legal status, they collectively promote integrated catchment management, ecosystem restoration, sustainable water use, and improved biodiversity monitoring.

The Convention on Biological Diversity (CBD) provides the overarching global framework for biodiversity conservation. Its Kunming-Montreal Global Biodiversity Framework, adopted in 2022, represents the global policy framework for halting and reversing biodiversity loss by 2030 and explicitly includes freshwater ecosystems. Among its key commitments are the restoration of at least 30 % of degraded terrestrial, inland water, coastal, and marine ecosystems by 2030 (Target 2) and the effective conservation and management of at least 30 % of these areas

by 2030 (Target 3; the ‘30 × 30’ target). These targets provide an important global framework for substantially expanding the conservation and restoration of freshwater ecosystems and their biodiversity. Freshwater ecosystems are further addressed through the United Nations Sustainable Development Goals, particularly SDG 6 (Clean Water and Sanitation), SDG 14 (Life Below Water), and SDG 15 (Life on Land). Additional international agreements – including the Ramsar Convention on Wetlands and the Paris Agreement – recognise the central role of healthy freshwater ecosystems for climate adaptation, carbon storage, flood regulation, and sustainable development (Fig. 3.8).

3.6.2 European freshwater policy

Within Europe, freshwater protection is primarily coordinated through the Water Framework Directive (WFD), which, in 2000, introduced an integrated river catchment approach and established the objective of achieving good ecological and chemical status for all surface waters. The WFD is complemented by the Floods Directive, the Habitats and Birds Directives, the EU Biodiversity Strategy for 2030, the Nature

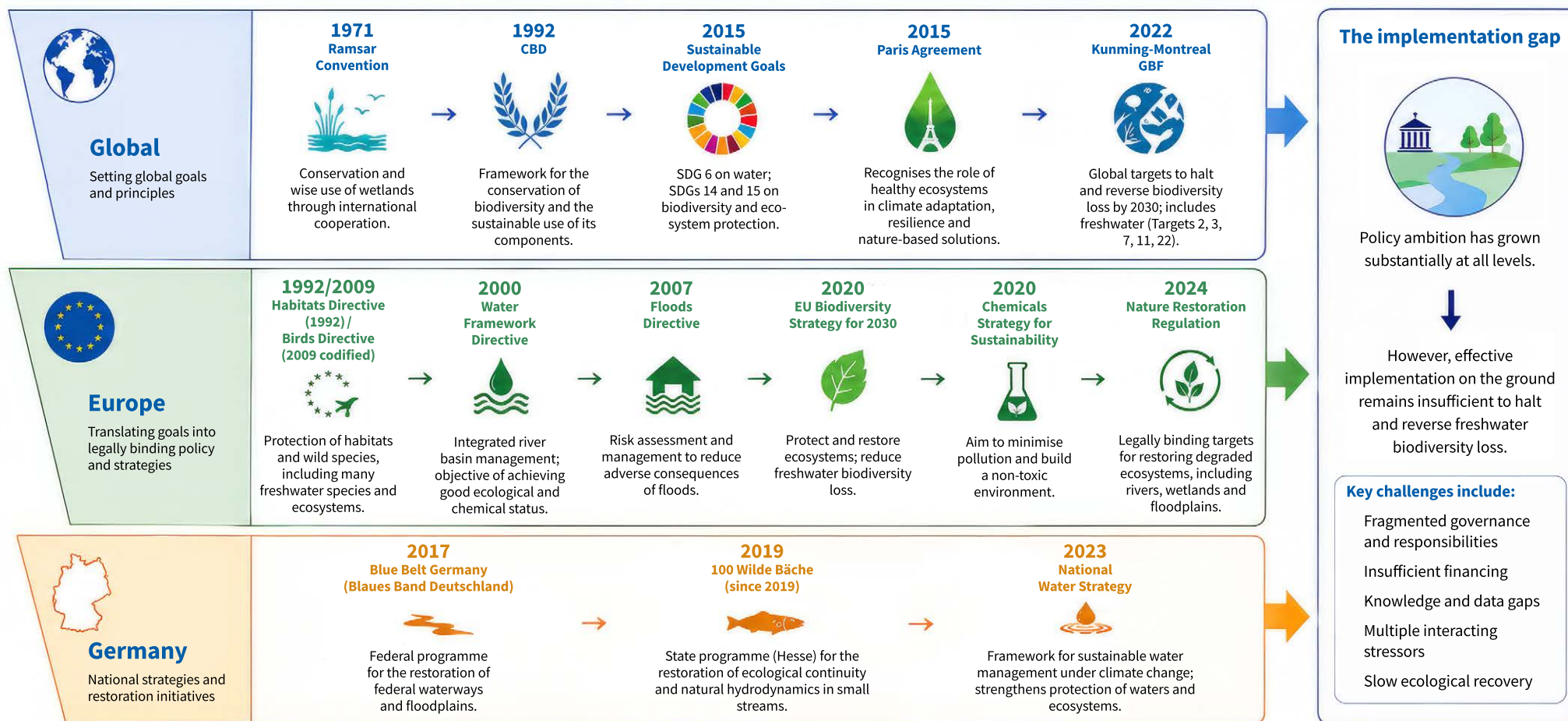
Restoration Regulation, and the Chemicals Strategy for Sustainability. Most recently, the Nature Restoration Regulation (2024) introduced binding restoration targets directly relevant to freshwater ecosystems, including the restoration of at least 25,000 km of rivers to a free-flowing state by 2030 and measures to restore river connectivity and associated floodplains. Together, these policies increasingly recognise that biodiversity conservation, water management, climate adaptation, and ecosystem restoration must be addressed in an integrated manner.

3.6.3 Freshwater policy in Germany

Germany has translated these European objectives into national strategies and restoration programmes. A central framework is the National Water Strategy, complemented by the Action Programme for Natural Climate Protection (*Aktionsprogramm natürlicher Klimaschutz*), which provides substantial federal funding for the protection and restoration of ecosystems, including the rewetting of peatlands and the restoration of rivers and floodplains. The Federal Programme Blue Belt Germany (*Blaues Band Deutschland*) specifically

International, European and national policy frameworks for freshwater biodiversity

A wide range of international, European and national agreements and strategies address freshwater biodiversity conservation, ecosystem restoration and sustainable water management. Together, they demonstrate strong political recognition of the freshwater crisis - yet effective implementation remains a major challenge.



Note: Years refer to adoption (or latest major update). Frameworks interact and mutually reinforce each other.

Fig. 3.8 Major international, European, and German policy frameworks addressing freshwater biodiversity conservation, ecosystem restoration, and sustainable water management. Together, these initiatives illustrate the growing political recognition of freshwater biodiversity loss while also highlighting the increasing complexity of freshwater governance across administrative levels.

promotes the ecological restoration of federal waterways and their floodplains, including the reconnection of rivers, banks, and floodplains and the development of a national habitat network, while the Green Belt Germany (*Grünes Band Deutschland*, part of the European Green Belt), contributes to broader ecological connectivity across the landscape. At smaller rivers and streams, complementary restoration programmes and initiatives such as 100 Wild Streams (*100 Wilde Bäche*) address hydromorphological restoration, floodplain connectivity, and the re-establishment of more natural river processes. A further key objective across these programmes is restoring longitudinal connectivity, for example by removing or modifying barriers (such as dams) to enable the movement and recolonisation of freshwater species. Together these initiatives address freshwater restoration across different spatial scales, from small streams to major rivers and their floodplains, while simultaneously supporting biodiversity, natural water retention, climate resilience, and ecosystem services.

3.6.4 Implementation gaps

Despite substantial progress in environmental legislation and restoration planning, achieving policy targets remains a major challenge. As demonstrated above, only a minority of

European rivers currently achieve good ecological status (and even the good ecological status allows for a deviation of up to 40 % percent from pristine conditions) under the Water Framework Directive. Thus, ambitious policy frameworks have not yet translated into ecological recovery at the scale and pace required to halt freshwater biodiversity loss.

This implementation gap reflects, among other factors, the complexity of freshwater degradation. Management and conservation actions often address individual drivers, whereas freshwater ecosystems are simultaneously affected by multiple interacting drivers operating across entire catchments. Establishing ambitious policy targets therefore does not ensure ecological recovery; their effectiveness depends both on successful implementation and on the ability to determine whether freshwater ecosystems actually respond to the actions taken.

This creates a second challenge. Evaluating whether ecosystems are degrading or recovering – and whether conservation actions achieve their intended outcomes – requires long-term, spatially representative and sufficiently comprehensive ecological monitoring. Current monitoring systems, however, capture only a part of freshwater ecosystems and their biodiversity. Furthermore, only a smaller fraction of environmental driver variables is sufficiently monitored.

Freshwater conservation therefore faces not only an implementation gap between political ambition and ecological outcomes but also a monitoring gap that limits our ability to quantify these outcomes comprehensively.

3.6.5 Monitoring gaps

Reliable assessment of biodiversity change and policy effectiveness requires monitoring systems capable of detecting ecological change across freshwater ecosystem types, organism groups, spatial and temporal scales, and levels of biological organisation. Existing monitoring programmes fulfil these requirements only partially and often focus on ecological status assessment rather than comprehensive biodiversity monitoring (Van Rees et al., 2025). Important spatial blind spots remain. In particular, small freshwater habitats, such as small streams and ponds, are frequently underrepresented in formal monitoring and river/lake-catchment management despite their high abundance and ecological importance, while groundwater biodiversity remains particularly poorly documented (Baattrup-Pedersen et al., 2018; Faktencheck Artenvielfalt, 2024; Koch et al., 2024).

Monitoring is also taxonomically and methodologically incomplete. Established bioassessment programmes typically focus on selected

organism groups and indicator species, and therefore do not provide a comprehensive representation of freshwater biodiversity. Even for comparatively well-established groups such as freshwater invertebrates, substantial differences exist among and even within national programmes in sampling methods, taxonomic identification, metrics, and assessment procedures. Taxonomically diverse or difficult-to-identify groups may consequently be represented only at relatively coarse taxonomic resolution or remain insufficiently captured altogether. A global survey covering 67 countries identified major gaps in freshwater biodiversity monitoring and highlighted insufficient harmonisation as an important constraint on the comparability and integration of monitoring data across larger spatial scales (Simaika et al., 2024).

An additional blind spot concerns genetic diversity. Although genetic variation represents a fundamental component of biodiversity and contributes to the adaptive potential of populations, it remains poorly integrated into biodiversity monitoring. A recent European assessment revealed substantial differences in genetic monitoring effort among countries and concluded that genetic diversity remains widely neglected in existing monitoring programmes (Pearman et al., 2024).

Emerging molecular approaches, including DNA metabarcoding and environmental DNA (eDNA), offer considerable potential to broaden taxonomic coverage and capture components of biodiversity that are difficult to assess using conventional approaches, but their systematic integration into large-scale biodiversity monitoring still requires further harmonisation and implementation (Altermatt et al., 2025).

Important limitations also arise from the temporal resolution of monitoring (Cortés-Guzmán et al., 2026). Routine sampling at fixed intervals may fail to capture short-lived disturbances and associated biological responses, including pollution pulses, droughts, heat events, or rapidly changing flow conditions. Monitoring may therefore adequately characterise average or prevailing conditions while underestimating episodic stressors that can have disproportionate ecological effects.

Finally, most established monitoring programmes primarily assess community composition and ecological status, whereas functional diversity and ecosystem processes are much less systematically monitored (Maasri et al., 2022). This limits the extent to which observed changes in biodiversity can be directly linked to changes in ecosystem functioning, resilience, and ultimately ecosystem-service provision.

Current monitoring therefore provides an indispensable but incomplete picture of freshwater biodiversity change. Conclusions about degradation and recovery depend partly on which ecosystems, organisms, levels of biodiversity, and temporal and spatial scales monitoring programmes are designed to capture. These monitoring limitations must be distinguished from the broader knowledge and implementation gaps that continue to constrain freshwater conservation and restoration.

3.7 Key messages

► **Freshwater ecosystems, including rivers, lakes, wetlands and groundwater ecosystems, are biodiversity hotspots.**

They provide a disproportionately large share of global biodiversity at limited spatial extent.

► **Freshwater ecosystems are among the most degraded ecosystems worldwide.**

They are heavily affected by habitat degradation, pollution, hydrological alteration and other anthropogenic drivers.

► **Freshwater biodiversity is declining rapidly.**

The documented losses likely underestimate the full extent of change: These declines affect populations, species, communities and genetic diversity, while major knowledge gaps remain for many organism groups, regions and freshwater ecosystem types.

► **Freshwater biodiversity loss is driven by multiple interacting drivers.**

Land-use change, pollution, direct exploitation, invasive alien species and climate

change frequently act simultaneously rather than independently. Their effects can propagate through hydrologically connected freshwater networks, making biodiversity change the result of multiple stressors operating across spatial and temporal scales.

► **Biodiversity loss reduces ecosystem functioning and resilience.**

Freshwater organisms contribute to primary production, decomposition, nutrient cycling, sediment stabilisation and water purification. The loss of species and functional redundancy can therefore impair ecosystem processes and reduce the capacity of freshwater ecosystems to withstand and recover from disturbance.

► **Freshwater ecosystems can recover when anthropogenic drivers are reduced, but recovery is often incomplete.**

European river biodiversity has partially recovered, primarily due to past conservation actions that improved water quality. However, this recovery is not yet complete, as new or persistent stressors such as pesticides, habitat destruction, disrupted

connectivity, invasive species, climate change, and legacy pollution continue to impact freshwater biodiversity.

► **Freshwater conservation has advanced, but ecological targets are still widely unmet and monitoring remains incomplete.**

International, European and national policy frameworks increasingly recognise the need to conserve and restore freshwater ecosystems, yet many freshwater ecosystems remain in poor ecological condition. Existing monitoring captures important trends but remains uneven across ecosystem types, organism groups, biodiversity dimensions, environmental drivers and short-lived disturbance events.

4 Interdependencies between water resource use and freshwater ecosystem integrity

Christiane Zarfl and Mark O. Gessner

4.1. A tripartite relationship

The multiple uses of water by humans, whether as a resource or for other ends, depend on the geophysical and geochemical setting in which freshwaters are embedded (Fig. 4.1). These settings are primarily defined by catchment characteristics such as basin geology and land-form, hydrological and temperature regimes, sediment dynamics, and water chemistry (Chapter 2). The very same drivers shape the biodiversity patterns and biogeochemical processes in ecosystems (Chapter 3), with water quantity and quality literally being the lifeblood in fresh waters. By setting habitat conditions, water quantity, quality and other environmental drivers have direct effects on biodiversity, including aquatic plants, animals and microorganisms, as well as on the biogeochemical and

other ecosystem processes facilitated by these organisms (Fig. 4.1). Other drivers produce indirect effects resulting, for example, from altered species interactions, and all of the relationships are subject to the influence of human activities and interventions.

The reciprocal influence of biodiversity and ecosystem processes on geophysical and geochemical freshwater properties (Fig. 4.1), some of which matter for human water resource use, is much less acknowledged. Yet, the ramifications are yet far-reaching. A convenient means for mapping the relationships in a cohesive framework is the concept of Ecosystem Services (Dodds et al., 2013) and, more recently, the related idea of Nature's Contributions to People

(Díaz et al., 2018). Ecosystem Services refer to the benefits that ecosystems provide to people, which includes both goods and services *sensu stricto*. Four categories are distinguished: provisioning, regulating, supporting and cultural services (Fig. 4.2). The perspective of Nature's Contributions to People (NCP) is broader, encompassing both benefits and detriments and expanding on relations between people and nature, although only three types of NCPs are distinguished: material, regulating and non-material contributions (Moreira et al., 2025).

For freshwaters, influences of biodiversity and ecosystem processes relating to water quality are of prime importance, although water levels, water volumes and water flow along with

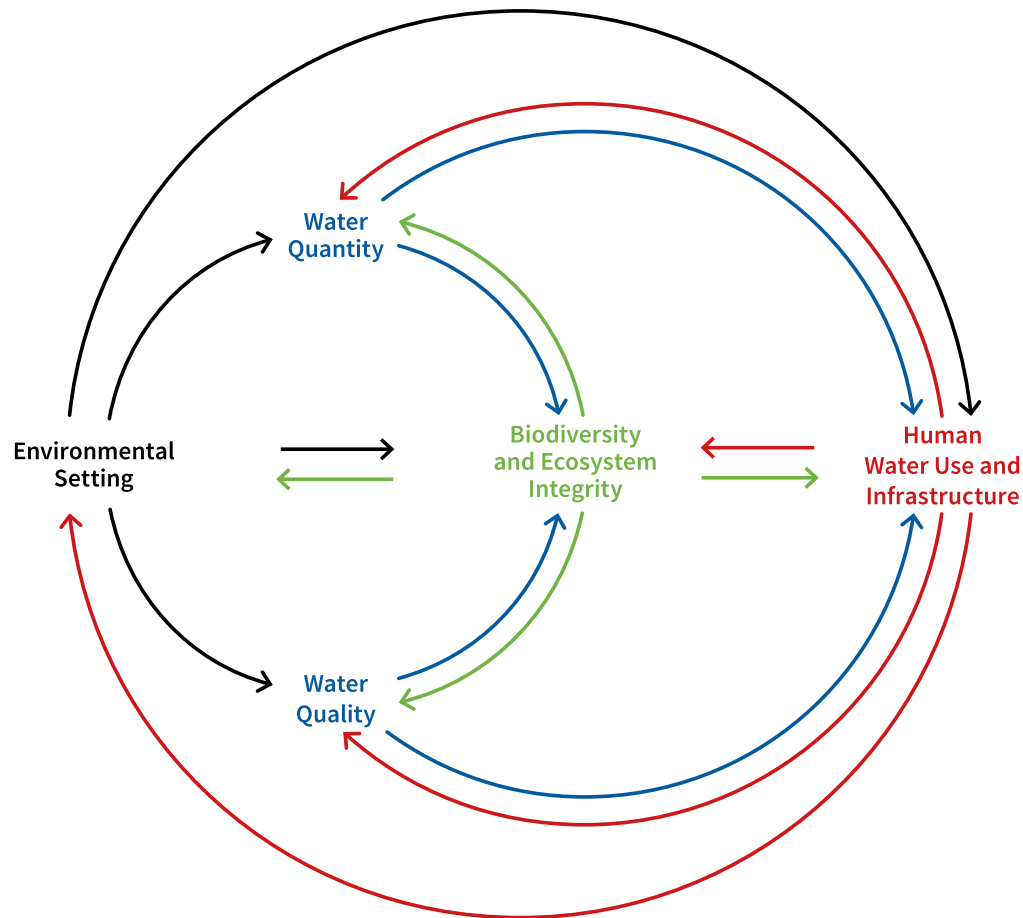


Fig. 4.1. Tripartite relationships between environmental conditions, including water quality and quantity (blue), freshwater biodiversity and ecosystems (green), and human water use and infrastructure (red).



Fig. 4.2 Overview of Ecosystem Services provided by freshwaters, emphasizing the multitude of consequences caused by changes in biodiversity and ecosystem processes for human water use, partitioned into provisioning, regulating, supporting, and cultural services.

a range of other freshwater properties can be affected as well: Engineering activities of giant tortoises (Froyd et al., 2014) and beavers (Larsen et al., 2021) create pools and impoundments. Dead wood derived from riparian vegetation has a major influence on the form of rivers channel and floodplains, as do living plants, which also protect riverbanks and lakeshores from erosion and affect not only sediment mobilisation but also patterns of material redistribution and deposition as well a hydraulic conditions (Ielpi et al., 2022; McCabe et al., 2025). Similarly, aquatic plants, animals and microbes are major players mediating nutrient fluxes (Atkinson et al., 2017; Brandt et al., 2024) and oxygen regimes (Jane et al. 2021) in freshwaters. Phytoplankton growth turns surface water turbid (Sommer et al., 2012), whereas grazers increase water transparency (Mauch et al., 2026; Sommer et al., 2012). Finally, particulate and coloured dissolved organic matter (DOM), produced mainly by plants and transformed by microbes and animals, contributes to controlling underwater light conditions. These examples illustrate the wide-ranging dependencies of environmental conditions on freshwater biodiversity and ecosystem processes.

Repercussions for human water resource use and infrastructure can be important, with biodiversity feedback to human activities and interventions being of particular interest. Such feedback can either mitigate or exacerbate the

anthropogenic impacts affecting freshwaters. A striking case of mitigation is the self-purification capacity of streams below domestic sewage outfalls. Examples of aggravation are the health risks associated with toxic algal blooms (Chorus & Welker, 2021) or with the proliferation of pathogens such as *Vibrio cholerae* in polluted waters (Levy et al., 2026). Similarly, the mass development of invasive submerged plants can block boat access to marinas (Villamagna & Murphy, 2010), and dreissenid mussels are causing massive economic damage by clogging drinking water intake pipes and colonising other underwater infrastructures (Karatayev & Burlakova, 2025).

An important intricacy of the described relationships is the fact that neither the environmental template nor biodiversity patterns and process rates are static. On the contrary, inherent variation across multiple scales is the norm, both over time (dynamics) and spatially (heterogeneity). For example, as conceived in the flood-pulse and flow-pulse concepts (Tockner et al., 2000), natural water availability and temperature vary across seasons, years and climatic zones. Freshwater species that have evolved in this landscape with periods of predictable low flow, high flow, floods and drought rely on this variability, which is instrumental in supplying resources, providing cues for migration and reproduction, facilitating dispersal and nutrient

exchange, and promoting habitat renewal (Poff et al., 1997).

The greatest impacts on freshwater biodiversity and ecosystems induced by humans tend to occur when the bounds of these natural variations are surpassed. Prominent examples are the construction of massive infrastructures in and along river courses (Grill et al., 2019; Poff et al., 2007), the release of synthetic chemicals contaminating water (Fang et al., 2019; Wilkinson et al., 2022), or novel communities created by population declines and the voluntary or inadvertent introduction and spread of invasive species (Ricciardi et al., 2013). Additionally, however, consequences for freshwater biodiversity and ecosystems can arise even when human interference operates within the natural bounds, notably when the frequency or timing of events such as droughts, flooding or mixing changes fundamentally (Poff & Zimmerman, 2010).

4.2 Competing water demands

The intricate interdependencies and competing demands for freshwater imply conflict. A fundamental dilemma is the multidimensional nature of human uses and requirements for freshwater protection, which are only partially congruent. Human water use traditionally hinges on the reliability of supplies, extractability and control achieved largely by building infrastructure. Strategies to maintain and restore ecosystem integrity, in contrast, must allow for natural structures and temporal and spatial variability (within natural bounds), along with sufficient space, longitudinal, lateral and vertical connectivity, and water availability in sufficient quantity and quality.

Managing and reconciling the conflicting demands is facilitated by treating water as a coupled social-ecological system (Chung et al., 2021; Ostrom, 2009; Pahl-Wostl, 2015), where human demands leading to altered biodiversity and ecosystem processes are constrained by stringent limits on human interventions so as to prevent minimum thresholds from being exceeded and where temporal and spatial variability mimics that of natural processes and structures. The widely adopted concept of Integrated Water Resources Management (IWRM)

(GWP, 2000) builds on such a systems perspective, aiming at balancing competing human water uses for agriculture, human consumption and industry, as well as the needs of nature. From a biodiversity and ecosystem protection perspective, however, outcomes are often unsatisfactory because the definition or application of environmental constraints is insufficient, even when economic, social and environmental repercussions are profound. The virtual disappearance of the Aral Sea, which had lost approximately 85–90 % of its lake surface area since the 1960s due to the diversion of the Amu Darya and Syr Darya rivers, for irrigation (Plotnikov et al., 2023; see [Box 3](#)), and the decline of historical river flow in the Rio Grande by an estimated 80–95 % (Blythe & Schmidt, 2018; Richter et al., 2024) are dramatic cases in point.

The key issues to resolve can be broken down into three domains: water quantity, water quality, physical structure of freshwater ecosystems and surroundings.. Water quantity or volume determines water levels and flow rate, which relate to drought stress, habitat availability and dilution capacity (see [Chapter 1](#)). Water quality encapsulates a broad range of water properties, including temperature,

odour, salinity and oxygen levels, as well as the nature and concentration of nutrients, organic matter, contaminants and pathogens. Finally, the structure of freshwaters refers to physical structures in the landscape, such as channel form, sediment distribution, riparian vegetation, and human infrastructure. These features largely govern upstream-downstream, floodplain and hyporheic connectivity in addition to providing or destroying habitat. The three domains are tightly interlinked. For example, water abstraction reduces water volumes or flow and thus the dilution capacity for pollutants. Impoundments alter water temperature and sediment transport in addition to flow regimes, and river channelisation, while improving hydraulic control for human use, changes the nature and reduces the extent and variability of aquatic habitat.

The greatest potential for conflict arises where water is scarce and human water uses are inefficient while requiring constant, predictable supplies of large volumes ([Table 4.1](#)). Consequently, water abstraction to satisfy demands for irrigation or drinking water supply directly competes with flow requirements for biodiversity protection especially in drylands and during dry periods. Similarly, the use of hydropower and

engineering measures to improve navigation tend to favour uniform hydraulic conditions, which generally involve adverse effects on freshwater biodiversity (He et al., 2024; Poff et al., 2007). River regulation for flood control and other purposes adds to these impacts by reducing both habitat complexity within river channels and the lateral connectivity with river floodplains.

Given the importance of natural variability, flow requirements for freshwater biodiversity protection cannot be stripped down to maintaining a minimum discharge (Poff, 2018). Many species depend on seasonal flow pulses, floodplain inundation, complex temperature regimes, and sediment movement (Poff & Zimmerman, 2010). Therefore, when high flows are unduly flattened, low-flow periods are prolonged, the seasonal flow distribution is altered, or active channels are disconnected from their floodplains, efficient water management can be insufficient to preserve freshwater biodiversity and ecosystem integrity even when mean annual discharge is adequate and minimum flow requirements are met (Table 4.1). Clearly, impacts due to absolute water scarcity need to be distinguished from those caused by altered variability and timing of water availability.

Table 4.1 Major conflicts arising from human water uses with consequences for freshwater biodiversity and ecosystems. The compilation is not intended to be exhaustive. Conflicts arising from teleconnections such as long-range atmospheric transport of mercury or nutrients from agricultural or industrial emissions are not considered.

Human Intervention	Purpose	Primary impact categories	Physical and chemical consequences	Biodiversity and ecosystem consequences	Conflicts
Water abstraction and diversion	Drinking water provision, crop irrigation, hydro-power generation, industrial cooling	Water quantity, structure	Water level, discharge, flow regime, hydraulics, bedload transport, suspended sediment dynamics, surface water-groundwater exchange, floodplain (horizontal) connectivity, salinisation, dilution capacity, ionic composition of water, thermal regime, groundwater depletion	Habitat loss and alteration, enhanced drought stress, community shifts, facilitation of invasive species, disease spread	Human water supply security and agricultural and energy production vs. environmental flow, aquatic habitat provision, and biodiversity protection
Dam construction and operation	Hydropower generation, crop irrigation, drinking water provision, flood control	Structure, water quantity	Longitudinal connectivity, discharge, flow regime (including hydropowering), bedload transport, suspended sediment dynamics, sediment clogging (vertical connectivity), thermal regime, oxygen regime	Habitat loss and alteration, still-water habitat gain, sediment retention, bedload deficit downstream, disruption of fish migration, hindrance of invasive species spread, greenhouse gas emissions	Human uses vs habitat provision, environmental flow, ecosystem connectivity, and biodiversity protection
Flow and water level regulation	Flood control, infrastructure protection, navigation, land reclamation, recreation	Structure	Channel form due to bank reinforcement, groins, dikes, channel incision (including by dredging), weirs; flow regime, hydraulics	Littoral, riparian and floodplain habitat alteration, habitat loss and homogenisation, community shifts, including community homogenisation	Flooding risk mitigation, navigation reliability, and land use vs. habitat integrity, lateral river connectivity, biodiversity protection, and ecosystem dynamics
Nutrient enrichment	Waste disposal, side effects of crop fertilisation	Water quality	Turbidity increase, light regime changes, oxygen depletion	Community shifts, (harmful) algal blooms, fish kills, nutrient cycling	Economic costs vs water quality protection for human use and ecosystems, agriculture vs water quality protection
Pollutant and thermal discharge	Domestic and industrial waste disposal, mining, heat dissipation, side effects of pesticide treatment	Water quality	Labile carbon and nutrient supply, oxygen depletion, warming, water and sediment contamination by metals and synthetic chemicals, thermal regime, salinisation	Widespread population declines, community shifts, nutrient cycling	Economic costs vs. water quality protection for human use and ecosystems; economic costs vs public health (pathogens, parasites, toxins)
Resource exploitation	Gravel extraction, mining, overfishing, floodplain forestry	Structure, water quality	Channel incision, floodplain decoupling, water and sediment contamination	Community shifts, (fish) population declines	Economic profit vs biodiversity and ecosystem protection

Box 3: Re-inundating dried up inland waters – lessons from the Aral Sea

Klement Tockner, Mark O. Gessner, Christiane Zarfl

The drying of lakes and wetlands due to water abstraction, river regulation and climate change affects not only water availability and biodiversity but also the carbon cycle, human health and local economies. Drying can turn aquatic ecosystems into important sources of CO₂: Accounting for exposed sediments increases the estimated global CO₂ emissions from inland waters by about 6 % (Keller et al., 2020).

The Aral Sea is an extreme example. Once the world's fourth-largest inland water body, covering about 65,000 km² in 1960, the lake has lost around 90 % of its water volume and surface area, largely because water from two rivers, the Amu Darya and Syr Darya rivers, were diverted for irrigation. As a consequence, around 60,000 km² of former lakebed became the Aralkum Desert, from which an estimated 15–75 million tonnes of salt, dust and sand are mobilised annually, affecting human health, agriculture and ecosystems (Micklin, 2007; World Bank, 2024).

The consequences of converting the Aral Sea to a desert extend to the global carbon cycle.

Since 1960, the exposed sediments have released an estimated 204±53 million tonnes of carbon, while vegetation colonising the former lakebed has compensated for less than 1 % of these losses. Reflooding could potentially prevent the release of a further 165±13 million tonnes of carbon. Therefore, re-inundating a sizeable part of the former lake is not only an important ecological and humanitarian mission but could also be a valuable contribution to climate mitigation.

Importantly, the dramatic case of the Aral Sea shows that partial restoration can indeed work. Following construction of the Kok-Aral Dam in 2005 and improved flow management of the Syr Darya, the water volume of the North Aral Sea has increased by about 68 %. The water level rose accordingly from approximately 38 to 42 m, salinity fell by around 50 %, and fish production more than tripled. Today, annual catches reach approximately 6,000–20,000 tonnes, demonstrating that hydrological restoration has positive ramifications also for biodiversity, ecosystem processes and livelihoods (World Bank, 2026).

Restoration of the Aral Sea to its historical extent is currently unrealistic given the continued human water demands upstream. However, a strategy combining continued recovery of the northern part and delta wetlands by further increasing environmental flows of the tributary rivers, promoting sustainable irrigation upstream and stabilising exposed sediments could bear fruit in the long run. It is promising that basin-wide cooperation is coordinated through the International Fund for Saving the Aral Sea (IFAS) established by the five Central Asian states.

The broader lesson extends far beyond the Aral Sea. Preventing desiccation and re-inundating lakes and wetlands is not entirely unfeasible, even when water demands are high, and can simultaneously maintain or restore biodiversity, promote carbon storage, reduce human health risks, support livelihoods and strengthen climate resilience. Thus, re-inundating and rewetting dried-up inland waters is not only an investment in biodiversity and ecosystem restoration, but also in people and economies.

4.3 Consequences of human water uses for freshwater biodiversity and ecosystems

The challenges arising from the dual exigency of ensuring human water security and halting freshwater biodiversity and ecosystem decline are vast. They are also asymmetric. Investments in infrastructure designed to regulate water levels, volumes and flow for the benefit of human water uses entail adverse and sometimes devastating effects on freshwater biodiversity and ecosystem processes (Vörösmarty et al., 2010). Dams for cropland irrigation, drinking-water provision and hydropower generation; flood-control measures; a suite of river engineering works meant to improve navigation; large-scale water diversions; lake water-level regulation; groundwater extraction; or extensive wetland drainage are prominent examples (Table 4.1). In contrast, improvements of water quality by wastewater treatment plants benefit human water security and biodiversity and ecosystems alike. Reservoirs can provide new still-water habitat for freshwater biodiversity to the extent that some have become precious nature conservation areas (Cantonati et al., 2020; Qu et al., 2023). At least four areas for action to alleviate impacts of human water uses on freshwater biodiversity and ecosystem deserve particular attention:

4.3.1 Water abstraction and diversion

Water abstraction is inevitable to drinking-water supply, crop irrigation, and industrial cooling (Vörösmarty et al., 2010; Mueller et al., 2021). The consequences of this and other human activities are manifest on all continents. As a global analysis has shown, river discharge during periods of low flow, which is particularly critical, has strongly declined across vast areas on all continents (Fig. 4.3). By far the largest user is agriculture, which accounts for as much as ~72 % of the global total water withdrawals, followed by industry with ~15 % and domestic or municipal uses with ~13 % (Ferguson & McIntosh, 2025). In total, approximately 4,000–4,200 km³ of freshwater are globally withdrawn each year. This represents an estimated 10 % of the renewable or accessible freshwater resources (FAO, 2025) and slightly more than 10 % of the global annual river discharge (Collins et al., 2024). Regional differences are large, however, with the proportions being substantially higher especially in water-stressed river basins (UNESCO WWAP, 2025). Furthermore, in many densely populated regions, more than half of the river discharge is appropriated for human use, and about one quarter of the world's rivers regularly

run dry before reaching the ocean (Best, 2019; Mueller et al., 2021). The extent of water abstraction and its hydrological consequences is well illustrated by the famous example of the Aral Sea, once the fourth-largest lake in the world (see Box 3). The lake almost completely dried up within five decades after large-scale water abstraction began and before measures were taken to preserve at least small pockets.

Given the constant replenishment by downstream flow, rivers are still the most sustainable source for water abstraction, as long as withdrawal rates remain moderate. At the other end of the spectrum is fossil groundwater in deep confined aquifers (Ferguson & McIntosh, 2025). Thousands to millions of years old, these mostly non-renewable freshwater stores are being exploited at large scale as well (Jasechko & Perrone, 2021). The resulting rapid depletion is well documented, for example in California (Liu et al., 2022), India (Bhattarai et al., 2023) and Spain (Junquera et al., 2025), but also occurs in other global regions. This includes the Saharan desert which is home to the single largest fossil groundwater reservoir in the world, the Nubian Sandstone Aquifer System (Abotalib et al., 2016). Although partially motivated by the need to

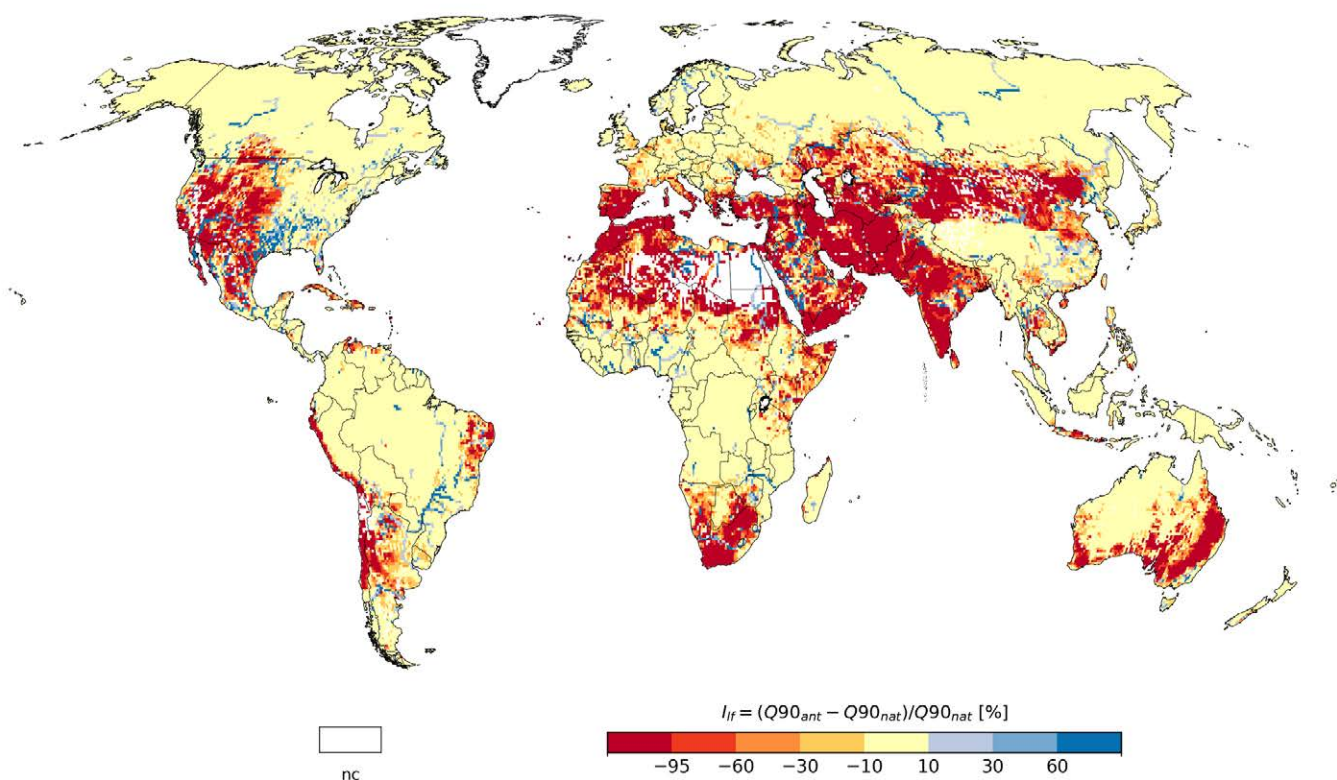


Fig. 4.3 Proportional alteration of natural monthly low flow (I_{lf}) resulting from human water uses and water retention in reservoirs between 1990 and 2019, as computed by the global hydrological model WaterGAP 2.3. $I_{lf} = (Q90_{ant} - Q90_{nat}) / Q90_{nat}$, where $Q90_{nat}$ is the natural streamflow exceeded 90 % of the time and $Q90_{ant}$ is the flow exceeded 90 % of the time when anthropogenic water use and retention are considered. White areas on the map (nc) refer to regions where $Q90_{nat} < 0.01$ l/s.

supply drinking water, the largest groundwater abstractions are mostly driven by agriculture, which, incidentally, is also the root cause of large-scale wetland drainage to gain arable land. Where neither river discharge nor groundwater stores are sufficient to meet human demands, inter-basin water transfers have been considered and implemented (Idrees et al., 2025), the largest of which is the South-to-North Water Diversion Project in China, extending to nearly 1500 km along each of its three routes (Zhuang et al., 2025). In addition to the source, these large-scale interventions affect not only the hydrology of the source but also of the recipient system.

Evidently, water abstraction and diversion systematically alter the hydrological characteristics of rivers, lakes, and ground water. River flow is reduced, the flow regime altered, and lake levels and groundwater tables drop. These effects can be interconnected. Springs can dry up when groundwater tables fall, reducing baseflow in streams and rivers even when no water is directly withdrawn. Conversely, groundwater recharge is reduced in semi-arid regions or during dry periods when river water is withdrawn from losing reaches where the river stage is above the groundwater level. In addition, there are indirect physical and chemical effects of water abstraction and diversion such as altered thermal regimes, including increased

water temperatures and temperature amplitudes, both of which are exacerbated by return flows of water used for cooling purposes (Du et al., 2026; Kirillin et al., 2013). Similarly, reduced flow results in the concentration of pollutants (see below). These direct and indirect changes caused by water abstraction can fragment river networks, weaken surface water–groundwater and river–floodplain connectivity. As a result, aquatic habitat area is reduced and conditions of the remaining habitat changed. This includes increased risks of toxic effects (Hartmann et al., 2021) – both due to concentration effects and because irrigation can mobilise agrochemicals applied to soils, which can leach into ground and surface waters (Fišer et al., 2025).

All of these physical and chemical impacts on freshwaters have repercussions for biodiversity and ecosystem processes, with periods of low flow being particularly important for freshwater communities. Specifically, as found in a meta-analysis, water abstraction significantly reduces aquatic invertebrate richness and abundance, promotes algal growth, and changes rates of organic-matter decomposition and whole-ecosystem metabolism in streams (Sabater et al., 2018). Fish communities are also affected (Albert et al., 2021). Inter-basin water transfers further compound risks by connecting previously isolated communities, facilitating invasions, and altering local species richness

patterns. For example, a model developed for Peninsular India indicates that an archetypal transfer scheme would reduce system-wide freshwater fish biodiversity and increase community similarity among basins (Campbell Grant, 2012), a phenomenon known as faunal homogenisation. At the global scale, freshwater vertebrate populations have declined by about 81 % since 1970, with water abstraction identified as one of the primary drivers (Albert et al., 2021).

4.3.2 Dam construction and operation

Dam construction and operation are similarly pervasive interventions in surface waters as water abstraction (Zarfl et al., 2015). Dams are also built for partly similar reasons, namely to ensure human water security for drinking water and irrigation. In addition, a large fraction of both large and small dams serves to generate electricity. Run-of-river hydropower takes advantage of the kinetic energy of flowing water. Accordingly, the water storage volume behind those dams is small. Storage hydropower, in contrast, relies on tall dams that hold back large volumes of water in reservoirs. Turbines at the bottom of the dam are driven by hydrostatic pressure when water is released from the reservoir. Pumped-storage hydropower is a mode of operation for storage hydropower where a second reservoir at the bottom of the dam retains

the turbinated water. From there, it is pumped back up at times when electricity is cheap to be later released again during peak electricity demands and correspondingly high prices on the energy market. Ordinary and pumped storage hydropower can result in hydropeaking where large and frequent discharge fluctuations occur in the channel receiving the turbinated water. Flood control can be an additional purpose for building dams, and there are numerous other reasons, ranging from flow and water-level regulation to navigation, and from stormwater to aquaculture and fire-fighting ponds.

The scale of human interventions in rivers by constructing and operating dams, weirs and other barriers is remarkable. At present, more than half of the world's large rivers are fragmented by dams and similar transversal structures in river channels; this proportion could rise to >90 % with the expansion of hydropower currently in the planning (Grill et al., 2019; He et al., 2021). In a detailed study across 36 European countries, Belletti et al. (2020) identified at least 1.2 million barriers, 0.74 per river kilometre. Dams fragment river networks by disrupting longitudinal connectivity (Belletti et al., 2020; Grill et al., 2019), disconnect river channels from their floodplains (lateral connectivity), and suppress natural high-flow events (Bejarano et al., 2017; Poff & Zimmerman, 2010).

When dams retain water for agricultural production, households, or industry, downstream flow is greatly reduced and often ceases completely. Ensuring minimal flow in the downstream reaches is a first step to provide some level of protection of freshwater biodiversity and ecosystems. This is not sufficient, however, because numerous other characteristics of natural flow regimes are impaired as well. For example, in their meta-analysis involving 47 comparisons, Sabater et al. (2018) found that flow variability was reduced by 33–96 % relative to upstream control sites, adding to the observed 42 % reduction in discharge (range: 6–98 %). Similarly, a large-scale synthesis on rivers worldwide showed that on average, flow variability was reduced by more than 40 % and high-flow frequency by more than 50 % (Poff and Zimmerman, 2010). Where hydropower operations involve hydropeaking, hydrological changes are particularly pronounced and cause rapid shifts between flow spikes and periods of low or no flow. The scope is illustrated by a European-wide assessment, where stream reaches affected by hydropeaking were found to experience flow fluctuations of up to $5\text{--}10\text{ m}^3\text{ s}^{-1}$ within 15–30 minutes, and daily flow amplitudes often exceeded 50–100 % of mean summer discharge (Bejarano et al., 2017).

Upstream of dams, the impoundment of water creates artificial lakes. Collectively, these

reservoirs now cover about 10 % of the total lake area globally ($0.26 \times 10^6\text{ km}^2$ vs $2.67 \times 10^6\text{ km}^2$), which includes the world's large saline lakes, and a significant part of the total freshwater volume contained in lakes (Messenger et al., 2016). Meant to buffer flow variation, water levels in reservoirs fluctuate strongly, the average being four times larger than in natural lakes (Cooley et al., 2021). This is a rare example where human intervention has increased rather than decreased natural dynamics of ecosystems. Similarly, the operation of reservoirs used for generating hydropower has greatly increased dynamics at frequencies and time scales (sub-daily) not previously experienced in the affected rivers and streams (except for glacial streams experiencing increased daytime flow during ice melt).

Temperature regimes are also fundamentally altered both downstream and upstream of dams. Upstream, impoundments convert formerly flowing river reaches into standing-water bodies prone to thermal stratification, with a warm surface layer and a cold bottom layer being established during the warm season (Cheng et al., 2020). If sufficient nutrient supplies promote abundant algal growth, the surface water turns turbid and the deep water can experience oxygen depletion (see below), an effect that can also be induced by organic matter from terrestrial vegetation when reservoirs are flooded.

Downstream, the release of water from specific depths of the reservoir, commonly cold hypolimnetic releases from deep outlets, decouples river temperatures from natural seasonal and diel cycles, often resulting in pronounced thermal dampening, delayed warming in spring, and prolonged cold conditions in summer (Olden & Naiman, 2010). Surface water release has the opposite effect. Hydropeaking not only produces flow spikes but also induces sudden increases or drops in temperature referred to as thermopeaking (Carolli et al., 2012).

In addition to slowing water movement and changing temperature regimes, dams trap sediment. The effects can be substantial. For example, based on satellite-derived observations from 414 major rivers, Dethier et al. (2022) estimated that sediment trapping by dams in the global hydrologic north had reduced fluxes of suspended sediment to approximately 49 % of pre-dam levels. The retention of bedload is even more important. Downstream impacts include channel incision caused by a deficit in sediment supply, which also results in a disconnection between river channels and their floodplains (Kondolf, 1997).

A large body of evidence shows that the numerous changes imposed by dams have strong impacts on aquatic habitats and often impair biodiversity (Ding et al., 2026; He et al., 2024).

Fragmentation blocks longitudinal connectivity, disrupting migration routes, and thus compromise life cycles of migratory species such as sturgeons (see [Box 2](#)), salmon (Goodman et al., 2026) and eels, but also of some crustaceans and gastropods (Chan et al., 2025). The meta-analysis by Sabater et al. (2018) showed that reductions in discharge and flow variability were associated with an average 3.3-fold increase in algal biomass and significant declines in the richness and abundance of aquatic insects and other invertebrates. The response ratios of 0.34–0.60 found in this meta-analysis demonstrates how relatively modest changes in flow conditions can translate into large biological responses (Sabater et al., 2018).

In another meta-analysis assessing impacts of hydropеaking, physical changes in affected reaches were also related to significant reductions in invertebrate density and richness. Rivers experiencing pronounced hydropеaking and dampened seasonal floods exhibit lower invertebrate abundances, reduced drift and insect emergence rates, and shifts toward generalist taxa, alongside declines in fish adapted to flowing water and the loss of species dependent on floodplains (Bejarano et al., 2017; Poff & Zimmerman, 2010). The extent of the effects sizes corresponded to a 30–50 % lower abundance and a 10–30 % lower taxon richness compared to unaffected sites (Bejarano et al., 2017).

Stranding mortality amongst juvenile fish was also increased.

Lastly, the multiple alterations of river temperature conditions induced by dams can collectively shift the thermal regime of dammed rivers outside the tolerance ranges of native species, interfere with phenological cues, and favour thermally tolerant, often non-native species, thereby restructuring freshwater communities due to temperature effects alone.

Clearly, multiple trophic levels and taxonomic groups are adversely affected by dams and the induced direct and indirect changes in physical conditions. At the global scale, declines in native fish richness and taxa specifically adapted to hydraulic conditions in riverine habitats were also found to be tightly correlated with reductions in flow variability and high-flow frequency (Jones, 2014; Poff & Zimmerman, 2010). Thus, it comes as no surprise that recent global analyses indicate that hydropower dams increase the extinction risk for freshwater populations, with the great majority of the species that were assessed shifting towards higher threat categories where impacts of dams are strong (Ding et al., 2026). With many dams to be decommissioned in the near future, dam removal and that of other transversal barriers in rivers, both tall and small, is beginning to become practice (see [Box 4](#)). The ramifications for freshwater

biodiversity and ecosystems are expected to be highly beneficial, despite major immediate disturbances both upstream and downstream.

Box 4: Dam removal

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The benefits provided by dams to humans, notably hydropower, water supply and flood control, come with substantial costs for freshwater biodiversity and ecosystems. In recognition of these costs caused by river fragmentation, massively altered hydrological, sediment, and temperature regimes, and which lead to blocked routes for migratory species, habitat loss and degradation, biodiversity decline and altered ecosystem processes, policy has increasingly moved towards the option of dam removal, rather than construction, for more than two decades (Poff & Hart, 2002). Strategic dam removal can indeed lead to significant advances towards restoring river connectivity, sediment and thermal regimes, freshwater biodiversity and ecosystem processes (McCaffery et al., 2024). Economic considerations related to aging dam infrastructure and high maintenance and safety costs, combined with sediment accumulation that reduces water storage capacity (Habel et al., 2020) have been instrumental in this development. At the same time, energy portfolios are changing, with small hydropower plants becoming obsolete or economically less lucrative. Therefore, dam removal

has been most common for small structures (<2.5 m height), but large-scale removal strategies are also gaining ground in certain regions such as the USA and France (McCaffery et al., 2024).

Removing a dam restores longitudinal connectivity, allowing fish and invertebrates to recolonise upstream reaches and creating the potential to re-establish migratory life histories of species, as has been the case in one of the best known examples of dam removal – the Elwha River located within the Olympic National Park in the northwestern USA (Fraik et al., 2021; Jones et al., 2022). Following the removal of four hydroelectric power plants on the Klamath River in the largest dam removal project to date, thousands of salmon recolonised nearly 90 % of their original habitat along 345 km of river length within two migratory seasons (Goodman et al., 2026). Dam removal also reactivates natural sediment and nutrient transport, re-engages geomorphic processes such as channel adjustment and floodplain reconnection, can improve water quality by eliminating oxygen depletion in stratified reservoirs, and reduces emissions

of potent greenhouse gases such as methane. While dam removal does cause short-term disturbances, downstream particularly by triggering the release of large pulses of mobilised fine sediment, long-term outcomes typically include enhanced biodiversity, re-established ecosystem processes characteristic of natural rivers, and the recovery of ecosystem services. Details evidently depend on the specific local conditions such as dam size and age, sediment load, removal strategy (e. g., staged vs rapid), and catchment characteristics. Upstream disturbances caused by the drainage of reservoirs will require long periods of terrestrial landscape succession.

Decisions to maintain or remove dams involve trade-offs between socio-economic benefits (energy production, water security, flood protection) and environmental costs (biodiversity loss, habitat degradation, impairment of ecosystem processes and services). Increasingly, these decisions are informed by basin-scale assessments of river connectivity, cumulative impacts, and restoration potential, as well as cost-benefit analyses incorporating ecological values and alternative for water

supply and energy production. Where removal is considered not to be feasible, mitigation measures such as fish passages, environmental flow regimes, and carefully targeted sediment management can partially offset impacts of dams, although natural river dynamics can barely be fully restored with large dams remaining in place. Conversely, dam removal will fail to re-establish natural river dynamics and biodiversity if any other drivers of river degradation persists (Grizzetti et al., 2017; Schinegger et al., 2012). This includes water abstraction, nutrient or contaminant inputs, channelisation, land use in riparian areas and on floodplains, climate-driven flow alterations (see [Chapter 4.3](#)). Thus, dam removal is not a silver bullet for river restoration but an important component that must be embedded in a broad multi-stressor and multi-scale strategy that seeks to re-establish key processes and structures within the constraints imposed by persistent human influences.

4.3.3 Flow and water level regulation for flood protection and navigation

Systematic large-scale flow and water level regulations in North America and Europe since the mid-19th century have massively altered the areal extent of wetlands but also the size of lakes and the water level of both (Fluet-Chouinard et al., 2023). Likewise, in rivers and streams, the magnitude, frequency, duration, timing, and rate of change of flow have been altered (Poff et al., 1997; Poff & Zimmerman, 2010), as well as the shape of river channels (Belletti et al., 2015). In contrast to dams, where hydrological and geomorphological river modifications are accepted side effects accompanying the use of built infrastructure, river regulation ultimately aims to alter the shape of river channels and the flow of water therein. The two main purposes of a range of engineering measures taken to this effect are flood control and navigation.

Flood control foremost involves the construction of dikes and embankments that isolate active river channels from riparian zones and floodplains. These measures are to protect built infrastructure, including cities and industrial facilities, and support agricultural practices that are out of phase with seasonal inundation patterns. Engineering river channels for navigation also involves bank reinforcements and groins, as well as weirs to regulate water levels, and the

removal of snags and other deadwood. These are secondary interventions, however, the main ones being channel straightening to shorten travel distances and deepening of rivers to maximise the loaded draught of barges. In addition, channel incision by natural vertical erosion occurs when channels are straightened and narrowed. In all cases, downstream water flow is accelerated, and river channel form and hydraulic patterns are fundamentally reshaped.

By effectively disconnecting rivers from their floodplains, flood-control measures and river channelisation modify habitat conditions for fish and invertebrates, reduce habitat heterogeneity and cut off spawning grounds, nursery areas, and refuges for various species during floods (Jones, 2014; Tockner & Stanford, 2002). Stranding risks for fish and invertebrates increase. Flow-tolerant species, often non-native, are favoured over taxa adapted to natural flow patterns (Jones, 2014). In addition to reducing invertebrate richness and abundance, flow regulation promotes algal biomass and changes organic-matter decomposition and ecosystem metabolism (Sabater et al., 2018). At larger scales, flow regulation is consistently associated with reduced richness of native fish, altered invertebrate community composition and simplified food-web structure, especially in rivers where natural flow variability is key in shaping river communities (Jones, 2014; Poff &

Zimmerman, 2010). Further evidence suggests that river regulation contributes to the global 81 % decline observed in freshwater vertebrate populations (Albert et al., 2021). Thus, although the severity of consequences for biodiversity depends on both the degree of flow alteration and the life-history traits of resident species, channel engineering emerges as an important driver of freshwater biodiversity impairment in regulated rivers (Albert et al., 2021; Poff & Zimmerman, 2010).

Beyond the impacts of river engineering on flow and channel form, the direct effects of ship traffic on navigable rivers are being increasingly recognised (Kucera-Hirzinger et al., 2009; Sexton et al., 2024; Verdonschot, 2012). Ship-induced waves and propeller jets create near-shore turbulence, increase shear stress at the river bottom, and enhance sediment resuspension, leading to scouring, bank erosion, and reduced shoreline stability (Gabel et al., 2017; Scherbaum et al., 2025; Verdonschot, 2012). Severity of the impacts depends on vessel size, speed and traffic frequency – in addition to channel form and the type and degree of bank reinforcement. Field and flume studies have shown that large cargo vessels and push-tows can generate wave heights of 0.3–0.5 m and near-shore current velocities exceeding 0.5–0.7 m/s. Turbulence levels are several times higher than under natural wind-wave conditions.

These physical changes degrade or eliminate shallow, low-energy habitats that are critical for aquatic plants to establish and as a refuge, spawning and nursery grounds for fish and invertebrates. As a result, opportunistic species tolerating a wide range of flow conditions are favoured over flow-sensitive specialists (Gabel et al., 2017; Kucera-Hirzinger et al., 2009; Scherbaum et al., 2025; Sexton et al., 2024). Accordingly, experimental exposure to turbulent conditions simulating ship traffic reduced survival of larval and juvenile fish preferring fast-flowing water by 20–40 % and slowed the growth of early life stages (Gabel et al., 2017; Scherbaum et al., 2025). Plant cover and biomass was also lower (Gabel et al., 2017; Scherbaum et al., 2025). Both of these experimental results are consistent with field observations where river sections experiencing high shipping intensity were found to have 10–30 % lower aquatic plant cover and simplified fish communities dominated by species tolerating a wide range of environmental conditions, compared with reaches where channel form was similar and ship traffic lower (Kucera-Hirzinger et al., 2009; Sexton et al., 2024). Thus, chronic ship-wave exposure in heavily navigated low-land rivers was found to be associated with reduced plant cover and simplified littoral habitat structures (Kucera-Hirzinger et al., 2009; Sexton et al., 2024; Verdonschot, 2012).

The increases in turbulence and current velocity observed in field and flume studies are sufficient to dislodge eggs, larvae, and small juveniles from vegetated margins and to uproot or damage aquatic plants (Gabel et al., 2017; Scherbaum et al., 2025; Verdonschot, 2012). Other potential mechanisms underlying the observed impacts include increased energetic costs for fish and invertebrates repeatedly exposed to ship-induced turbulence; interference with feeding and predator-avoidance behaviour can also be important, as well as direct mortality through stranding or abrasion of organisms on hardened banks (Gabel et al., 2017; Kucera-Hirzinger et al., 2009; Scherbaum et al., 2025; Sexton et al., 2024). Overall, these findings suggest that ship traffic contributes to the simplification and homogenisation of river communities, compounding other impacts in navigable waterways such as channelisation, changes in flow patterns and possibly other stressors (Albert et al., 2021; Kucera-Hirzinger et al., 2009; Sexton et al., 2024).

4.3.4 Water pollution

Water pollution comes in many forms. Domestic wastewater has traditionally been of greatest concern, because improper disposal dramatically increases disease risk. According to the World Health Organization (WHO), global

incidences and the annual death toll remain high because of deficits in sanitation and hygiene. Each year, more than one million people continue to die from diarrhoea alone, more than 40 % of them children under the age of 5 (WHO, 2026). Pathogens causing other water-borne diseases add to the dire situation. They include bacteria, viruses, protozoa, and metazoan parasites. Infections occur via runoff or infiltration of water contaminated with faecal pathogens into surface or ground water. Huge investments into wastewater treatment and hygienic measures, along with education, have greatly reduced casualties in industrial countries and many urban areas around the world. However, the problem continues to be immense in many rural and deprived urban areas, compromising human water security at the global scale (Vörösmarty et al., 2010).

Wastewater treatment being an effective remedy, there are now nearly 10,000 plants in Germany alone that treat domestic wastewater. Furthermore, with technology constantly improving, treatment plants are becoming increasingly effective at removing nutrients from wastewater, both phosphorus and nitrogen, thereby partly addressing nutrient enrichment as the second leading cause of freshwater pollution (Wurtsbaugh et al., 2019). However, besides this point pollution by treated and untreated wastewater, ground and surface waters receive

nutrients from diffuse sources, particularly fertilised agricultural fields, and other pathways such as atmospheric deposition play a role as well. Furthermore, chemical pollution other than by nutrients is widespread, permanently evolving and extremely diverse (Wang et al., 2024) – and so are the entry and transformation paths. The range of chemical pollutants extends from synthetic organic chemicals such as pesticides and pharmaceuticals, personal care products, illegal drugs, microplastics, non-ionic liquids, and numerous other industrial compounds to organic and inorganic nanoparticles, salt, radionuclides, mercury, and a broad range of other metals. Only some of these chemicals are removed by wastewater treatment plants (Schwarzenbach et al., 2006) and the scope of this chemical pollution remains greatly underappreciated overall (Bernhardt et al., 2017; Sigmund et al., 2023; Schäfer et al., 2026).

The impacts of chemical pollution on freshwaters are similarly diverse. Particularly well-documented is the importance of nutrient enrichment for stimulating algal growth in lakes, but nutrients can also be significant in streams and rivers, wetlands and groundwater. In lakes, this eutrophication, defined as an increase in plant productivity, entails numerous knock-on effects. Most notable are an increase in fish production, benefitting from an enhanced resource base; the development of algal blooms

compromising water clarity and often resulting in toxin production when toxicogenic cyanobacteria become dominant; and oxygen depletion in deep water of stratified lakes when sinking organic matter decomposes, which in turn can lead to fish kills. Other impacts are rarely acknowledged. One example is the stimulation of organic matter decomposition in streams (Ferreira et al., 2015), with continental-scale consequences for carbon cycling in flowing waters (Woodward et al., 2012).

Many of the non-nutrient chemical pollutants are directly toxic, causing mortality or sublethal impairments or stress responses, and in the case of endocrine disruptors interfering with the hormonal balance of animals (Escher et al., 2020; Rico et al., 2025). All of these effects have repercussions for population dynamics, evolutionary trajectories, species interactions, and shifts in the structure of freshwater communities and ecosystem processes associated with the activity of species (Schäfer et al., 2026). Since many of the synthetic compounds are specifically designed to have biological effects (Bernhardt et al., 2017), particularly pesticides and pharmaceuticals and often at low doses, effects on biodiversity are inevitable. Furthermore, it is important to realise that species interactions in affected communities can produce cascading indirect effects that may lead to counterintuitive outcomes (Gessner & Tlili, 2016).

Thermal pollution from power-plant cooling water, industrial effluents, urban runoff and assisted groundwater recharge, superimposed by climate-induced warming (O'Reilly et al., 2015) and potentially mitigated by thoughtful flow and riparian plant management, adds to chemical pollution and to effects on water temperature caused by water extraction and dams (see above). Raising river and lake temperatures directly affects the metabolism of organisms, including the balance of energy intake with food and energy expenditure by respiration. Species can be pushed to their tolerance limits, life-history patterns and phenologies can shift (e. g. the timing of leaf fall), and species interactions and distributions can change.

Species adapted to warm water, often non-native taxa, tend to be favoured by thermal pollution, while species restricted to a narrow low-temperature range, such as many salmonid fishes and cold-adapted invertebrates, are at a disadvantage (Caissie, 2006; Olden & Naiman, 2010). Shifts in invertebrate community composition towards warm-adapted species and reduced growth and survival of cold-water fish by 10–30 % have indeed been documented in response to increases in summer temperature by 2–4 °C (Caissie, 2006; Olden & Naiman, 2010). Indirect temperature-induced changes in environmental conditions can mitigate or compound direct effects of warming, for example when

fish are squeezed between deep-water oxygen depletion caused by a temperature-induced increase in productivity and intolerably high surface water temperatures in stratified lakes.

The introduction, spread and establishment of species previously absent from a location has been referred to as biological pollution (Boon et al., 2020). The concept comprises invasive freshwater megafauna such as beavers in southern Chile (Anderson et al., 2009) or hippopotamuses in Colombia (Shurin et al., 2020) as well as invasive predatory fish (Cucherousset & Olden, 2011), nuisance aquatic plants (Di Lernia et al., 2025) toxic algal blooms (Köhler et al., 2024) or the microbial communities replacing resident communities in response to raw sewage inputs into streams, but also the clearance of turbid lakes (Mauch et al., 2026; Sommer et al., 2012). Impacts of biological pollution can be dramatic, ranging from massive fish kills (Köhler et al., 2024) to extensive ecosystem engineering (Anderson et al., 2009; Mauch et al., 2026), altered food-web structure, increased competition and predation pressure on native species, potentially triggering ecosystem regime shifts towards fundamentally new states (Dudgeon, 2010; Ricciardi, 2007).

Chemical, thermal, and biological pollution rarely act in isolation on freshwater biodiversity and ecosystems, but rather jointly and often

interactively (Dudgeon, 2010; Vörösmarty et al., 2010), along with the variety of other environmental factors (see above). These complex relationships have been captured in a concept referred to as the exposome (Rappaport & Smith, 2010). Estimates at the global scale suggest that more than 60 % of large rivers are affected by multiple types of pollution (nutrients, organic compounds, pathogens), coinciding with regions where freshwater vertebrate populations tend to decline (Albert et al., 2021; Vörösmarty et al., 2010). Raising temperatures can increase chemical toxicity and metabolic demand, and nutrient enrichment and organic pollution promote harmful algal blooms and hypoxia that interact with toxicants and invaders to harm biodiversity (Caissie, 2006; Escher et al., 2020). The effects of even just chemical mixtures commonly exceed those predicted based on information on single substances (Rico et al., 2025). For example, mixture toxicity at river sites in Europe exceeding pesticide and nutrient thresholds (which showed a marked shift towards tolerant taxa and 30–60 % lower invertebrate richness) often exceeded risk thresholds even when individual compounds were below regulatory limits (Rico et al., 2025).

The combined impacts of multiple drivers affecting freshwater biodiversity and ecosystems are evident across taxonomic groups and types of ecosystem. Meta-analyses and large-scale

assessments show that nutrient and pesticide loads are strongly associated with reduced invertebrate richness, loss of sensitive taxa, and dominance of tolerant generalists, especially in agricultural and urban catchments (Rico et al., 2025). Thermal stress compounds these effects by narrowing thermal safety margins, increasing susceptibility to disease, and altering reproductive success, leading to local extirpations of temperature-sensitive fish and invertebrates (Caissie, 2006; Olden & Naiman, 2010). Invasive species, often facilitated by physical disturbance, nutrient enrichment and warming, further affect native biodiversity through competition, predation and hybridisation, contributing to biotic homogenisation (Dudgeon, 2010; Ricciardi et al., 2013). Clearly, chemical, thermal, and biological pollution are core drivers of global freshwater biodiversity decline, in tandem with extensive hydrological alterations and habitat loss that erode ecosystem integrity (Albert et al., 2021).

4.3.5 Climate change

Climate change exacerbates the conflict between human water uses and related interventions on the one hand and the requirements to maintain and restore freshwater biodiversity and ecosystem processes on the other. The mechanisms underlying the aggravation of the conflict include reduced water supply and supply reliability, increased human water demand and flow extremes, and a change in the timing of both. Warming, altered precipitation patterns, altered timing of snowpack and glacier melt, increased drought frequencies and intensities, and more extreme events affect the quantity, temporal and spatial distribution, and quality of water, including temperature, salinity, acidity, and concentrations of dissolved oxygen, nutrients and contaminants. Coping with the impacts of climate change on freshwater biodiversity and ecosystems thus means recognizing climate change as an integral part of a syndrome rather than as a separate problem that can be dealt with in isolation. In particular, climate change is projected to exacerbate competition for water during low-flow periods, raising the risk that current allocation rules will fail to protect freshwater biodiversity and ecosystems under future hydrological regimes.

4.4 Significance of freshwater biodiversity and ecosystem processes for water security

Freshwater biodiversity and ecosystem processes provide a wide range of services for people (Fig. 4.2), some of which relate directly to the multidimensional concept of water security (Lynch et al., 2023; Song et al., 2024).

Four dimensions are particularly noteworthy:

- ▶ Water availability: volumes of water sufficient for human and ecosystem needs
- ▶ Water quality: clean water that meets standards for human health, agriculture and industrial uses
- ▶ Water reliability: reliable availability over time, including in the face of seasonal and interannual variability
- ▶ Safety: protection from floods, droughts, contamination and other hazards

Water security is not only a function of climatic and geological conditions, allocation rules, and water infrastructure. It is also influenced by biodiversity and the ecosystem processes depending on biological communities (Izah et al., 2026). The functional traits of the species constituting these communities regulate key hydrological, geomorphological and biogeochemical processes, including water infiltration, flow regulation, sediment transport, organic

matter and nutrient transformations, and water purification, which in turn determine water availability, quality, supply reliability, and safety for people. This means that biodiversity supports ecosystem processes with a bearing on water security, and water security feeds back on biodiversity. The resulting feedback loop can either stabilise resilient systems or, if degraded, lead to self-reinforcing decline.

Vegetation, including floodplain forest and other wetland plant communities, regulate fluxes of water through evapotranspiration and by increasing soil infiltration and slope stability. Vegetation also reduces overland flow, dampens peak flows, enhances low flows, and reduces erosion (Herath et al., 2025; Kingsbury-Smith et al., 2023). Wetlands and riparian areas are important sites for groundwater recharge, for the immobilisation of pollutants, and for buffering floods (Gurnell & Grabowski, 2016). Groundwater supplies drinking water to 1.5–3 billion people. Thus, sustaining groundwater-dependent ecosystems is a biodiversity issue. In rivers and lakes, diverse communities of fish, invertebrates, microbes, and plants regulate water quality through nutrient uptake and recycling, organic matter decomposition, control of

algal and pathogen proliferation, sediment stabilisation by rooted vegetation and bioturbation (Kuntz & Tyler, 2018).

Losing biodiversity weakens the ability of rivers to clean water, provide food, and resist floods (Dudgeon & Strayer, 2025). Freshwater ecosystems harbouring rich communities are more likely to break down organic waste and cycle nutrients efficiently. When sensitive species are lost, processes can slow down and become more variable, potentially leading to poorer and more variable water quality. Ecosystems whose communities are less diverse are also less likely to be resilient to extreme events like droughts or floods, as fewer species are likely to be capable of playing complementary roles (Mendonça et al., 2026). Consequences for humans can be increased costs of water treatment and loss of commercial fishing or recreational opportunities (Birk et al., 2025). Ecosystems processes also shape river channel forms and habitat structures. Vegetated banks stabilise sediments and reduce erosion. Roots and bioturbation influence channel shape and sediment transport. Aquatic plants and invertebrates create microhabitats and affect flow patterns. Fishes and invertebrates influence sediment dynamics

through feeding and movement. These processes determine whether rivers maintain habitat diversity or become impoverished, unstable channels. In turn, the form of river channels strongly influences habitat availability, flow regimes, and connectivity (Zhou & Toda, 2026).

4.5 Key dimensions underlying effective water governance

Impacts on freshwaters vary regionally but interconnections over large scales often require addressing them in a broader context. This is particularly true for transboundary rivers, where catchments transcend national or domestic borders, often delimited by river courses or involving upstream-downstream relationships. Water governance, in contrast, is organised by administrative units (countries, regions, municipalities) typically defined by political borders, where boundaries of catchments as the fundamental hydrological unit are mostly ignored. The situation applies not only to rivers, but also to lakes, wetlands and groundwater, all of which are intimately connected with one another and their catchments. This geographic reality creates a complex web of competing demands and interests surrounding water issues that can result in contentious transnational, regional and even local politics (see [Box 5](#)).

The spatial mismatch between natural water boundaries and administrative units makes it difficult to distribute water resources fairly, protect biodiversity, and resolve conflicts. Effective governance clearly requires an overarching cooperative body that operates at the

catchment scale. It must explicitly identify conflicts, trade-offs and potential synergies, and be given a mandate to balance the environmental, economic, and political dimensions of shared water bodies (De Bruin et al., 2024; IUCN, 2024).

Impediments to address the challenges are:

- ▶ Mental traditions that neglect physical ecosystem connections and catchments as a coherent management unit
- ▶ Jurisdictional fragmentation of different countries, regions, and sectors claiming authority over parts of the same catchment
- ▶ Institutional inefficiencies resulting in overlapping or missing mandates and in weak coordination mechanisms
- ▶ Competing stakeholder interests between upstream vs downstream, agriculture vs energy vs environment, etc.

Transboundary water governance involves multiple scales, often independent legal frameworks, different stakeholders that come with variable power, and varied political cultures and attitudes (Schmeier, 2013). It is useful to distinguish four levels:

- ▶ Global: conventions, financing mechanisms, and international institutions (e. g., UN Water Convention, Global Environment Facility (GEF) of the CBD)
- ▶ Regional: river basin commissions, joint monitoring programs, transboundary agreements
- ▶ National: ministries, water laws, sectoral policies
- ▶ Local: municipalities, irrigation associations, communities, Indigenous groups.

Different aspects must be considered at each level (e. g., agriculture, fisheries, biodiversity, energy), which will be influenced by political goals, technocratic interests, expertise, and executive power (De Bruin et al., 2024). It is also important to acknowledge that functional responsibility and sovereignty can be dynamic and need to be contextualised according to, for example, political circumstances, climate conditions, and economic priorities.

Time dimensions to respect include:

- ▶ Seasonal variability (floods, droughts)
- ▶ Interannual variability (e. g., El Niño Southern Oscillation cycles)
- ▶ Long-term trends (climate change, land-use change)
- ▶ Policy cycles
- ▶ Institutional memory.

Different spatial dimensions also play a role, particularly:

- ▶ Local catchments vs. entire drainage basins
- ▶ Sub-basins vs. main river basins
- ▶ National territory vs. transboundary networks.

Ignoring these dimensions leads to governance arrangements that are too rigid or too fragmented to address coupled socioeconomic and natural environmental dynamics.

A fundamental spatial pattern in transboundary river systems is the upstream–downstream asymmetry (Jiang et al., 2025). Upstream countries control water quantity, land use, and pollution sources. Downstream countries depend on upstream decisions for water availability, quality, and flood risk mitigation. Key factors fuelling conflicts include scarcity and inequality, power asymmetries, and infrastructure effects. Growing populations, climate change, and unsustainable agricultural practices exacerbate

water scarcity, creating tensions between upstream and downstream interests and fuelling conflicts over access and distribution. Larger, richer, or more politically powerful countries can often impose their preferences (Bendz et al., 2026).

Downstream consequences can be profound, especially when dams, water diversions, and drainage projects alter flow regimes and sediment transport or when chemical and thermal pollution compromise water quality. Geographic variation of impacts includes some regions facing drying trends and reduced snowpacks, while others are confronted with increased flood risk and extreme rainfall or experience shifting seasonal patterns (e. g., earlier snow melt, altered timing of floods and low flow). These changes affect water availability for agriculture, energy production, drinking-water provision, and ecosystems, and they often intensify upstream–downstream tensions. Climate change can exacerbate drought and flood events as well as pollution, further complicating water management and underscoring the need for transboundary cooperation.

Finally, different sectors experience spatially and temporally distinct dependencies:

- ▶ Irrigation agriculture is sensitive to water quantity and seasonal water distribution

- ▶ Hydropower is sensitive to flow regimes and sediment transport
- ▶ Drinking water provision is sensitive to water quality and reliability of supplies
- ▶ Shipping is sensitive to water depth and channel stability
- ▶ Fisheries and biodiversity are sensitive to flow regime, habitat quality and connectivity.

Global and other transnational or transregional institutions can alleviate potential conflicts by elaborating sustainable policies and practices and fostering institutional innovation to meet this objective. For example, globally operating institutions (i) play a role in setting norms and standards (e. g., UN Water Convention, SDG 6), (ii) provide financial support for transboundary projects (GEF, regional development banks), and (iii) facilitate dialogue and technical cooperation (IUCN, 2024). The Operational Strategy of the GEF recommends using a variety of tools to this end:

- ▶ Transboundary Diagnostic Analysis (TDA): a joint, science-based “fact-finding” assessment of transboundary problems and their root causes
- ▶ Strategic Action Plan (SAP): a negotiated policy and investment plan derived from the TDA and endorsed at the ministerial level

- ▶ National inter-ministerial committees: country-level coordination bodies that feed into the TDA/SAP and translate regional agreements into national reforms
- ▶ Local demonstration activities: on-the-ground projects used to build commitment, test approaches, and link regional priorities to local implementation.

These tools aim to facilitate building capacity, trust and confidence among nations to focus on shared basins and country commitments to reforms and investments that take an integrated approach to land and water policies and management (Duda & La Roche, 1997). Some transboundary agreements have contributed to sparing water demands from leading to geopolitical tensions. However, water resources are increasingly instrumentalised in disputes and conflicts. As climate change intensifies water shortages, governance fragmentation and limited access to data, knowledge, and funds constrain joint action, particularly in fragile, conflict-affected, or climate-vulnerable regions.

4.6 Key messages

► **Linkages between freshwater and biodiversity are reciprocal.**

Freshwater biodiversity and ecosystems not only respond to physical and geochemical environmental conditions, but also feed back on the environment by influencing water quantity, water quality, and water-associated physical structures (e.g., beaver dams creating impoundments, vegetation shaping river channels, filtering invertebrates regulating water clarity).

► **Intact freshwater ecosystems underpin human water security.**

Freshwater biodiversity and ecosystems are instrumental in regulating water purification, recycling nutrients, sequestering carbon, stabilising sediments, buffering floods, and providing various other ecosystem services. These functions mean that declines in biodiversity and ecosystem integrity not only reflect impacts on nature, but also undermine flood-risk mitigation, water availability, the reliability of water supplies, and water quality for human uses.

► **Most interventions intended to ensure human water security harm freshwater biodiversity and ecosystems.**

Water abstraction, dam construction and operation, river regulation and navigation alter hydrological regimes, multiple dimensions of connectivity, temperature regimes, and sediment dynamics. Chemical pollution and the spread of biological entities ranging from viruses to megafauna add to the physical impacts. These changes are tightly linked to declines in freshwater biodiversity and ecosystem integrity.

► **Dams have especially strong and multidimensional impacts on biodiversity.**

Half of the world's large rivers are fragmented by dams (and similar structures), the expansion of hydropower generation could raise this share to over 90 %. Apart from retaining water and creating standing water bodies, dams alter thermal regimes both upstream and downstream, trap sediment, cause downstream channel incision and floodplain disconnection, block migration routes for fish, and promote methane emissions to the atmosphere.

► **When abstraction capacities for surface and renewable groundwater are reached, societies turn to seriously damaging and costly large-scale interventions.**

Prime examples are fossil groundwater mining and inter-basin water transfers, which can solve local water shortages but generally entail irreversible hydrological, ecological, humanitarian and adverse economic impacts in the donor and recipient system or both (e.g., aquifer depletion, salinisation and other changes in water geochemistry, faunal homogenisation through species invasions and extinctions). More benign alternatives include water-saving measures, virtual water trade and desalinisation.

► **Not all human interventions have negative consequences for freshwater biodiversity and ecosystems.**

Investments that primarily improve water quality (e.g., wastewater treatment) immediately benefit both people and biodiversity, providing opportunities for synergies in the implementation of socioeconomic and environmental objectives. Restoration efforts such as dam decommissioning and removals, floodplain reconnections to river

channels, riparian plant restoration, beaver reintroductions in their native range, or the partial recovery of the Aral Sea, show that ecological damages caused by human interventions on surface and groundwaters are not always irreversible if appropriate action is taken. The dynamic nature of streams and rivers creates particular opportunities for restoring the physical template supporting freshwater biodiversity and ecosystem processes.

► **Spatial and institutional mismatches between catchment delineations and administrative units are serious obstacles to effective water governance.**

Water governance is typically organised by institutions that are fragmented and whose political and administrative boundaries are incongruent with the delineation of catchments as the fundamental hydrological unit for rivers, lakes, wetlands and groundwater. Improvement of water governance requires cooperative organizations that operate at the catchment scale. To be effective, they need an explicit mandate to identify conflicts, navigate trade-offs and explore synergies among environmental, economic, and political dimensions of shared water bodies, especially in transboundary contexts.

Box 5 Iran's overexploited water resources and inland aquatic biodiversity

Petra Döll and Aidin Niamir

In 1971, the Ramsar Convention on Wetlands of International Importance was signed in the eponymous Iranian town of Ramsar on the Caspian Sea. Iran has since designated 27 Ramsar sites, covering almost 15,000 km² (Ramsar, 2026). These wetlands form part of a much wider network of inland aquatic ecosystems, including freshwater rivers, lakes, and marshes as well as brackish wetlands and hypersaline lakes. The term inland aquatic biodiversity is therefore more appropriate in this context than freshwater biodiversity, as several ecologically important water bodies are saline while remaining dependent on freshwater inflows and basin-scale water management.

Lake Urmia in north-western Iran ([Fig. Box 5.1](#)) is the country's largest Ramsar site and perhaps its most striking example of ecological degradation. Once covering 5,000–6,000 km², this shallow hypersaline lake is surrounded by extensive brackish marshes and is protected as both a Biosphere Reserve and National Park. Its marshes support reed beds and large breeding colonies of waterbirds,

including flamingos, and provide important resting and feeding habitats for migratory species. Lake Urmia thus illustrates that aquatic biodiversity in drylands is not limited to freshwater habitats but also includes highly specialised saline and brackish ecosystems that depend on stable hydrological conditions. Since the late 1990s, however, Lake Urmia has undergone a dramatic decline in surface area and water volume. Water levels have fallen from a relatively stable 1,278 m a.s.l. to below 1,270 m in 2015, 2022, and 2025 (Sima et al., 2023, 2024; Tehran Times, 2026). At such levels, the lake covers less than one-third of its former surface area and has lost more than 90 % of its water volume (Mousavi, 2024). The ecological consequences have been immediate: waterbird numbers have declined sharply, strongly correlated with falling water levels and the related increase in salinity (Shariati & Hemami, 2024). The survival of *Artemia*, the brine shrimp at the base of the lake's food web, depends on salinity thresholds that may no longer be recoverable in practice. The wider ecological system, from crustaceans to

birds, has already been fundamentally altered (Sima et al., 2021, 2023).

Lake shrinkage is driven primarily by the expansion of irrigated agriculture and the construction of artificial reservoirs within the basin, alongside climatic pressures (Hojabri & Nguyen-Huy, 2026; Hosseini-Moghari et al., 2020). Internationally supported restoration efforts, including artificial water transfers, modified reservoir management, improved irrigation technology, and wastewater reuse for irrigation, have so far failed to restore the lake to its ecological target level. Achieving this target would require an estimated 40 % reduction in consumptive irrigation water use (Mousavi, 2024; Tehran Times, 2026). Lake Urmia has therefore moved beyond what is conventionally understood as a water (and biodiversity) crisis. A crisis implies a severe but temporary disruption, from which a system can, in principle, return to its previous state (see [Box 1](#)). In Lake Urmia, as in many parts of Iran, irreversible changes have already occurred. Desiccation and the water transfers have irreversibly altered the shape of Lake Urmia (Sima et al., 2023), while

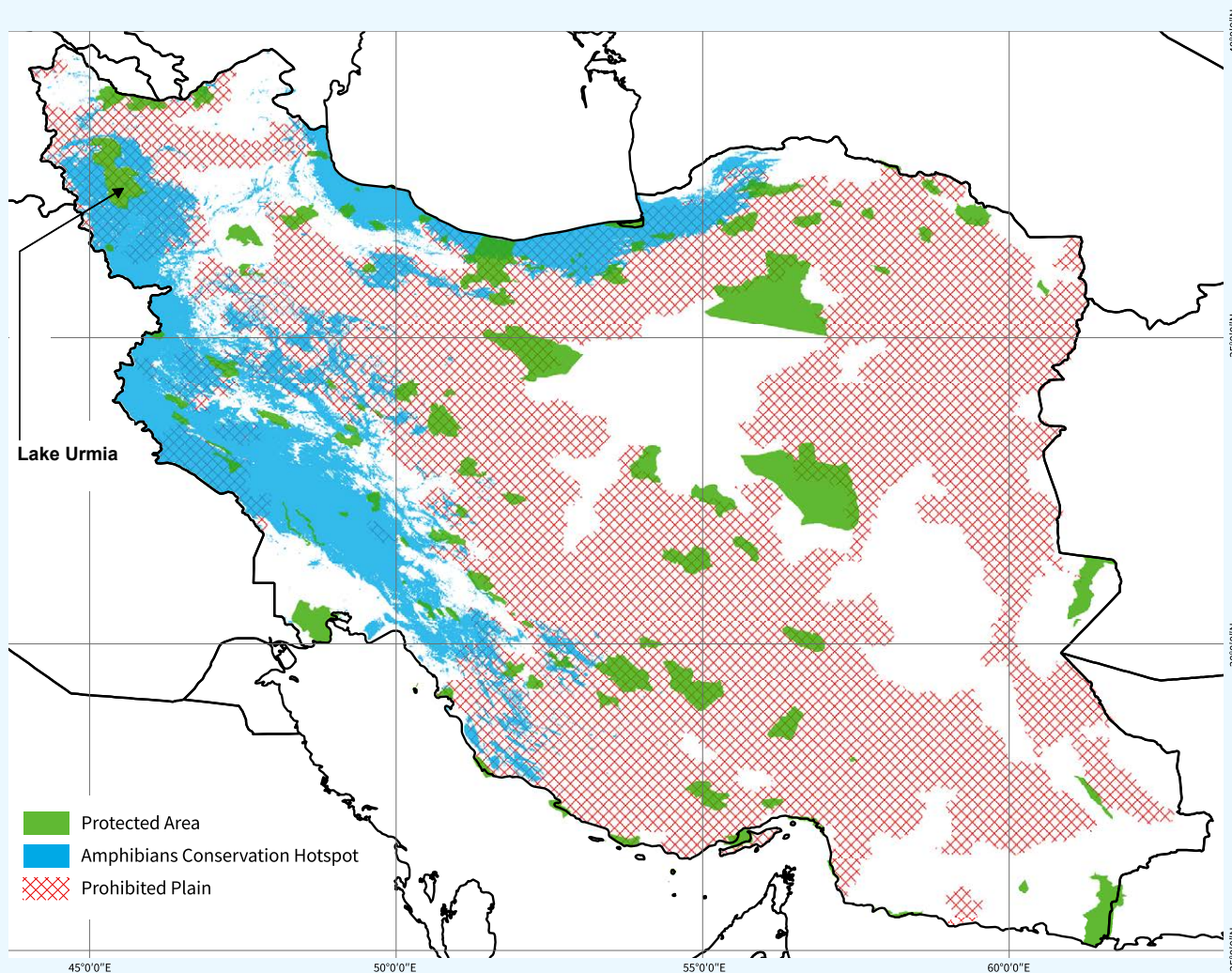


Fig. Box 5.1 Groundwater depletion and conservation priorities in Iran. Areas designated as “prohibited plains” by the Iran Water Resources Management Company (IWRMC), where new groundwater abstractions are only allowed for potable water supply but not for irrigation, are shown with red hatching (IWRMC, 2024). National protected areas are shown in green (DOE, 2024) and amphibian conservation hotspots in blue (Ghaffari et al., 2024). The map illustrates the broad geographic extent of hydrological pressure and its overlap with areas of recognised conservation importance.

ecological recovery remains uncertain even if water levels rise again (Sima et al., 2021).

Lake Urmia is far from unique. Similar patterns are seen across Iran. Of the 25 Ramsar sites currently assessed, 15 are classified as declining, six as improving, and four as stable. Human water use, with 90 % of consumptive water use for irrigation, and the construction of many artificial reservoirs, has profoundly altered river flows. Of 139 long-term river gauging stations across Iran, 56 % show a significantly declining discharge, while 20 % of formerly perennial rivers, including the Zayandeh-Rud, have become intermittent (Maghrebi, 2023). Groundwater depletion, caused by groundwater abstractions exceeding groundwater recharge, is perhaps the clearest sign of irreversible changes. Since 1979, natural groundwater levels have fallen dramatically across nearly all regions of Iran (Noori, 2021). Between 2002 and 2015 alone, groundwater tables declined by an average of 49 cm per year, despite an almost 10 % reduction in groundwater consumption resulting from both water demand management and limited groundwater resources (Noori, 2021).

Groundwater depletion, which is mainly caused by water abstractions for irrigation, is exacerbated by declining groundwater recharge since 2009. This decline has been attributed primarily to the depletion of surface water resources, which recharge the groundwater less than before, rather than climate change (Noori, 2023). Groundwater depletion has caused widespread and largely irreversible aquifer compaction and land subsidence, permanently reducing the subsurface water storage capacity and thus eroding the country's hydrological capital (Payne, 2025). The Iranian government has reacted by designating so-called "prohibited plains", where drilling of new groundwater wells is not permitted except for potable water supply; they cover large parts of Iran (Fig. Box 5.1). Land subsidence due to groundwater depletion is observed in many of the "prohibited plains" (Payne, 2025), which include many protected ecosystems and some amphibian conservation hotspots (Fig. Box 5.1). At the same time, human water appropriation in Iran has exceeded the country's renewable water resources, while climate change has further reduced the availability of these resources (Maghrebi, 2023).

Water stress in Iran is further complicated by the transboundary nature of several major

river systems, linking ecological degradation to regional geopolitics. The Hamoun wetlands in eastern Iran, for example, depend heavily on inflows from the Helmand River, which originates in Afghanistan. Reduced inflows due to upstream dam construction, water diversions, and prolonged droughts have contributed to the near collapse of the Hamoun system, with severe consequences for biodiversity, livelihoods, and dust generation. This has also become a recurrent source of political tension between Iran and Afghanistan (Bidabadi & Afshari, 2020; Nagheeby, 2024). Similar, although less acute, pressures affect other shared basins such as the Aras River. These examples highlight that water scarcity in Iran cannot be understood purely as a domestic management challenge but must also be considered within a wider regional hydro-political context.

Water quality has also deteriorated in Iran. Salinisation, irrigation with untreated wastewater, and pollution by heavy metals, organic compounds and microplastics have degraded rivers, wetlands, and groundwater systems (Maghrebi, 2021; Zarei et al., 2025). In drylands, where flows are already low, pollution has an even more pronounced effect, because less water means less dilution and therefore greater ecological stress.

The consequences of diminished water quantity and quality for Iran's inland aquatic biodiversity have been significant. Despite its predominantly dry climate, Iran harbours a surprisingly rich and highly specialised biodiversity. Its freshwater fish fauna alone includes 241 native and endemic species; many are confined to single basins and are therefore particularly vulnerable to flow reduction, habitat fragmentation and pollution (Jouladeh-Roudbar et al., 2020). Iran's amphibian fauna comprises 18 recorded species, most of which depend on aquatic habitats for breeding; 40 % of these species are endemic to the country (Fig. Box 5.2), and more than 40 % are considered at risk of extinction (Safaei-Mahroo et al., 2023). Amphibian diversity is heavily concentrated in the Caspian Hyrcanian forests and the Zagros Mountains, which contain the country's primary richness- and rarity-based hotspots, more than 90 % of which fall outside the current protected-area network (Fig. Box 5.1) (Ghaffari et al., 2024).

A few examples illustrate how the degradation of aquatic systems can result in severe and potentially irreversible biodiversity changes. The Sistan snow trout (*Schizothorax zarudnyi*), historically native to the Hamoun Lakes and Helmand River system, now survives mainly



Fig. Box 5.2 Gorgan Brook Salamander (*Iranodon gorganensis* (Clergue-Gazeau & Thorn, 1979) synonym of *Paradactylodon persicus* (Eiselt & Steiner, 1970), an endangered amphibian endemic to Iran. Photograph courtesy of Barbod Safaei-Mahroo (c)

in artificial reservoirs following widespread wetland desiccation and degradation (Akbari et al., 2022; Gharaei, 2012). The Persian bream (*Acanthobrama persidis*) has disappeared from parts of its former range (Jouladeh-Roudbar et al., 2020; Mousavi-Sabet & Eagderi, 2016). Populations of Caspian brown trout (*Salmo trutta caspius*) and Persian sturgeon (*Acipenser persicus*) have also declined sharply as a consequence of river fragmentation, pollution, and overexploitation (IUCN, 2021, 2022).

Plants and invertebrates tell a similar story. In the Anzali Wetland, eutrophication, hydrological alteration and invasive species have displaced native aquatic plants such as *Nymphaea alba*, while populations of the habitat-forming submerged plant *Stuckenia pectinata* show strong genetic isolation linked to fragmentation and water loss (M. Albrecht et al., 2016; Chaichisemsari et al., 2021). Benthic invertebrates and freshwater insects, which respond quickly to changes in water quality and flow conditions, have shifted towards more pollution-tolerant communities in wetlands such as Shadegan and Hawr al-Azim (Nasirian & Salehzadeh, 2019). These changes matter because they not only cause a loss of biodiversity, but also a loss of ecological

functions, including nutrient cycling, food-web dynamics and water purification.

Overall, there is clear evidence of a long-term degradation of Iran's inland aquatic systems, with altered water flows and storages, water quality, biodiversity, and ecosystem functions. In many regions, ecological recovery to the healthy state before the onset of degradation is not longer possible. Rivers that once flowed year-round now dry seasonally. Wetlands have collapsed. Aquifers have compacted. Ecological communities have been depleted, fragmented, or lost. And the climate is rapidly changing. Iran illustrates what happens when long-term overuse pushes dryland water systems beyond ecological and hydrological tipping points, and where it is no longer possible to '*meet the needs of the present without compromising the ability of future generations to meet their own needs*' (Brundtlandt, 1987).

The task now is to work with this new reality. This means stabilising what remains, preventing further tipping points from being crossed and managing water and biodiversity jointly under altered, far less forgiving conditions. Difficult systemic transformations are required that go far beyond siloed water and biodiversity management. Iran does not lack technical expertise or scientific capacity. It has

considerable experience in water engineering, supply and demand management, and a large community of well-trained water and biodiversity experts (Madani, 2014). Large-scale water transfer and artificial groundwater recharge projects merely redistribute a scarce resource between regions and water storage compartments. Meanwhile, human water use has not been sufficiently capped; this is mainly due to an increase in food demand and the goal of achieving food self-sufficiency (Saatsaz, 2020). In addition, the rural population has not yet decreased and needs to gain their livelihoods by means of (mostly irrigated) crop production.

While a large number of systemic policies have been proposed, related to, for example, population distribution and growth, farming regulations and organisations, water and energy pricing, water markets, stakeholder involvement and empowering Iran's Department of Environment (Madani, 2014), effective solutions to Iran's water and biodiversity problems are economically and politically costly (Madani et al., 2016). They require a reduction in human water use, and it remains unclear how such a systemic transformation can be achieved in the near future given the current socio-economic and political conditions.

5 Economic implications of water stress and declining ecosystem services

Bernd Hansjürgens and Arne Leifels

5.1 The economic perspective: natural capital and ecosystem services at risk

From an economic perspective, the water and biodiversity crisis – and the interdependencies and reinforcing feedback between the two – represents a depletion of natural capital and, consequently, the erosion of a fundamental basis for economic value creation (TEEB, 2010; Dasgupta, 2021). As natural capital declines, so do the benefits it provides to people and society through ecosystem functions and services. The situation has reached a point where the discourse is shifting beyond a ‘water crisis’ or ‘biodiversity crisis’ or ‘water shortages’ towards the notion of ‘water bankruptcy’ (Madani, 2026) - an irreversible depletion of natural assets that can no longer be restored (see [Box 1](#)).

The interactions between societal and economic activities and natural capital can be analysed through four complementary lenses:

- ▶ **Impacts (drivers):** To what extent do economic and societal activities contribute to the depletion of natural capital? What are the direct and indirect drivers, as well as the underlying economic causes, of the water and biodiversity crisis, and how can these be assessed from an economic perspective? ([Chapter 5.2](#))
- ▶ **Dependencies and risks:** How and to what extent do the consequences of the water and biodiversity crisis affect the economy, society, and the environment? Where do critical dependencies on natural capital exist, and how can they be identified and assessed? ([Chapter 5.3](#))
- ▶ **Economic value:** What economic value is lost as natural capital and ecosystem services decline, and where are limits of economic valuation of the environment? ([Chapter 5.4](#)).
- ▶ **Economic policy recommendations:** Which strategies and policy instruments can help address the water and biodiversity crisis? What approaches to risk assessment, risk mitigation, and financing should be pursued to halt and reverse the loss of natural capital? ([Chapter 5.5](#))

5.2 Impacts: from broad drivers to sector-specific relationships

5.2.1 Studies on drivers and impacts: largely general and abstract

Several drivers of water stress and pressures on freshwater ecosystems have already been mentioned in [Chapter 2.2.3](#) of this report. These include land-use change, pollution, ground-water and surface-water depletion, the direct exploitation of water resources, invasive alien species, and climate change. The overall picture associated with these drivers is the ‘Great Acceleration’: following the Second World War, there has been a sharp rise in socio-economic trends exerting unprecedented pressure on ecosystems and water resources (see [Chapter 1](#) and [Chapter 2](#)). Mid-18th century industrialisation, in contrast, had no global impacts on this scale. Similarly, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), with its focus on global biodiversity loss, identified as primary drivers land-use change, the overexploitation of natural resources, pollution, climate change, and invasive alien species (IPBES, 2019; IPBES, 2026).

Most of these drivers are focusing on direct drivers, and they are general and abstract. They are closely associated with GDP growth and

expanding production. At the same time, in many countries, rising water demand from agriculture, industry, energy production, and households is placing increasing pressure on freshwater resources, exacerbating water scarcity and intensifying competition among different water uses. Consequently, biodiversity loss and the water crisis are intrinsically linked and mutually reinforcing. In many cases, economic activities contribute to water stress, ecosystem degradation, and the loss of natural capital, while economies and societies are simultaneously highly dependent on nature (TNFD, 2023b). However, the underlying economic causes are not addressed.

The broad perspective on drivers can be illustrated, among other things, by comparing the development of natural capital with that of produced and human capital. Between 1992 and 2019, global per capita economic wealth increased by 60 %. During this period, per capita produced capital nearly doubled (+92 %), while per capita human capital increased by 38 %. In contrast, per capita natural capital was halved over the three decades ([Fig. 5.1](#)).

Change in worldwide inclusive wealth per capita and other indicators, 1992–2019

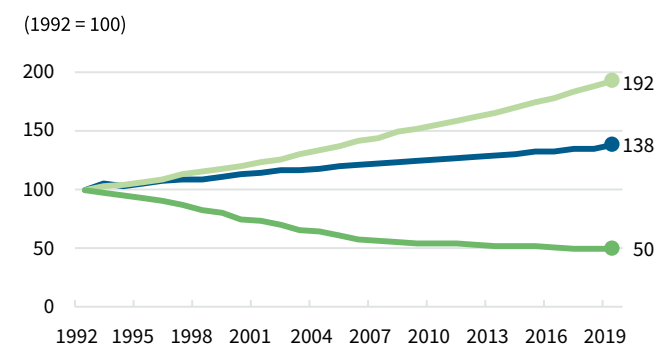


Fig. 5.1 Global Capital Stocks per Capita. From 1992 to 2019, there has been a strong increase in produced capital (from 100 to 192), an increase in human capital (from 100 to 138), while at the same time there has been a significant shrinkage of natural capital (from 100 to 50). (Source: UNEP, 2023b)

The general considerations regarding natural capital (in relation to produced capital) can also be illustrated by examining long-term economic trends. While the global economy expanded from approximately US\$ 1.18 trillion in 1820 (Bolt & Zanden, 2024) to more than US\$ 130 trillion in 2022 (World Bank, 2024),

much of this growth has been underpinned by the depletion of natural capital. These figures highlight a fundamental point: Biodiversity and ecosystem services have sustained economic development, yet their value has not been adequately reflected in economic decision-making. As a consequence, the loss of natural capital has emerged as a systemic economic risk, with far-reaching implications for business performance, financial stability, and long-term economic resilience (Dasgupta, 2021).

Apart from general considerations of GDP growth and Natural Capital, drivers of the global water crisis (see [Chapter 2.2](#)) are largely similar to those identified in the context of ecosystem degradation and biodiversity loss. Following the OECD's (2026) terminology (OECD, 2026a), the key dimensions of the water crisis can be characterised as 'too much' (for flooding), 'too little' (for droughts and increasing scarcity), and 'too polluted' (for pollution and contamination). Globally, key drivers identified in the water literature include excessive freshwater withdrawals for agriculture, which account for approximately 70 % of global freshwater withdrawals (FAO, 2021), glacier retreat, increasing frequency and severity of droughts (UN Water, 2023; UNESCO, 2024), and deteriorating water infrastructures leading to significant water losses through leakage and inefficiencies (World Bank, 2016; WRI, 2023) (see [Chapter 2.2](#)).

However, all of these drivers are framed from an ecological perspective and formulated at a highly aggregated level, mainly focusing on direct impacts on freshwater and ecosystems. While they provide first entry points for policy intervention, they are frequently too broad to inform specific economic strategies, sector-specific measures, or actionable recommendations. Examining the causes and drivers of the water and biodiversity crises through such a global lens therefore does not, on its own, provide sufficient insight into their underlying causes and dynamics. Against this background, greater attention should be paid to indirect drivers, such as economic growth, overconsumption, and the depletion of natural resources through specific economic activities. This can help identify entry points for policy instruments and measures that influence these dynamics. It is therefore important to take a closer look at the economic mechanisms underlying these developments. The economic literature has identified a range of economic mechanisms and market failures that contribute to the depletion of natural capital.

5.2.2 Drivers from an economic perspective

Markets function efficiently only when prices reflect the full social costs and benefits of economic activities. However, if the value of natural

resources is not captured in prices, ecosystem services – such as freshwater provision, flood regulation by wetlands, carbon sequestration in forests, or pollination by insects – are not adequately reflected in market decisions. Therefore, ecosystem services often lack explicit market prices despite their substantial economic value – they are so-called 'public goods' (Samuelson, 1954; Musgrave & Musgrave, 1989). As a result, these benefits are frequently treated as free inputs, while the costs of their degradation are externalised to society and future generations (Dasgupta, 2021). Negative externalities or 'social costs' (Pigou, 1920) arise, but these costs of economic activities are not reflected in the decisions of private economic agents.

The TEEB initiative *The Economics of Ecosystems and Biodiversity* (2010) and the Dasgupta Review (Dasgupta, 2021) have impressively demonstrated that the value of nature has historically been largely invisible in economic measurement and decision-making. Nature is often treated as a free and unlimited asset in national accounting systems and corporate decision-making, rather than a critical form of capital underpinning economic activity (Grunewald et al., 2024). Consequently, conventional measures of economic growth provide a misleading picture of economic progress, as the depletion of natural capital is not adequately reflected.

Another important driver is the overuse of common-pool (or common-good) resources. Resources such as freshwater systems, fisheries, forests, and grazing lands are economically characterised by shared access and limited availability (Ostrom, 1990). Individual users may have strong incentives to maximise their own benefits from resource extraction, while the resulting costs of depletion are dispersed among all users and future generations. This challenge is mirrored in Hardin's concept of the 'Tragedy of the Commons' (Hardin, 1968). In water systems, this phenomenon is evident, for example, in the unsustainable extraction of groundwater for agricultural irrigation and industrial production. While the above-mentioned Samuelson-Musgrave theory of public goods mainly focuses on the free-rider-problem of resource use, the common-goods perspective by Hardin, Ostrom, and others places the focus on the overuse and depletion of natural resources. From an economic perspective, the challenge is to derive governance rules and institutions that offer solutions for the phenomenon of common-pool goods – this is particularly challenging when the degradation of freshwater resources and biodiversity occurs on the global level (GCEW, 2024). The reason is that the allocation of property rights (understood as rights to use a resource) is extremely difficult when different national and international legal systems, governance

structures, and institutions are involved as is the case in transboundary water systems.

A major economic driver of biodiversity loss and the depletion of aquatic systems is the prevalence of short-term decision-making over long-term issues and the discounting of future environmental costs. Economic decisions often prioritise immediate utility or financial returns, while long-term ecological damages and the loss of ecosystem services and biodiversity are insufficiently reflected in current valuations (Stern, 2007; Weitzman, 2007; Dasgupta, 2021). Consequently, activities such as deforestation, wetland conversion, and groundwater extraction may appear economically beneficial in the short term despite generating significant long-term ecological and economic costs (Millennium Ecosystem Assessment, 2005).

In addition to the distortion of economic incentives by market failures and myopic behaviour, the water and biodiversity crises are further intensified by environmentally harmful policies. Globally, governments continue to provide financial support for activities that directly contribute to significant environmental degradation (UNEP, 2026; ELD, 2026). These include subsidies that encourage intensive agricultural practices, fossil fuel consumption, and water-intensive production systems (see [Chapter 5.2.3](#)).

The above-mentioned high-level assessments of the economic drivers of ecosystem degradation and biodiversity loss, as well as the underlying economic explanations, theories, and concepts, provide deeper insights into the causes of freshwater degradation and ecosystem losses. However, they remain insufficient to identify specific drivers and derive appropriate solutions. A more granular analysis is required in two respects: First, the underlying drivers need to be further disaggregated to identify their contributions at the level of specific sectors, regions, and, where appropriate, individual companies. Only with a sufficiently detailed understanding of the sources and distribution of these impacts can effective strategies, interventions, and policy instruments be designed to address and reduce the underlying pressures. A central challenge is that the required degree of spatial, temporal, and sectoral granularity varies depending on the specific analytical objective and decision context. Second, it is necessary to develop and consolidate relevant metrics and indicators to support decision-making by actors in the public sector (e.g., policymakers) and the private sector (e.g., banks, insurers, investors, and other financial institutions).

5.2.3 Financial flows impacting nature and nature-based solutions

There are several options for gaining a more granular understanding of the activities of different economic sectors that cause environmental degradation. One option is to examine financial flows. The State of Finance for Nature Report (UNEP, 2026) provides an important entry point for such a sector-specific analysis. The Report highlights the substantial imbalance between financial flows that contribute to nature degradation on the one hand and investments in nature-positive solutions on the other hand. An estimated US\$ 7.3 trillion are invested annually in activities with negative impacts on nature (US\$ 4.9 trillion on private financial flows, particularly for sectors like energy and industry, and US\$ 2.4 trillion annually for subsidies and public expenditure supporting nature-negative activities (see [Fig. 5.2](#)), while only a fraction of this amount (US\$ 220 billion) is directed towards Nature-based Solutions (NbS) and ecosystem restoration. The relationship between nature-destroying and nature-positive financial flows is more than 30 to one (UNEP, 2026), leading to the conclusion that nature is ‘in the red’ from an accounting perspective.

Private nature-negative finance is highly concentrated in a limited number of sectors. The utilities sector alone accounted for approximately

US\$ 1.6 trillion in nature-negative finance in 2023, mostly in electrical utilities (US\$ 1.3 trillion), and on a smaller scale in water utilities (US\$ 49 billion). The industrials sector generates the second largest nature-negative private finance flow (US\$ 1.4 trillion). Public expenditures also severely impact nature, water resources and biodiversity. Almost half of the US\$ 2.4 trillion negative public finance flows are fossil fuel subsidies (47 %) in 2023, with agriculture and water subsidies accounting for 17 % each. Thus, the water resources sector itself represents an important component of these harmful financial flows. Approximately US\$ 400 billion of environmentally harmful subsidies are associated with the use of water resources. These water-related subsidies are identified as encompassing the underpricing of water for irrigation and industrial uses, as well as utility pricing structures that encourage unsustainable water extraction and consumption. It should be noted, however, that a much larger part of nature-negative finance affects freshwater ecosystems directly, e. g., in agriculture, manufacturing, utilities, and construction.

The term NbS refers to a range of approaches to address societal challenges through the sustainable management, protection, and restoration of ecosystems. NbS leverage ecosystem services to generate benefits for both people and nature. They can address a wide range of challenges,

including biodiversity loss, climate change, soil degradation, food insecurity, and public health. For an intervention to qualify as an NbS, it must generate net positive outcomes for nature through the protecting, restoration, or sustainable management of ecosystems. Measures that merely compensate for environmental damage elsewhere do not, in themselves, qualify as NbS. NbS are often presented as investments capable of delivering multiple ecological and societal benefits simultaneously. UNEP (2026) highlights evidence that NbS can be highly cost-effective: a review of disaster-risk-reduction projects found that in 65 % of cases NbS were more effective than conventional engineering solutions, while the restoration of degraded land can generate returns ranging from US\$ 7 to US\$ 30 for every dollar invested (UNEP, 2026).

Nature-based Solutions and “green infrastructure” are gaining popularity in the water sector. They can improve water availability and quality and help manage water-related disaster risks. NbS strengthen water security across watersheds and water supply systems by regulating precipitation, soil moisture, and natural water storage. This improves the quantity, timing, and spatial distribution of water for multiple uses. Nature-friendly storage options include natural wetlands, reforestation, and enhanced groundwater recharge. NbS can also support better water quality by protecting source waters.

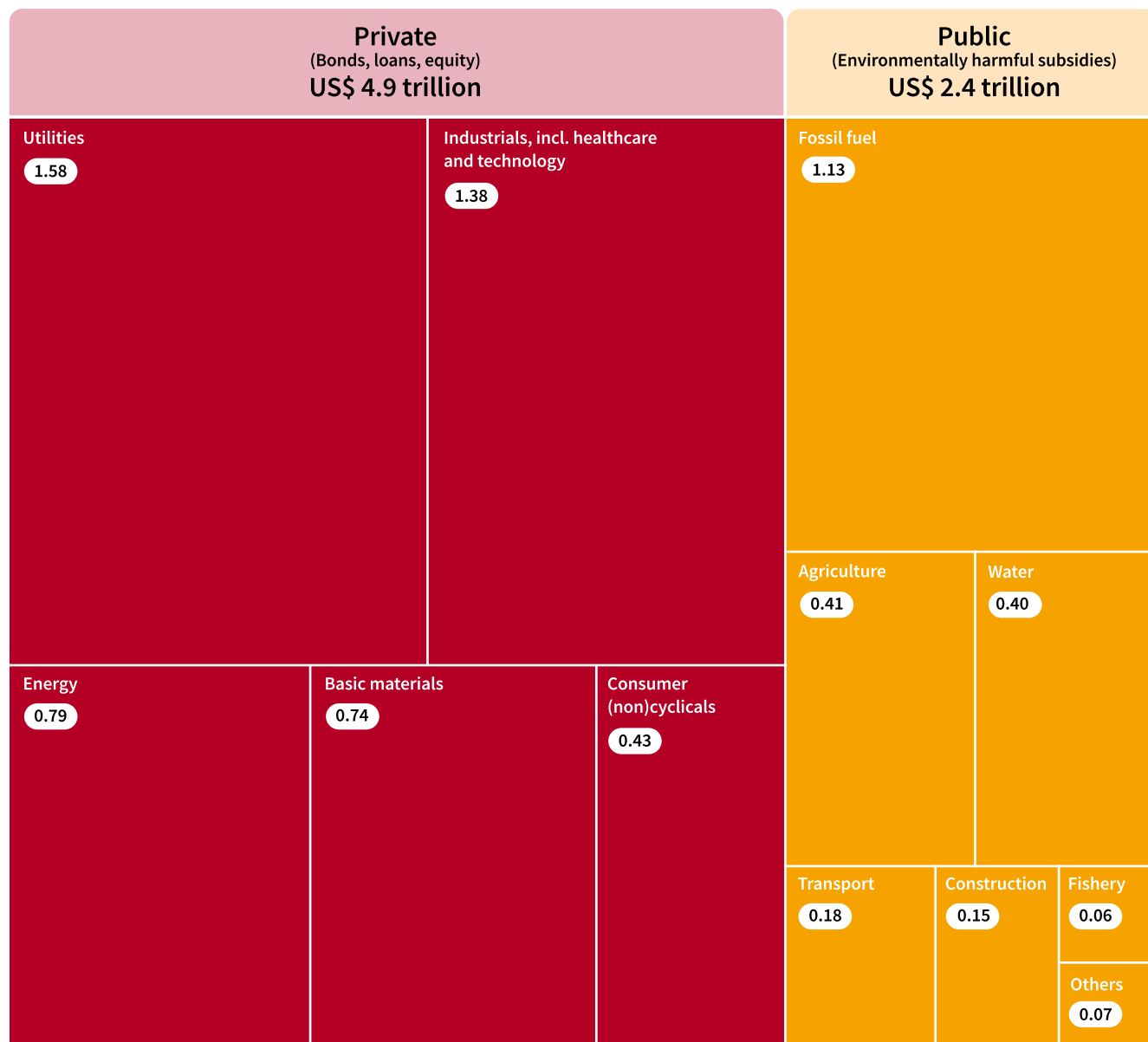
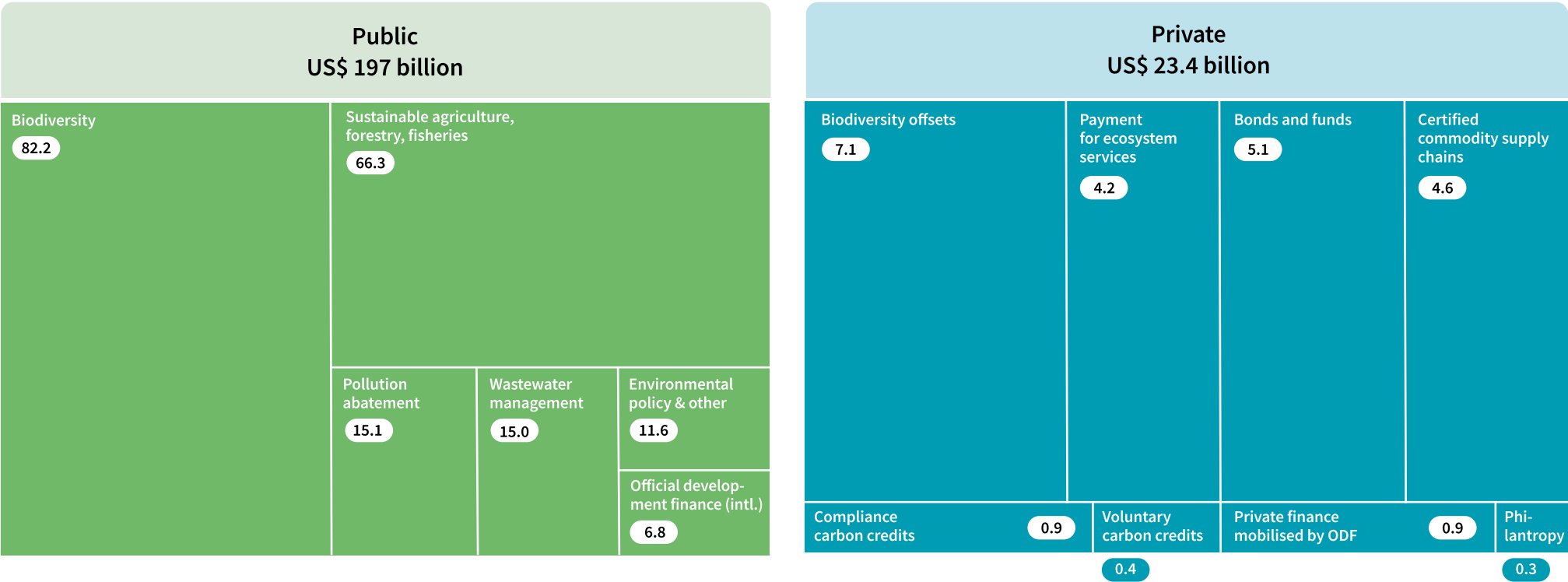


Fig. 5.2 Nature in the Red: The figure shows negative financial flows in the private and the public sector. The size of every field in the figure mirrors the monetary amount of negative flows. Roughly two thirds of the negative flows emerge in the private sector, and one third in the public domain. (Source: UNEP, 2026, Figure ES.2)

Particularly in water infrastructure projects, NbS may lower both capital and operational costs, for example through erosion control that reduces sedimentation, or via complementary wastewater treatment measures such as constructed wetlands. For risks like floods and droughts, green infrastructure includes features such as mangroves in coastal areas and riparian gallery forests along rivers, supporting flood protection and stormwater management.

Fig. 5.3 Public and Private Finance targeting NbS, and by sectors in 2023. The Figure shows annual public (left) and private (right) financial flows targeting nature-positive solutions. The volume of public and private financial flows differs considerably. Public financial flows amounted to US\$ 197 billion in 2023 whereas private financial flows were substantially lower amounting to only US\$ 23,4 billion. The size of the boxes within the public and the private fields reflect the volume of financial flows. On the public side, the largest share was allocated to biodiversity spending, while on the private side, biodiversity offsets accounted for the largest share. (Source: UNEP, 2026)



Nature-positive finance flows to NbS reached US\$ 220 billion in 2023, representing a 5 % increase compared with 2022. Public domestic expenditure represented the largest share of NbS finance, accounting for approximately US\$ 197 billion, while private finance represented US\$ 23 billion (UNEP, 2026) (see [Fig. 5.3](#)). The largest shares of the public US\$ 197 billion are allocated for biodiversity (US\$ 82.2 billion) and agriculture, forestry, and fishery (US\$ 66.4 billion), while the largest share of the private US\$ 23 billion is used for biodiversity offsets (US\$ 7.1 billion).

The increase in public expenditure is a positive sign. It does not only reflect growing efforts to address ecosystem degradation and biodiversity loss through NbS, but it may also be linked to commitments under the Global Biodiversity Framework, which aims to mobilise at least US\$ 200 billion annually for biodiversity by 2030. Public expenditure included investments in biodiversity protection, sustainable agriculture, forestry, fisheries and wastewater management. Wastewater management accounted for approximately US\$ 15 billion of public expenditure in 2023, demonstrating the relevance of water-related investments within the broader portfolio of NbS (UNEP, 2026).

Although Official Development Finance (ODF) represents only a small fraction of public NbS finance in the amount of US\$ 6.8 billion (3.5 %), it is particularly important. ODF is highly concentrated in biodiversity hotspots that are acutely threatened and can cause global repercussions if degraded to the point of irreversible damage (e. g., the Amazon rainforest). Furthermore, there are usually weaker safeguarding institutions and smaller investment capabilities in ODF countries. Within these ODF finance flows there are several water-related sectors receiving international capital, including water resources conservation, river basin development, water policy, and water supply and sanitation.

Environmental NGOs also play an important role in financing NbS, particularly in developing and emerging economies where financial risks discourage private investment. There is evidence that private-sector investment in NbS with water-related objectives, including flood-risk mitigation, water supply and water quality improvements, reached approximately US\$ 345 million in 2023. In addition, approximately US\$ 39 million of private NbS finance mobilised by Official Development Finance was directed towards water-related sectors in 2023 (UNEP, 2026).

5.2.4 Sector-specific impacts

The global water crisis that affects ecosystem service provision and biodiversity is primarily driven by three major economic activities: Absolute water consumption (water withdrawal), the pollution of water bodies, and the “virtual water trade” (embedded water in products) play a decisive role (see also [Chapter 2.2](#)).

Agricultural food production is currently the dominant land use, covering 34 % of the Earth’s ice-free terrestrial surface, and has the largest overall environmental impact (ELD, 2026). By 2050, a global population of 9.7 billion may require up to 56 % more food than in 2010. Meeting this demand would require an additional 593 million hectares of land for cultivation – equivalent to approximately twice the size of India (Searchinger et al., 2018). These trends will increase pressure on water resources and freshwater biodiversity through expanded irrigation and virtual water trade – apart from the direct biodiversity effects through land-use change. Agriculture already accounts for approximately 70 % of global freshwater withdrawals. In arid regions or emerging economies, this figure is often above 80 %, including field irrigation (UNEP, 2026; FAO, 2021; GCEW, 2024). Around 40 % of global food production comes from irrigated agriculture (World Bank, 2026; UNESCO, 2026). This leads to massive groundwater

depletion in major farming regions of the world. Inefficient irrigation technologies cause high water losses through evaporation and seepage. Furthermore, pesticides and fertilisers as diffuse pollutants impair water quality (OECD, 2017, 2026a).

Industry, manufacturing, and utilities account for approximately 20 % of global freshwater withdrawals. In highly industrialised regions, such as Europe, the industrial sector often represents more than 50 % of total water withdrawals. In Germany, e. g., industrial water use is three times higher than private household use. This high demand is primarily driven by cooling water used in electricity generation, as well as by cleaning, dyeing, and solvent-intensive processes in manufacturing (UNESCO, 2024; IEA, 2016; EEA, 2021). Although cooling water is returned to rivers or lakes, it is often discharged at elevated temperatures, contributing to thermal pollution and potentially harming the respective aquatic ecosystems (IEA, 2016).

Textile manufacturing is a particularly water-intensive industry. Producing 1 kg of cotton fibre requires up to 10,000 litres of water, depending on climatic conditions and agricultural practices (Mekonnen & Hoekstra, 2011). Large-scale irrigation for cotton production has been identified as one of the principal drivers of water scarcity in the Fergana Valley in Uzbekistan and the

Table 5.1 Sectors and industries with the most severe and systemic impacts on freshwater ecosystems and resources: the impacts are differentiated into supply chain impacts, direct operations, and product use. Very high impact is marked by red colour, high impact – orange, medium impact yellow. (Source: WWF, 2026b, p. 35, based on Famiglietti et al., 2022)

GICS Sector	Industry	Impact Pathways					
		Supply Chain		Direct operations		Product use/end of life	
		Water quantity	Water quality	Water quantity	Water quality	Water quantity	Water quality
Consumer staples	Food products	very high impact	very high impact	high impact	high impact	not enough information	not enough information
	Beverages	very high impact	very high impact	high impact	high impact	not enough information	very high impact
Consumer discretionary	Textiles, apparel and luxury goods	high impact	high impact	medium impact	very high impact	not enough information	high impact
Energy	Oil and gas	high impact	very high impact	high impact	very high impact	not enough information	medium impact
Healthcare	Pharmaceuticals	not enough information	not enough information	not enough information	very high impact	not enough information	very high impact
Materials	Chemicals	high impact	high impact	medium impact	very high impact	not enough information	not enough information
	Metals and mining	high impact	high impact	high impact	very high impact	not enough information	not enough information
	Paper and forest products	high impact	high impact	high impact	very high impact	not enough information	not enough information
IT	High-tech and electronics	high impact	high impact	high impact	high impact	not enough information	not enough information
	Semiconductors and circuit boards	not enough information	not enough information	medium impact	high impact	not enough information	not enough information
	Battery manufacturing	high impact	high impact	not enough information	high impact	not enough information	high impact
Utilities	Renewable electricity (hydropower)	not enough information	not enough information	very high impact	medium impact	not enough information	not enough information

shrinkage of the Aral Sea (Martínez-Aldaya et al., 2010; Chapagain et al., 2006). Additional sectors with high water demand include the chemical industry, semiconductor manufacturing, and, increasingly, data centres, where substantial quantities of water are required for cooling and process operations (IEA, 2025; UNESCO, 2024).

Sector-specific impacts on water quantity and quality arise throughout the production cycle (and thus in different locations). [Table 5.1](#) depicts high-impact sectors with their respective impacts through supply chains, direct operations and product use. Food and beverage industries have the largest impacts on both water quantity and quality in their supply chains, mainly due to agricultural inputs. This is a major example of the afore-mentioned virtual water trade. Several industries particularly affect water quality through direct operations, e. g., textiles, fossil energy, chemicals/pharmaceuticals, timber/paper, and metals/mining. Pharmaceuticals and beverages also have a pronounced effect on water resources and freshwater ecosystems during product use (WWF, 2026b).

Municipal and domestic water use accounts for approximately 10 % of global freshwater withdrawals (UNESCO, 2024). The principal uses include drinking water, sanitation, cooking, washing, and irrigation of private gardens. Although per capita domestic water

consumption has declined in many industrialised countries—for example, from 147 litres in 1990 per person per day in Germany to approximately 121 litres in 2023—rapid urbanisation in many parts of the world continues to place increasing pressure on local water supply and distribution systems (UNESCO, 2024; BDEW, 2024; UN-Habitat, 2022). While pressures on water resources and ecosystem services can be observed at the global scale, many challenges emerge most acutely at the regional scale, where over-abstraction, pollution, climate change impacts, and high levels of consumption may occur either in isolation or cumulatively (see [Box 6](#)).

Box 6: Case study – impacts on regional level may vary – extreme water scarcity in Jordan

Bernd Hansjürgens

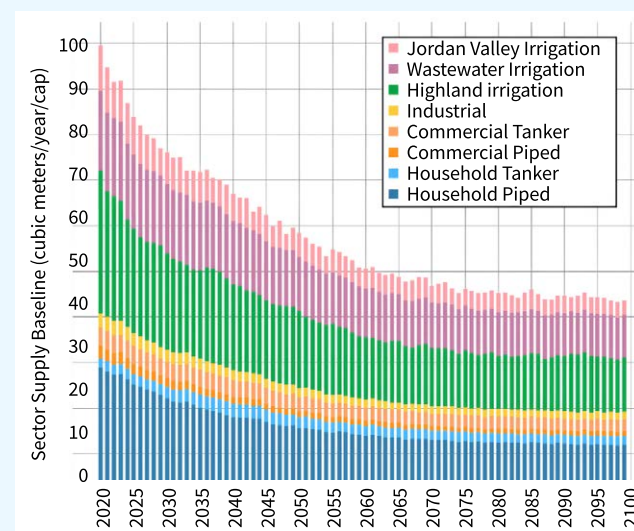
Jordan's water crisis reflects more than physical water scarcity alone. It results from the interaction of limited natural water availability, groundwater depletion, infrastructure constraints, population growth, governance failures, and climate change. Modelling scenarios suggest that due to dwindling freshwater resources and a rapidly growing population Jordan's future per-capita water availability could decline from currently 95 litres to 43 litres by 2100, which is less than half of today's level (Yoon et al., 2021) (Fig. Box 6). This is expected to further intensify water scarcity in the entire region. A particular challenge is the unequal distribution of water: while the average household receives 36 hours of piped water per week (21 % of the time), some households receive water for only six hours per week (4 % of the time). Given these infrastructural shortcomings, informal tanker markets are not a peripheral phenomenon but a structurally relevant feature of urban water provision in Jordan. They represent an often overlooked dimension of the climate and equity debate (Klassert et al., 2023). Scientific analysis sheds light on both the structural drivers of water insecurity and

the long-term implications of continued reliance on tanker-based supply rather than the established piped water system.

As a response to Jordan's extreme water scarcity a proposal to the Green Climate Fund (GCF) for the Aqaba-Amman Water Desalination and Conveyance Project is currently being developed, also known as the National Water Carrier. The project is designed to transport desalinated Red Sea water to highland regions across Jordan and provide approximately 300 million m³ of drinking water annually. Public project documents report total costs of approximately US\$ 6 billion, including US\$ 295 million in Green Climate Fund financing. The National Water Carrier project is urgently needed; however, additional measures to address increasing water scarcity are indispensable, as supply-side measures alone will only partially improve future water security in Jordan. While such measures may stabilise aggregate water availability at levels comparable to today's supply, they will leave many households exposed to persistent water stress. Improving water distribution is therefore equally

important. This has important practical implications, as many households currently rely on informal water markets supplied by tanker trucks. If supply augmentation is not accompanied by improvements in water distribution, water insecurity will continue to affect households unevenly.

Fig. Box 6 Projected water supply in Jordan until 2100. Water demand is differentiated into eight sectors. Water supply by sectors (in cubic meters per capita and year) will significantly decrease, from 95m³/yr in 2020 to 43m³/yr in 2100. (Yoon et al., 2021)



5.3 Dependencies and risks: how economy and society depend on freshwater and ecosystems

5.3.1 Economic implications of land degradation and droughts

Economic activities not only drive nature degradation, increase freshwater scarcity, and threaten biodiversity – they also depend fundamentally on nature, i. e. on freshwater, functioning ecosystems, biodiversity, and the ecosystem services they provide. While these dependencies were long overlooked in scientific analyses and public discourse, they have gained increasing attention in recent years, particularly in the wake of extreme weather events such as droughts and floods.

Freshwater resources and the degradation of land ecosystems are closely interconnected. Excessive water withdrawals and water pollution cause and exacerbate land degradation and soil depletion. The resulting loss of ecosystem functions can, in turn, undermine the provision of essential ecosystem services, including water regulation, water purification, soil fertility, and carbon sequestration. The ecological foundation of the economy is eroding at an alarming pace amid an accelerating environmental crisis: up to 40 % of the world's land area is degraded,

droughts are becoming more widespread and severe, and biodiversity loss and climate change are intensifying (UNCCD, 2022). The severity of this crisis is partly rooted in the persistent failure to account for the true value of land-based natural capital. This contributes to the misallocation of capital and systemic underinvestment in land restoration and sustainable land management (ELD, 2026).

Land degradation poses significant risks to the prosperity and well-being of present and future generations. According to Economics of Land Degradation (ELD, 2026) the decline in ecosystem services resulting from land degradation generates estimated economic losses of approximately US\$ 92 billion annually. Of this total, cropland degradation accounts for the largest share, at US\$ 36 billion per year. Forest degradation accounts for US\$ 33 billion, and grassland degradation for US\$ 23 billion in global losses. All world regions show increases in the extent of degraded forests, grasslands, and croplands, except for Central America and the Caribbean. The lost value of ecosystem services is highest for forests in Northern America and South America (US\$ 12.8 billion per year and US\$ 11.6

billion per year respectively) and grassland in Sub-Saharan Africa (over US\$ 17 billion per year).

Droughts associated with extreme water scarcity are one of the costliest threats to economies and societies, affecting every continent and impacting more than 1.8 billion people (ELD, 2026). The economic costs of drought-related damages are estimated at approximately US\$ 307 billion per year (Thomas et al., 2024). These costs can escalate through the cascading effects of drought across sectors and the wider economy, for example through disruptions to supply chains. A striking example is the severe drought around Lake Gatún in Panama in 2023: As the lake provides water for the operation of the Panama Canal, the drought resulted in a 33 % reduction in the number of daily ship transits. This reduced Panama's canal revenues, which normally amount to approximately 6 % of GDP, or around US\$ 2.5 billion per year, while also increasing global transport costs and disrupting shipping and logistics, with potential knock-on effects on consumer prices. The drought also created trade-offs between allocating water to Panama City, hydropower generation, and canal

operations (IMF, 2024; McKinsey & Company, 2024).

Land degradation and drought, caused by a lack of water, lead to increasing economic risks. For the private sector increasing land degradation and drought, closely intertwined with the loss of natural capital and water stress, are physical risks that increase operational costs, jeopardise revenues and decrease the value of assets, threatening business performance and financial stability. This results in increasing scarcity and the risk of supply shortages of critical goods and services that ecosystems provide such as clean water, healthy soils, pollination, and carbon storage.

As a recent example, the Economics of Land Degradation Report *Financing Land* compiles evidence for the biodiversity and water crisis impacting large parts of the economy (ELD, 2026). For example, decreasing natural pollination already has companies hiring pollination services or even resorting to mechanical pollination at significant costs. In the US, farmers paid more than US\$ 400 million for such services in 2024 (Ceres, 2025). A partial collapse of critical ecosystems could reduce global GDP by 2.3 % by 2030, with low- and lower-middle-income countries facing GDP losses of up to 10 % (World Bank, 2021). In Europe, an extreme but plausible drought with a 25-year return period could put

nearly 15 % of economic output at risk (ECB, 2025). Over the next decade, valuation adjustments of around 4–5 % may be expected for the seven largest UK banks in their domestic holdings due to physical nature-related risks (Green Finance Institute, 2024). Looking at individual commodities, losses in yields and higher prices or volatilities are already occurring—or are likely to materialise—such as for olives in Tunisia (ELD and GIZ, 2024), agave and mezcal production in Mexico (ELD and GIZ, 2022), and other, more general crops (Ortiz-Bobea et al., 2021).

5.3.2 The EU economy's dependency on nature and ecosystem services

The loss of biodiversity and ecosystem services has an impact on economic value creation. This is due to the fact that ecosystem services are an important natural factor of production in numerous economic sectors. The relationship between ecosystem services and economic production is often not immediately visible and may vary depending on the specific activity or subsector. To capture these 'invisible' dependencies, the ENCORE framework (Exploring Natural Capital Opportunities, Risks and Exposure) assesses 271 economic subsectors across 25 ecosystem services based on literature reviews, expert judgments and quantitative indicators. Dependency levels are expressed through materiality

ratings ranging from very low to very high. A high dependency rating indicates a potentially significant exposure to nature-related shocks. Disruption or degradation of the relevant ecosystem service may therefore affect the ability of an economic activity to maintain production and result in financial losses.

Direct dependency on ecosystem services is particularly evident and pronounced in the case of agriculture and forestry – and it spans a wide range of ecosystem services. On the one hand, this primary economic sector is heavily reliant on the provision of services from forest and agricultural land: the production of biomass, water supply and the provision of genetic resources. On the other hand, there is also a very high degree of dependence on numerous regulatory ecosystem services – such as soil, air and water quality, climate and precipitation regulation, natural pollination and protection against storms and flooding (see [Table 5.2](#)).

Table 5.2 Direct dependency of (selected) economic sectors on (selected) ecosystem services. Dependency levels are expressed through materiality ratings ranging from “1 = very low” to “5 = very high” based on literature reviews, expert judgements and quantitative indicators. (Source: ENCORE Partners, 2020)

		Provisioning services			Regulating and maintenance services																Cultural services			
		Biomass provisioning	Water supply	Genetic material	Solid waste remediation	Soil and sediment retention	Water purification	Soil quality regulation	Biomass provisioning	Biological control	Air filtration	Flood mitigation services	Global climate regulation	Local climate regulation	Nursery/habitat maintenance	Noise attenuation	Pollination	Storm mitigation	Water flow regulation	Rainfall pattern regulation	Recreation related services	Visual amenity services	Education/scientific	Spiritual/artistic/symbolic
Agriculture/forestry	Growing of cereals, leguminous crops and oil seeds	very high	high	very high	medium	very high	very high	very high	medium	high	medium	high	very high	very high	very low	n. a.	high	high	high	very high	n. a.	n. a.	n. a.	n. a.
	Growing of perennial crops	very high	high	very high	medium	very high	very high	very high	medium	high	medium	high	very high	very high	very low	n. a.	very high	high	high	very high	n. a.	n. a.	n. a.	n. a.
	Raising of cattle and buffaloes	very high	high	medium	medium	very high	high	high	low	medium	medium	medium	medium	medium	very low	very low	n. a.	high	high	very high	no data	no data	very high	very high
	Logging	very high	high	very high	medium	very high	very high	very high	no data	high	medium	high	very high	very high	high	n. a.	medium	medium	high	very high	n. a.	n. a.	n. a.	no data
	Fishing	very high	high	high	very high	very high	very high	medium	medium	high	low	medium	very high	medium	very high	no data	n. a.	high	high	very high	no data	no data	very high	very high
Quarrying of stone, sand and clay		low	high	n. a.	no data	high	very high	n. a.	medium	n. a.	very low	high	high	low	n. a.	very low	n. a.	medium	high	very high	n. a.	n. a.	n. a.	n. a.
Manufacturing	Manufacture of food products	n. a.	high	n. a.	medium	low	very high	n. a.	low	very low	very low	medium	very low	low	n. a.	n. a.	n. a.	medium	high	n. a.	n. a.	n. a.	n. a.	n. a.
	Manufacture of beverages	n. a.	high	very high	medium	low	high	n. a.	low	no data	n. a.	medium	very low	low	n. a.	n. a.	n. a.	medium	high	n. a.	n. a.	n. a.	no data	no data
	Manufacture of wearing apparel	n. a.	medium	n. a.	medium	low	medium	n. a.	low	n. a.	very low	medium	very low	low	n. a.	very low	n. a.	medium	medium	very low	n. a.	n. a.	n. a.	n. a.
	Manufacture of paper and paper products	n. a.	medium	n. a.	medium	low	n. a.	n. a.	low	very low	very low	medium	very low	low	n. a.	very low	n. a.	medium	medium	medium	n. a.	n. a.	n. a.	n. a.
	Manufacture of pharmaceuticals/medicinal products	n. a.	high	high	low	medium	very high	n. a.	low	n. a.	very low	medium	very low	low	n. a.	n. a.	n. a.	medium	high	n. a.	n. a.	n. a.	very high	n. a.
	Manufacture of plastics products	n. a.	low	n. a.	low	low	medium	n. a.	low	n. a.	very low	medium	very low	low	n. a.	low	n. a.	medium	medium	very low	no data	no data	no data	no data
	Manufacture of basic iron and steel	n. a.	high	n. a.	low	low	medium	n. a.	no data	n. a.	very low	medium	very low	low	n. a.	low	n. a.	medium	high	medium	n. a.	n. a.	n. a.	n. a.
	Manufacture of electrical equipment	n. a.	medium	n. a.	low	low	medium	n. a.	low	n. a.	very low	medium	very low	low	n. a.	low	n. a.	medium	medium	medium	n. a.	n. a.	n. a.	n. a.
	Manufacture of machinery and equipment	n. a.	medium	n. a.	low	low	medium	n. a.	low	n. a.	very low	medium	very low	low	n. a.	low	n. a.	medium	medium	low	n. a.	n. a.	n. a.	n. a.
	Manufacture of motor vehicles	n. a.	low	n. a.	low	medium	medium	n. a.	low	n. a.	very low	medium	very low	low	n. a.	low	n. a.	medium	medium	low	n. a.	n. a.	n. a.	n. a.
	Manufacture of parts/accessories for motor vehicles	n. a.	low	n. a.	low	medium	medium	n. a.	low	n. a.	very low	medium	very low	low	n. a.	low	n. a.	medium	medium	low	n. a.	n. a.	n. a.	n. a.
	Manufacture of furniture	n. a.	medium	n. a.	medium	low	medium	n. a.	low	n. a.	very low	medium	very low	low	n. a.	low	n. a.	medium	medium	very low	n. a.	n. a.	n. a.	n. a.

		Provisioning services			Regulating and maintenance services																Cultural services			
		Biomass provisioning	Water supply	Genetic material	Solid waste remediation	Soil and sediment retention	Water purification	Soil quality regulation	Biomass provisioning	Biological control	Air Filtration	Flood mitigation services	Global climate regulation	Local climate regulation	Nursery/habitat maintenance	Noise attenuation	Pollination	Storm mitigation	Water flow regulation	Rainfall pattern regulation	Recreation related services	Visual amenity services	Education/scientific	Spiritual/artistic/symbolic
Energy	Fossil fuels energy production	n. a.	high	n. a.	medium	medium	medium	n. a.	n. a.	n. a.	very low	medium	medium	low	n. a.	low	n. a.	low	high	n. a.	n. a.	n. a.	n. a.	n. a.
	Solar energy production	n. a.	medium	n. a.	n. a.	medium	n. a.	n. a.	n. a.	n. a.	n. a.	medium	very high	medium	n. a.	low	n. a.	medium	medium	n. a.	n. a.	n. a.	n. a.	n. a.
	Wind energy production	n. a.	very low	n. a.	n. a.	medium	n. a.	n. a.	n. a.	n. a.	n. a.	high	very high	medium	n. a.	medium	n. a.	medium	medium	n. a.	n. a.	n. a.	n. a.	n. a.
Constr.	Construction of buildings	n. a.	medium	n. a.	very low	high	medium	n. a.	low	no data	very low	medium	medium	low	n. a.	very low	n. a.	medium	medium	very high	n. a.	n. a.	n. a.	n. a.
	Construction of roads and railways	n. a.	medium	n. a.	no data	high	medium	n. a.	low	no data	very low	high	medium	low	n. a.	very low	n. a.	high	medium	very high	n. a.	n. a.	n. a.	n. a.
Logist.	Passenger air transport	n. a.	very low	n. a.	no data	low	no data	n. a.	very low	very low	very low	medium	low	n. a.	very low	n. a.	low	low	very high	n. a.	very high	n. a.	n. a.	
	Warehousing and storage	n. a.	very low	n. a.	no data	medium	no data	n. a.	n. a.	very low	very low	very low	very low	low	n. a.	no data	n. a.	low	very low	very low	no data	no data	no data	no data
Accom./food	Short term accommodation activities	n. a.	low	n. a.	medium	low	very high	n. a.	no data	very low	very low	very low	very low	low	n. a.	medium	n. a.	low	low	very low	very high	very high	very high	very high
	Food services	n. a.	medium	no data	no data	low	very high	n. a.	no data	very low	very low	very low	medium	low	n. a.	no data	n. a.	low	low	medium	very high	very high	no data	very high
Inf./Comm.	Publishing	n. a.	low	n. a.	n. a.	low	n. a.	n. a.	n. a.	very low	very low	very low	very low	low	n. a.	n. a.	n. a.	very low	low	very low	n. a.	very high	very high	n. a.
	Telecommunications	n. a.	very low	n. a.	n. a.	low	n. a.	n. a.	n. a.	very low	n. a.	medium	very low	low	n. a.	n. a.	n. a.	medium	low	very low	n. a.	n. a.	n. a.	n. a.
	Information service activities	n. a.	very low	n. a.	n. a.	very low	n. a.	n. a.	n. a.	very low	n. a.	very low	very low	low	n. a.	n. a.	n. a.	very low	very low	very low	n. a.	n. a.	n. a.	n. a.
Services	Real estate activities	n. a.	very low	n. a.	n. a.	medium	n. a.	n. a.	n. a.	n. a.	very low	very low	very low	low	n. a.	very low	n. a.	low	very low	n. a.	n. a.	very high	n. a.	n. a.
	Advertising	n. a.	very low	n. a.	n. a.	very low	n. a.	n. a.	n. a.	no data	n. a.	very low	very low	very low	n. a.	very low	n. a.	low	very low	n. a.	n. a.	n. a.	n. a.	n. a.
	Travel agency and tour operator activities	n. a.	very low	n. a.	no data	very low	medium	n. a.	no data	medium	n. a.	very low	medium	low	n. a.	very low	n. a.	very low	very low	medium	very high	very high	very high	very high
	Private security activities	no data	very low	n. a.	n. a.	very low	no data	n. a.	n. a.	no data	n. a.	very low	very low	low	n. a.	n. a.	n. a.	very low	very low	very low	no data	no data	no data	no data
	Cleaning activities	n. a.	medium	n. a.	no data	very low	very low	n. a.	n. a.	no data	n. a.	very low	very low	low	n. a.	n. a.	n. a.	very low	low	very low	n. a.	n. a.	n. a.	n. a.
	Landscape care and maintenance service activities	n. a.	medium	n. a.	low	high	very high	medium	medium	medium	medium	medium	medium	medium	no data	n. a.	very low	medium	medium	medium	no data	very high	very high	very high
	Pre-primary and primary education	n. a.	very low	n. a.	very low	low	no data	n. a.	n. a.	very low	very low	medium	very low	low	n. a.	very low	n. a.	medium	low	very low	very high	very high	very high	very high
	Residential care and hospitals	n. a.	medium	n. a.	medium	low	very high	n. a.	n. a.	very low	very low	high	very low	low	n. a.	very low	n. a.	high	high	n. a.	n. a.	very high	n. a.	very high
	Medical and dental practice activities	n. a.	medium	n. a.	medium	low	very high	n. a.	n. a.	very low	very low	medium	very low	low	n. a.	very low	n. a.	medium	medium	n. a.	n. a.	very high	no data	very high
	Creative, arts and entertainment activities	n. a.	very low	n. a.	very low	very low	very low	n. a.	n. a.	very low	very low	very low	low	low	n. a.	very low	n. a.	very low	very low	very low	no data	very high	n. a.	very high
	Sports activities	low	low	n. a.	medium	very low	very low	very low	n. a.	very low	low	medium	medium	low	very low	very low	n. a.	medium	medium	medium	very high	very high	n. a.	n. a.

The manufacturing sector, too, is directly dependent on nature – particularly on a sufficient supply of water of the proper quality. Furthermore, nature's protective functions in relation to flooding, storms and local climate regulation (including protection against heat) are particularly important for industry, as is waste recycling. The pharmaceutical industry is also particularly reliant on genetic resources and research-related cultural achievements – a large proportion of medicinal substances originate from nature. Damage to the relevant ecosystems can result in significant costs and production losses.

Whereas fossil-fuel-based electricity generation is heavily dependent on water-related environmental services, solar and wind power generation is particularly reliant on global climate regulation. The catering industry, hotels, travel agencies and aviation depend on the wide range of cultural environmental services. This also applies to other branches of the service sector, such as education and childcare, health services, and the sports and entertainment sectors. The property sector is heavily dependent on aesthetic values. Regulatory services such as natural water purification, waste disposal and nature's protective functions are important for the entire service sector.

A striking result of this dependency analysis is that a wide range of sectors are at least

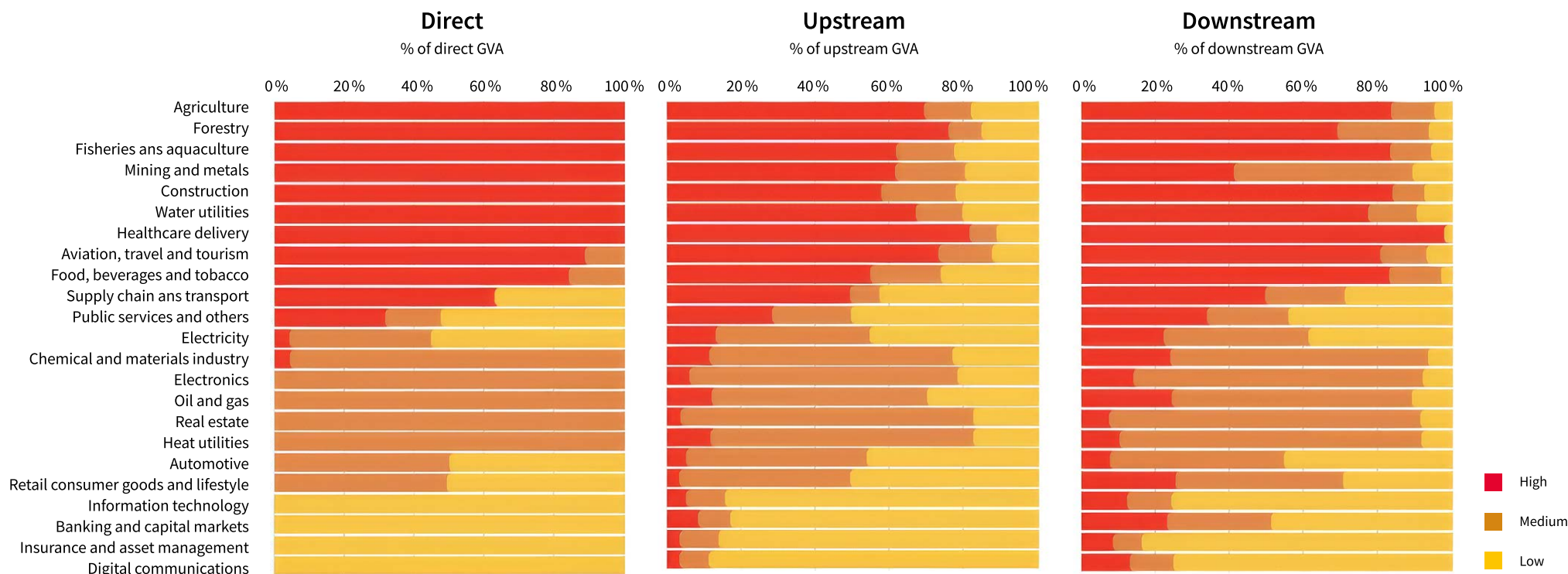
dependent on water-related ecosystem services, and that water-related dependencies are the highest across ecosystem services in many sectors. Large parts of the economy show at least moderate direct dependency on water supply, water purification, flood mitigation, and water flow regulation. Water purification, e. g., is of the utmost importance for all of agriculture and forestry; the same holds for rainfall pattern regulation, which is also indispensable for construction and air transport. All cultural ecosystem services are very closely linked to freshwater bodies and are of very high importance for many service sectors with low dependency on provisioning and regulating ecosystem services.

The potential consequences of this direct dependence on ecosystem services (and biodiversity) are far-reaching. However, this is not the whole picture, as large parts of the economy are indirectly dependent on ecosystem services via supply and value chains. Ultimately, all sectors are linked to those sectors directly dependent on nature in two ways: as consumers of their products and services (downstream) and as part of the upstream supply chain. These dependencies are not always obvious and can have an impact over long distances and over extended periods of time.

Economic modelling studies combine materiality ratings with input-output-tables capturing

granular upstream and downstream linkages between economic subsectors. This approach allows the analysis to assess how ecosystem-service dependencies are transmitted across sectors and throughout the economy. A recent study by the Joint Research Centre of the European Commission (Hirschbühl et al., 2025) uses ENCORE materiality ratings and EXIOBASE input-output tables to model the direct, upstream and downstream dependencies for different sectors of the EU economy depicted in [Fig. 5.4](#). The report finds that both upstream and downstream dependencies are distributed much more evenly across economic sectors than direct dependencies, which are heavily concentrated in primary sectors and parts of the manufacturing industry. When considering the value chain, for example, IT and financial services are much more dependent on ecosystem services.

Aggregating direct and indirect dependencies (only upstream to avoid double-counting), Hirschbühl et al. (2025) conclude that approximately two-thirds (65 %) of the EU's gross value added is significantly dependent on ecosystem services – 36 % of which is highly dependent and a further 29 % is moderately dependent. The global economy shows a similar share of highly dependent value added (33 %); for China and the US, the figures are 35 % and 22 %, respectively. This is in range with similar studies, e. g., WEF, (2020) estimated in a widely cited study



that more than half of global GDP depends strongly or moderately on ecosystem services.

The results should be interpreted as indicative estimates of the order of magnitude rather than as precise quantitative estimates. They depend on methodological assumptions regarding dependency ratings, sectoral aggregation, and the definition of thresholds. Because economic sectors are interconnected through supply and value chains, the entire economy is potentially exposed to nature-related dependencies and

risks, although the degree of dependency varies considerably across sectors. Accordingly, the IPBES emphasises in its latest “Business and Biodiversity Assessment” that 100 % of businesses not only impact biodiversity but are also dependent on it (IPBES, 2026).

Fig. 5.4 The Figure illustrates the EU economy’s dependencies on nature, distinguishing between direct, upstream, and downstream dependencies. The highest direct dependencies occur in sectors that rely heavily on water, particularly agriculture and forestry. The banking sector has relatively low direct dependencies on nature. However, its upstream and downstream dependencies are substantially higher. This indicates that banks are indirectly exposed to ecosystem dependencies through their economic activities and value chains. (Source: Hirschbühl et al., 2025)

5.3.3 Implications of nature-related risks for the financial sector

For businesses, financial institutions, and economies, the degradation of natural capital can translate into a broad range of risks, including physical risks, transition risks (such as revenue losses resulting from shifts in market demand), and systemic risks (TNFD, 2023b; WRI, 2025). In the context of land degradation, these risks include, inter alia (ELD, 2026):

- ▶ Supply chain disruptions: Industries that are highly dependent on land and water resources may be particularly vulnerable to further ecosystem degradation and disruptions in the availability of critical inputs.
- ▶ Resource scarcity: Increasing scarcity or reduced availability of water, raw materials, and ecosystem services can constrain economic activity and increase production costs.
- ▶ Increasing costs for land-intensive industries: Businesses may face higher costs associated with ecosystem restoration, regulatory compliance, adaptation, and risk mitigation.
- ▶ Reputational and regulatory risks: Environmental degradation may increase the risk of reputational damage, regulatory sanctions, and loss of investment, particularly for

companies and financial portfolios subject to environmental, social, and governance (ESG) considerations.

- ▶ Climate and disaster risk amplification: Land and ecosystem degradation can reduce the capacity of ecosystems to buffer environmental hazards and may therefore increase vulnerability to climate-related and other natural hazards.
- ▶ Social and geopolitical risks: Land degradation can contribute to food insecurity, livelihood losses, migration, and social tensions, potentially exacerbating existing inequalities and contributing to conflict.
- ▶ Climate and disaster risk amplification: Land and ecosystem degradation can reduce the capacity of ecosystems to regulate water flows, retain soil, store carbon, and buffer environmental hazards, thereby increasing vulnerability to climate-related and other natural hazards.

These risks may arise across sectors that are exposed to land-use change, water stress, and/or the degradation or reduced availability of ecosystem services. Sectors with high dependencies on both water resources and ecosystem services, such as agriculture, are particularly vulnerable (see [Chapter 5.3.2](#)).

The increasing interdependencies associated with the water crisis – including more frequent droughts and extreme precipitation events – as well as ecosystem degradation and biodiversity loss, are becoming increasingly relevant for the financial sector. This is primarily because financial activities often involve long-term time horizons, while the degradation of natural systems can create persistent and cumulative risks. Lending decisions, infrastructure investments, development projects, and related financing activities often extend over several years or even decades. As a result, financial institutions may be exposed to nature-related risks over extended periods through the economic activities and projects they finance.

Water scarcity, ecosystem degradation, and climate-related disruptions can affect economic activities, asset values, and investment portfolios, with significant implications for financial institutions (WEF, 2023; WWF, 2026a). Since agriculture, industry, energy production, trade (i. e. economic growth), and also public health depend on reliable water resources and functioning ecosystems, nature-related risks can translate into financial impacts through both physical and transition risk channels. The Network for Greening the Financial System (NGFS, 2023) highlights the growing relevance of water- and nature-related risks for financial

stability and for the work of central banks, financial supervisors, and regulatory authorities.

The economic risks associated with water insecurity and ecosystem degradation are becoming increasingly measurable and quantifiable. With regard to water, research highlighted by the World Wide Fund for Nature (WWF, 2026a) indicates that severe water-related disruptions could threaten up to 15 % of economic output in the Eurozone, reflecting the strong dependence of companies and financial assets on freshwater ecosystem services. Furthermore, the World Resources Institute (WRI, 2023) estimates that a substantial share of global GDP could be generated in regions exposed to high levels of water stress by 2050. These projections illustrate the growing economic significance of water scarcity, drought risks, and declining ecosystem resilience for businesses, investors, and financial institutions.

Water- and ecosystem-related disruptions can contribute to inflationary pressures and increased price volatility whenever the impairment of ecosystem services leads to losses in productivity, thereby driving up prices. The most significant transmission channel is agricultural production, which is coming under increasing pressure from soil erosion, salinisation, drought and the loss of natural pollinators. For example, a 12 % decline in productivity by 2040 due to soil

degradation could lead to a 30 % rise in prices. A study by the Bank of France shows that a temporary crop failure can increase the inflation rate by approximately 0.5 percentage points beyond the year in which it occurs. Around 30 % of medium-term inflation volatility in the Eurozone is attributable to global crop failures.

Other transmission channels are constrained manufacturing activities and electricity generation due to environmental shocks as well as disrupted transportation networks that affect availability and prices of commodities (Wegner et al., 2025). The European Environment Agency (EEA, 2024) highlights that climate-related risks, including droughts and water scarcity, can be expected to increasingly affect economic sectors and supply chains across Europe.

Similarly, the WWF (2026a) emphasises that water-related disruptions can generate broader economic consequences, including impacts on food production and global trade. The drought in the Po River basin in Italy in 2022, which significantly affected agricultural production, including rice cultivation, and drought-related restrictions on shipping through the Panama Canal illustrate how regional water constraints can propagate through international supply chains and contribute to price volatility. Another example is flood damage at a Slovenian automotive components supplier's plant in 2023

leading to delays in the production of 150,000 vehicles and losses in the mid three-digit million range.

Water- and ecosystem-related risks are increasingly relevant for fiscal stability, economic performance, and financial systems. Countries with a strong economic dependence on water-intensive sectors or high exposure to floods, droughts, and chronic water stress may face growing fiscal pressures. These pressures can arise from higher adaptation and disaster-response costs, reduced public revenues, disruptions to economic activity, and increased government expenditure. Moody's Investors Service (2023) estimates that around one-third of the assessed sovereign states already face material increases in nature-related risks (Moody's Investors Service, 2023).

Beyond their impact on economic output, water-related risks are increasingly relevant for sovereign creditworthiness. Moody's Investors Service (2023) highlights that water stress can contribute to sovereign credit risk by weakening economic resilience and increasing fiscal vulnerabilities. (Moody's Investors Service, 2023). Similarly, the WWF (2026a) emphasizes that water-related risks are becoming increasingly relevant for financial markets and sovereign risk assessments, demonstrating the growing importance of water security for public

finances. A study by Agarwala et al. shows that the creditworthiness of poor countries, such as Madagascar, the Democratic Republic Congo, Bangladesh, Angola, Egypt, or India may lead to an increasing probability of default and thus reduce creditworthiness (Agarwala et al., 2026).

Rising temperatures, droughts, and heatwaves can reduce labour productivity and working capacity, particularly in climate-sensitive sectors such as agriculture, construction, and manufacturing. The International Labour Organisation highlights that heat stress can significantly impair workers' productivity and working conditions, especially in outdoor and physically demanding occupations (ILO, 2019). At the same time, declining water quality can increase health risks through the spread of infectious diseases and antimicrobial resistance, which the World Health Organisation (WHO, 2014) identifies as a growing global health and economic challenge.

These impacts are not isolated but can propagate through the economy through physical risks, including floods, droughts, water scarcity, and ecosystem degradation, as well as through transition risks arising from regulatory changes, technological developments, litigation, and changing market expectations. Ultimately, these risks can affect companies through asset damage, operational disruptions, and supply-chain interruptions, and can translate into credit,

market, liquidity, and insurance risks for financial institutions (NGFS, 2023).

The economic impacts described above are not isolated events but are interconnected and can have systemic effects. Water-related shocks can propagate through the economy and ultimately affect financial stability. Physical risks arise from floods, droughts, water scarcity, groundwater depletion, and declining water quality, while transition risks may result from regulatory changes, technological innovation, litigation, and shifting market and consumer preferences. As highlighted by the Network for Greening the Financial System (NGFS, 2023) these risks can affect firms through asset damage, operational disruptions, reduced productivity, and supply-chain interruptions, and can subsequently be transmitted to the financial sector through credit, market, liquidity, and insurance risks. Similarly, the Taskforce on Nature-related Financial Disclosures (TNFD, 2023b) emphasises that dependencies and impacts on nature, including freshwater ecosystems, can create material financial risks with potentially systemic implications across the economy.

Central banks increasingly regard the economy's dependence on ecosystem services and water resources as a systemic financial risk: If disruptions to production processes impair companies' ability to repay their loans, this creates risks for

the banks and, consequently, for the financial system. Building on the combination of sector materiality ratings and input-output models (see [Chapter 4.3.2](#)), an ECB study (ECB, 2025) analyses the implications of nature-related dependencies for the European financial sector by combining the economic dependencies with sector-specific credit data. According to ECB (2025), approximately 72 % of euro-area companies are highly dependent on at least one ecosystem service, creating indirect exposure for banks through their lending activities. This corresponds with 75 % of the analysed corporate loan volume (€ 3.2 trillion). The strongest dependencies identified in banks' credit portfolios relate to surface-water provision, groundwater provision, erosion control, and flood protection, highlighting the financial relevance of water-related ecosystem risks—not only in Europe, but also significantly in Northern America and Asia ([Fig. 5.5](#)). In the eurozone, 60 % of loans are exposed to flood risks at the respective company's location. For Germany, the Deutsche Bundesbank (2026) estimates in a similar analysis that 52 % of the corporate loan volume is highly dependent on at least one ecosystem service. Water-related ecosystem services prove to be particularly important: the exposed credit volume in Germany also depends almost entirely on water supply and flood protection (51 % of the total) (Deutsche Bundesbank, 2026).

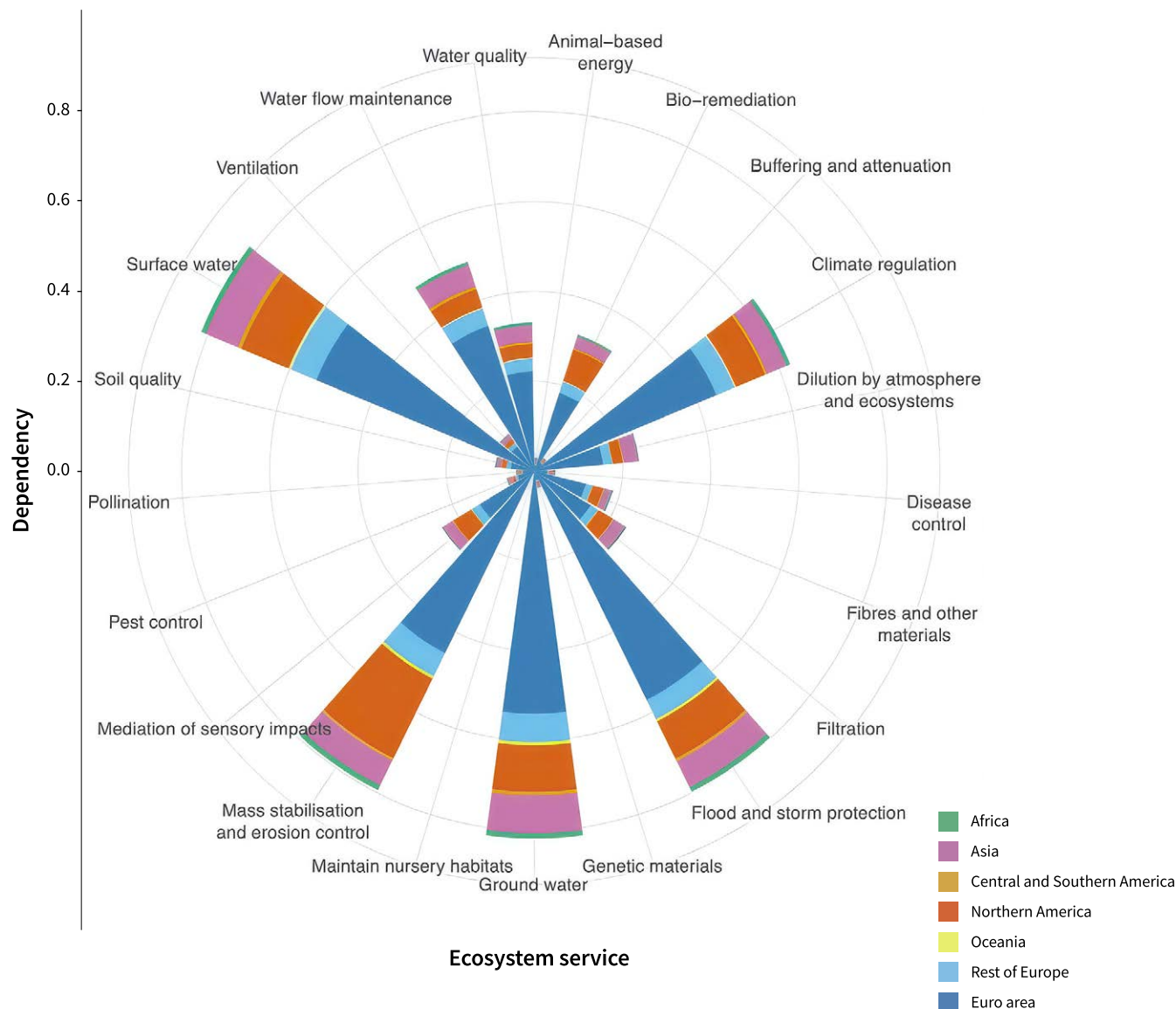


Fig. 5.5 Dependencies of banks on ecosystem services are illustrated for 21 ecosystem services. The highest dependencies relate to climate regulation, flood and storm protection, ground water, mass stabilisation and erosion control, and surface water. Highest dependencies affect the euro area (Source: ECB, 2025).

The ten euro area banks with the largest biodiversity footprints account for almost 40 % of the total biodiversity impact associated with corporate lending, and the 100 most impactful banks account for approximately 87 % (ECB, 2023). This concentration suggests that targeted supervisory and policy measures could have a significant impact on the management of nature-related financial risks.

A key contribution of Ceglar et al. (2025, ECB) is their analysis of compound climate-nature risks (ECB, 2025). The authors show that biodiversity loss and climate change should not be assessed in isolation, as they interact through reinforcing mechanisms that can amplify economic and financial risks. Water-related ecosystem services illustrate these interdependencies: healthy ecosystems can reduce flood and drought risks by regulating hydrological processes, whereas ecosystem degradation can weaken this buffering capacity. The study finds substantial exposure of the euro area banking sector to these compound risks. More than 60 % of the analysed corporate loan portfolio is associated with companies operating in areas where the demand for flood protection is not fully met, while more than 40 % is associated with companies that are highly exposed to drought risk and dependent on surface-water provision (ECB, 2025).

The private sector, including the financial sector, is increasingly recognising the importance of building resilient land and water systems and is beginning to take steps to address related risks. An increasing number of investors recognise water- and biodiversity-related risks as financially material, as these risks can affect corporate valuations, the cost of capital, and long-term investment performance (PRI, 2023; TNFD, 2023a). However, this trend is unfolding against a backdrop of growing political and regulatory resistance to environmental policies and ambitions, which may slow or constrain further progress.

As of 2025, approximately one in six US dollars of global assets under management (AUM) is or will be subject to nature-related risk assessment aligned with the Taskforce on Nature-related Financial Disclosures (TNFD, 2025). By November 2025, more than 730 organisations had committed to TNFD-aligned reporting, representing more than US\$ 22 trillion in assets under management – only four years after the TNFD was established. The Network for Greening the Financial System (NGFS), an international network of central banks and financial supervisors, has also explicitly recognised nature-related risks and biodiversity loss as relevant to financial stability and systemic risk (NGFS, 2024). In addition, companies including multinational corporations such as Kering, Ørsted, Danone,

L'Oréal, Nestlé, and Unilever are increasingly setting targets related to the sustainable use of natural resources (ELD, 2026).

5.3.4 Economic implications of interdependencies between water, biodiversity, and climate

As this report focuses primarily on the nexus between freshwater resources, ecosystem services, and biodiversity, it is important to recognise the strong interdependencies among these three elements. Their interactions can reinforce one another through cascading ecological and economic feedback loops, creating increasingly significant economic challenges. Water is often a critical limiting factor: when water becomes scarce, ecosystems can degrade, reducing their capacity to provide essential ecosystem services (see [Chapter 2](#) and [Chapter 3](#)). Yet this relationship is bidirectional: biodiversity is itself essential for maintaining ecosystem functioning, regulating the water cycle, and sustaining healthy freshwater systems. The following examples illustrate these interdependencies.

A key consequence of biodiversity loss and ecosystem degradation is the impairment of natural water regulation. Ecosystems function as natural infrastructure by storing water, retaining it within the landscape, and gradually releasing

it to rivers and groundwater. When forests are cleared, peatlands drained, and wetlands degraded, this natural water-storage capacity is reduced or lost. As a result, precipitation can run off more rapidly as surface water, groundwater recharge can decline, and periods of drought can become more frequent or severe.

These interdependencies are particularly evident in drained peatland landscapes. Although peatlands cover only around three percent of the Earth's land surface, they store vast quantities of water and play a crucial role in regulating local hydrological systems (IPCC, 2019, 2022a; Leopoldina, 2024; UNEP, 2022). In addition, they represent the world's largest terrestrial carbon store (IPCC, 2022a). Their drainage not only results in biodiversity loss and substantial greenhouse gas emissions but also increases the risk of drought and land degradation, thereby further accelerating climate change. As peatlands lose their natural 'sponge function', their capacity to retain and gradually release water is severely reduced (IPCC, 2019, 2022a; UNEP, 2022). Over the long term, peatland drainage can contribute to soil degradation, declining water availability, and – particularly in drought-prone regions – to desertification processes that reduce agricultural productivity and increase irrigation demand (Leopoldina, 2024).

Agriculture is particularly dependent on the water–biodiversity nexus (FAO, 2019; IPBES, 2019). On the one hand, agricultural production requires reliable and sufficient water resources; on the other, its productivity depends on the functioning of healthy ecosystems. Pollinators such as wild bees and butterflies, soil fauna, microorganisms, and aquatic communities provide essential ecosystem functions that underpin food production. In turn, these organisms and ecological communities are themselves highly dependent on the availability and quality of water. Estimates suggest that approximately three-quarters of the world’s leading food crops depend at least partly on animal pollination (IPBES, 2016; Klein et al., 2007). At the same time, pollinator diversity is declining due to habitat loss, pesticide use, and changes in hydrological conditions (FAO, 2019; IPBES, 2016).

Nature-based flood protection measures are gaining increasing attention as a cost-effective strategy for climate adaptation. International studies show that restoring floodplains, wetlands, and natural retention areas often provides more favorable cost–benefit ratios than relying solely on engineered flood protection infrastructure. Beyond reducing flood damage, these measures simultaneously improve water quality, enhance biodiversity, and increase carbon sequestration, thereby generating multiple societal and economic co-benefits (“multiple

dividends”). The recent IPBES Nexus Assessment (IPBES, 2024) identifies such nature-based solutions as a key strategy for simultaneously addressing interconnected biodiversity, water, and climate risks (see also Gallois et al., 2026).

In conclusion, biodiversity conservation, ecosystem restoration, climate adaptation, and financial stability should be addressed through integrated policy frameworks rather than as separate policy objectives. Healthy ecosystems underpin essential services such as water provision, flood protection, water quality, and climate regulation, creating important links between environmental integrity and economic resilience. Investments in ecosystem conservation and restoration can therefore generate both ecological and economic benefits while reducing exposure to climate- and nature-related risks. Recognizing the interconnectedness of climate change, biodiversity loss, and financial risk is essential for developing effective policy responses. An integrated approach can strengthen economic resilience, safeguard financial stability, and contribute to long-term sustainable development. Taking these interdependencies into account is therefore also crucial from an economic perspective. An integrated assessment of water, ecosystem, and biodiversity dynamics can help identify measures that generate multiple environmental and economic co-benefits, improve cost-effectiveness, and avoid higher future

costs associated with ecosystem degradation, resource scarcity, and the need for more costly corrective interventions.

At the same time, it is important to recognise that the dependence of economies and societies on nature, as well as the associated nature-related risks, is highly context-specific. These dependencies and risks vary considerably across regions and economic sectors, reflecting differences in geographical location, ecological conditions, climatic characteristics, economic structures, and societal circumstances. Consequently, there is rarely a one-size-fits-all solution. Effective adaptation and risk management therefore require locally tailored and context-specific strategies that account for regional vulnerabilities, dependencies, and opportunities. This also applies to regions in Germany that have traditionally been regarded as comparatively less vulnerable to climate change and the water and biodiversity crisis, and better able to adapt. (see [Box 7](#) for a case study from Germany demonstrating the need for comprehensive and proactive adaptation measures).

Box 7: Future irrigation expansion may lead to an irrigation trap – evidence from Germany

Bernd Hansjürgens

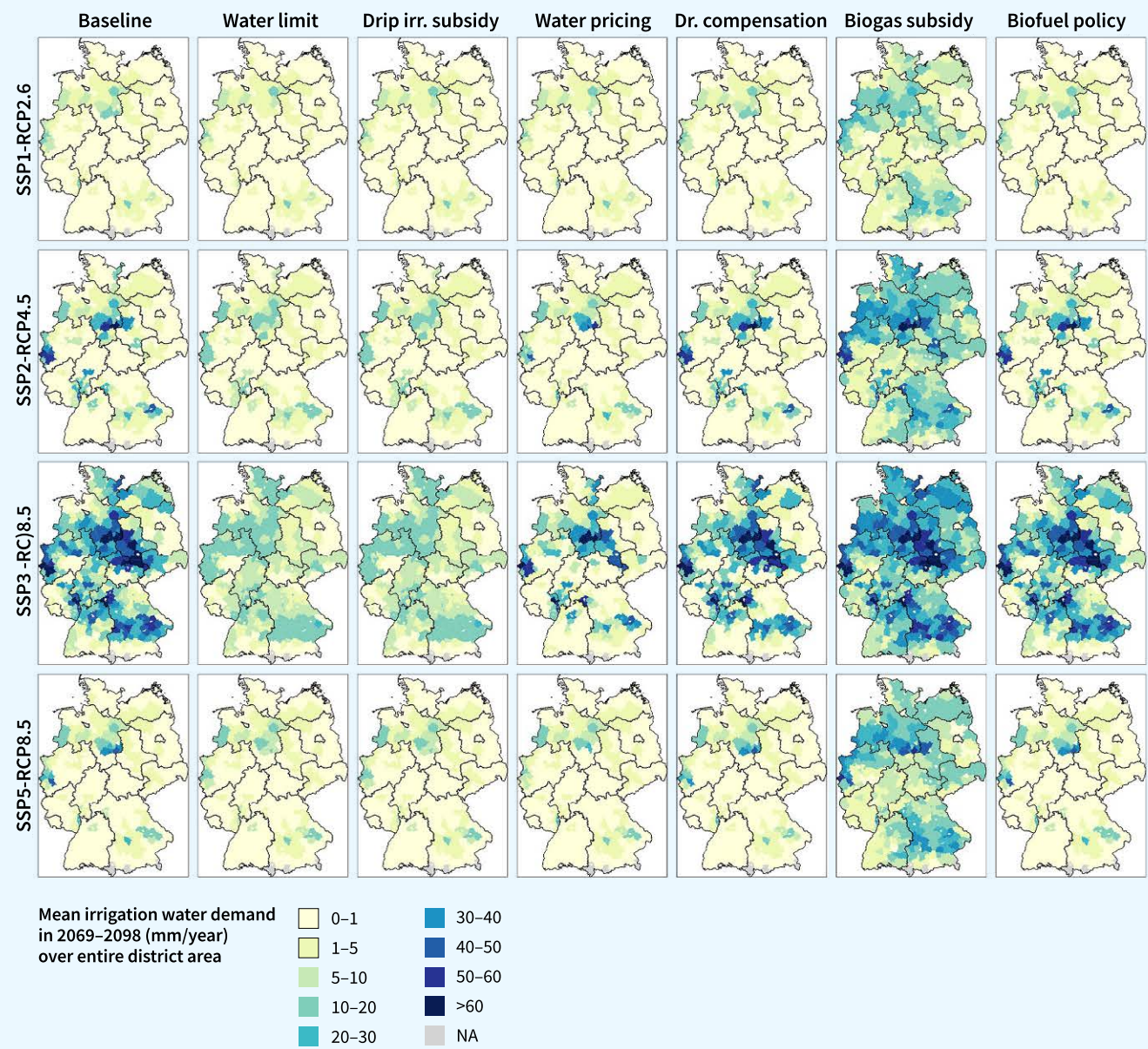
Climate change is expected to increase the frequency and severity of droughts and heatwaves not only in regions such as the Mediterranean, but also in Germany. This will pose growing challenges, particularly for agriculture. As rainfall patterns become less reliable and periods of water scarcity more frequent, farmers may increasingly rely on irrigation to maintain crop production. The extent of these pressures will also depend on land-use and policy choices. In Germany, for example, increased bioenergy cultivation, the introduction of water pricing, or subsidies for bioenergy could alter patterns of land use and water demand. Depending on how these measures are implemented, they may have different implications for irrigation requirements and overall water scarcity. At the same time, increasing irrigation can intensify competition for limited water resources among agriculture, public water supply, industry, energy production, and ecosystems. This may, in turn, place additional pressure on freshwater ecosystems and biodiversity (Heilemann et al., 2025).

Research highlights that the unregulated expansion of irrigation can create an ‘irrigation trap’, characterised by increasing trade-offs and conflicts among food production, water resource conservation, and energy objectives (Heilemann et al., 2026). These findings underscore the need for integrated and coordinated policy approaches across sectors to ensure that agricultural adaptation to increasing drought risks remains sustainable. Without such approaches, greater reliance on irrigation may reinforce long-term dependence on increasingly scarce water resources and further intensify competition among different water users.

While irrigation intensity is projected to increase under climate change across all future scenarios, the extent of irrigation expansion and the resulting total water demand vary considerably. Heilemann et al. (2026) demonstrate that water management measures, such as restrictions on water withdrawals and water pricing, are most effective

when implemented at an early stage. These measures can substantially slow the growth in irrigation demand while having only limited effects on farm profitability (Heilemann et al., 2026). By contrast, policies that promote bioenergy production may significantly increase irrigation demand, thereby intensifying pressure on already constrained water resources. The study further shows that drought compensation payments can weaken farmers’ incentives to adopt climate adaptation measures, while subsidies for more efficient irrigation technologies alone do not necessarily reduce overall water consumption. Taken together, these findings highlight the need for coordinated policy packages that combine demand-side water management with incentives for sustainable agricultural adaptation, rather than relying on individual measures in isolation.

Fig. Box 7 In the long term, water scarcity in Germany is also expected to increase significantly under different land-use and water-use scenarios (rows: Shared Socioeconomic Pathways, SSP), and by assumptions regarding bioenergy cultivation, water prices, and subsidies for bioenergy (columns). (Source: Heilemann et al., 2026)



5.4 The economic value of freshwater resources and biodiversity

5.4.1 The Total Economic Value as a comprehensive valuation approach

Economic valuation is one of the most established approaches for making the benefits of freshwater resources, biodiversity, and ecosystem services visible. Its basic rationale is that freshwater resources and biodiverse ecosystems contribute to human well-being and that their value becomes particularly relevant where choices have to be made between alternative uses of scarce resources. Economic valuation therefore seeks to express environmental benefits in terms of the trade-offs people are willing to make to obtain, maintain or improve them (Bateman et al., 2011; Pascual et al., 2011). It has become an important component of broader ecosystem-assessment initiatives such as the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2005; The Economics of Ecosystems and Biodiversity (TEEB, 2010), and the Dasgupta Review 2021 (Dasgupta, 2021).

A common criticism states that economic valuation reduces the value of nature to the preferences of self-interested individuals and ultimately to monetary values. This characterisation, however, overlooks the breadth of the

economic valuation framework. In particular, the established concept of Total Economic Value (TEV) distinguishes between use and non-use values and therefore covers a much wider range of human relationships with nature than direct market consumption alone (Pascual et al., 2011; Turner et al., 2003)(see [Fig. 5.6](#)).

Direct use values arise from actual interactions with ecosystems and their outputs, such as water use, food, timber and other products (consumptive), as well as recreation and inspiration (non-consumptive). Indirect use values arise from ecosystem functions and regulating services that contribute to human well-being, including climate regulation, water purification, pollination, soil formation and protection against natural hazards such as droughts or flooding. TEV also encompasses non-use values, which are not associated with actual or planned use of an ecosystem. Existence value refers to the value people attach to knowing that a species or ecosystem continues to exist, while bequest value reflects the importance attached to preserving environmental assets for future generations. Option value refers to the value of retaining the possibility of using ecosystem services in the future, even when they are not

currently used. The potential future benefits associated with the genetic diversity of ecosystems such as the Amazon rainforest provide an illustrative example. Economic valuation can therefore also address benefits that are not associated with direct use or market consumption (TEEB, 2010).

The inclusion of non-use values is particularly important because willingness to pay need not reflect narrow self-interest. Individuals may support environmental protection because they care about other people, future generations or the continued existence of a species. Economic and environmental valuation research has described such motivations in terms of altruism, ethical commitments and the satisfaction associated with contributing to the provision or protection of environmental goods (Hansjürgens et al., 2016; Spash, 2000). A monetary valuation should therefore not automatically be interpreted as a measure of purely private consumption benefits. Rather, stated willingness to pay may incorporate concerns for others, future generations and the preservation of environmental resources independent of direct personal use (Hansjürgens et al., 2016).

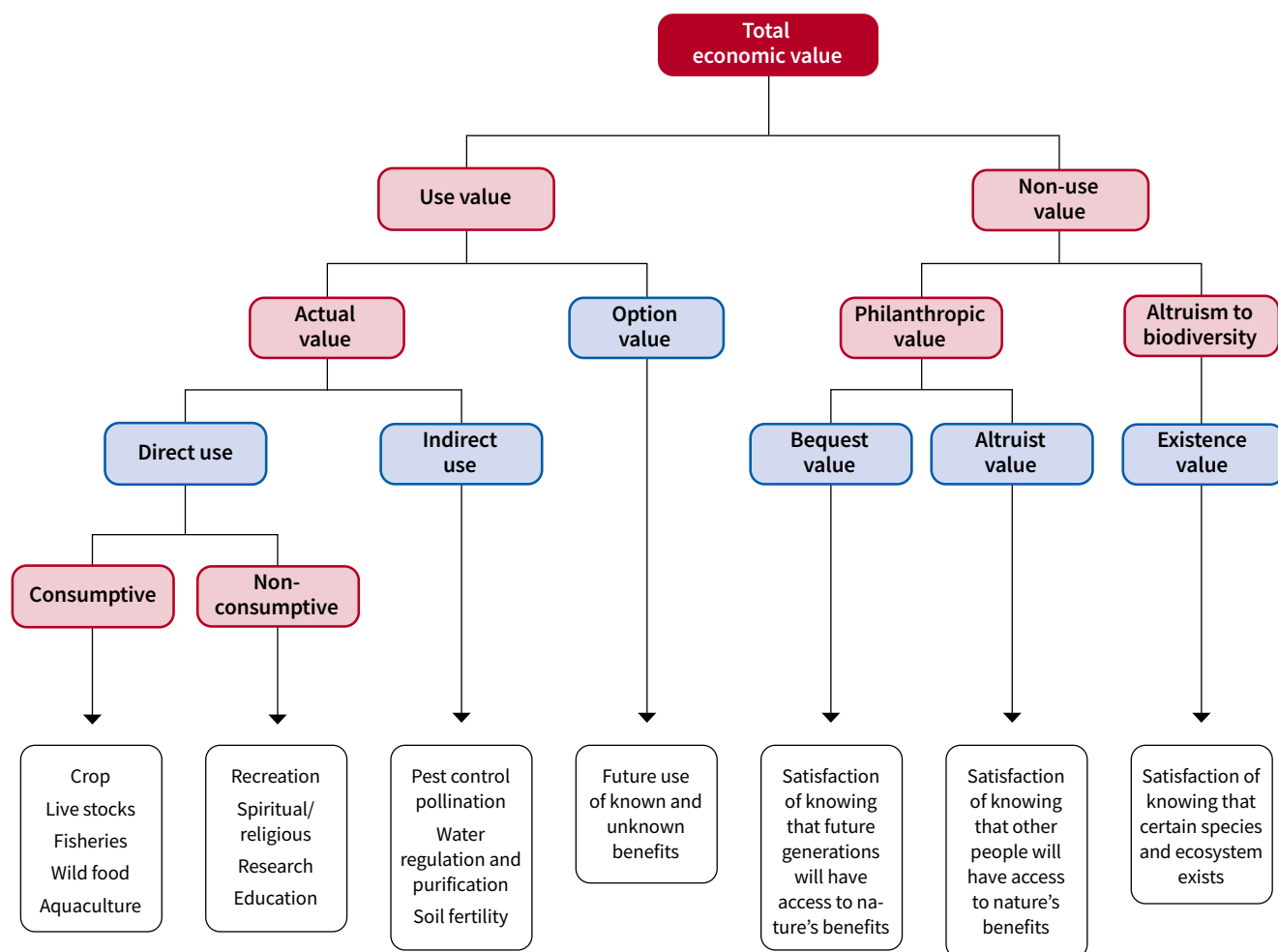


Fig. 5.6 Total Economic Value. The comprehensive economic valuation approach does not only cover direct and indirect use values, but also non-use values like bequest value, altruistic values, and existence values. The option value lies between use and non-use values. (Re-drawn after Pascual et al., 2011)

This broader understanding of economic values is consistent with the IPBES conceptualisation of values. IPBES distinguishes between instrumental, relational and intrinsic values of nature (Díaz et al., 2015; IPBES, 2022). Economic values are primarily located within the sphere of instrumental values because they – as an anthropocentric concept – concern the contribution of nature to human well-being and preferences. However, IPBES does not equate instrumental values with narrowly material or selfish interests. Instrumental values may include values associated with direct and indirect uses as well as non-use values. Economic valuation can therefore provide information about a considerably wider range of human valuations of nature than market prices alone suggest (IPBES, 2022).

The distinction between “I-preferences” and “We-preferences” further illustrates this point (Vatn, 2009). Individual preferences may include concerns for other people, communities, future generations and broader social outcomes. The aggregation of individual preferences therefore does not necessarily imply the aggregation of narrowly selfish interests. Kenter et al. (2015) emphasise that personal, social and shared values may coexist and interact (Kenter et al., 2015). Economic valuation can consequently capture some social and other-regarding concerns when individuals incorporate them into their preferences (Hansjürgens et al., 2016).

Nevertheless, such values remain represented indirectly through individual preferences rather than necessarily constituting an independently articulated social value.

Similarly, the IPBES framework further emphasises that values are not only diverse but can also be relational. Relational values concern meaningful relationships between people and nature and between people through nature, including notions of identity, responsibility, care, stewardship and a good life (Díaz et al., 2015; IPBES, 2022). Some of these considerations may enter economic valuation indirectly when individuals express them as preferences. However, their meaning is not necessarily exhausted by the resulting willingness-to-pay estimate. A landscape, river or species may, for example, be valued because it contributes to a community's identity or because people feel a responsibility to protect it. Such values may be partly reflected in individual economic preferences, but they may also require social, cultural or deliberative forms of valuation to become fully visible (IPBES, 2022; Kenter et al., 2015).

The breadth of economic valuation should therefore be understood in a differentiated way. The TEV framework covers direct and indirect use values as well as non-use values. Individual preferences may incorporate altruistic, other-regarding and intergenerational concerns, while

deliberative approaches can allow preferences to be informed and reconsidered through communication and reflection. From the IPBES (2022) perspective, economic valuation captures an important part of the instrumental value of nature and may indirectly reflect some relational concerns, but it represents only one part of the broader diversity of values associated with nature (IPBES, 2022).

5.4.2 Freshwater-related economic valuation studies

Empirical studies of freshwater ecosystems illustrate how these different dimensions of value can become relevant in practice. Dehnhardt and Meyerhoff (2007), for example, examined the economic value of ecological services provided by riparian wetlands and floodplains along the River Elbe in the context of the European Water Framework Directive. Their analysis showed that riparian wetlands provide significant benefits that should be taken into account in river-basin management. Neglecting such benefits can bias cost-benefit analyses and potentially lead to inappropriate policy decisions (Dehnhardt & Meyerhoff, 2007).

A related study by Grossmann (2012) examined the economic value of nutrient retention provided by restored floodplain wetlands in the

Elbe River basin. Using a cost-minimisation model, the study estimated the economic value of phosphorus and nitrogen retention and assessed the costs and benefits of 45 potential restoration sites. The results showed that, under favourable conditions, nutrient-retention benefits alone could be sufficient to generate an economic efficiency gain from floodplain restoration. At the same time, Grossmann emphasised that restoration projects should be assessed as multifunctional interventions, with improvements in water quality representing only one of several benefit dimensions (Grossmann, 2012).

A particularly illustrative example is provided by the restoration of the Emscher River and its tributaries in Germany. Gerner et al. (2018) assessed a range of ecosystem services associated with the restoration, including self-purification, habitat provision and flood protection, as well as cultural services such as aesthetic, recreational, educational and existence values. These services were monetised using different economic methods, including avoided damage costs, contingent valuation and benefit transfer. The study estimated a direct economic impact of approximately € 21.4 million per year and an additional non-market value of approximately € 109.1 million per year associated with non-use values (Gerner et al., 2018). The case demonstrates particularly well that economic valuation

can encompass both market-related benefits and values arising from people's concern for an ecosystem that they may not directly use.

An earlier study of the Danube floodplains provides another example of the broader economic valuation of freshwater wetlands. Gren et al. (1995) estimated the economic value of several ecosystem services provided by Danube floodplains, including recreation, nutrient purification and the provision of natural resources. Their analysis estimated an average annual value of approximately ECU 374 per hectare and an aggregate value of around ECU 650 million per year for the floodplains considered. Nutrient purification accounted for approximately half of the estimated total value. The authors stressed, however, that the estimates were based on simplified assumptions and benefit-transfer methods and should therefore be interpreted with caution (Gren et al., 1995). Nevertheless, the study illustrates that the economic value of floodplains extends beyond direct resource use to include regulating and recreational ecosystem services.

A further prominent example is provided by New York City's Catskill–Delaware watershed. Rather than constructing a large-scale filtration facility to replace the natural water purification functions of the watershed, New York City chose to invest in watershed protection and restoration.

In the 1990s, the estimated cost of constructing and operating a filtration system was approximately US\$ 6–8 billion, whereas investments in protecting and restoring the watershed were estimated at roughly US\$ 1–1.5 billion (Chichilnisky & Heal, 1998; National Research Council, 2000). The case is particularly relevant for economic valuation because it illustrates a replacement-cost approach: the value of the watershed's water purification service can be approximated by considering the cost of replacing that service with a technical alternative. At the same time, watershed protection generated additional ecological benefits, including the protection of biodiversity and other ecosystem functions. The Catskill case illustrates how economic valuation can make otherwise invisible ecosystem services relevant to infrastructure and investment decisions.

5.4.3 Limits of the economic valuation approach

The breadth of economic valuation should nevertheless not be confused with completeness. Economic values remain fundamentally anthropocentric because they represent values attributed by human beings. They are also context-dependent and generally marginal: they describe the value of changes in environmental quality relative to alternatives rather than an

absolute value of nature as a whole (Bateman et al., 2011; Pascual et al., 2011). This distinction is particularly important for ecosystems characterised by uncertainty, ecological thresholds, irreversibility or critical natural capital, where marginal values may change substantially as ecological conditions deteriorate (Farley, 2008).

The distinction between marginal and total value also helps explain the controversy surrounding Costanza et al. (1997). Their influential attempt to estimate the global economic value of ecosystem services demonstrated the enormous contribution of ecosystems to human welfare. However, interpreting the resulting aggregate figure as the absolute 'value of nature' would be misleading. Economic valuation is generally more meaningful when applied to changes, alternatives and trade-offs within a specific decision context than when used to assign a single monetary value to the natural world as a whole (Costanza et al., 1997; Farley, 2008). A further limitation concerns monetary commensurability. Monetary valuation is attractive in policy analysis because it provides a common metric for comparing costs and benefits. Yet not all environmental values can necessarily or appropriately be reduced to a monetary scale.

These limitations are particularly relevant in light of the IPBES Values Assessment. IPBES (2022) explicitly rejects the idea that a single

valuation method can adequately represent the full diversity of values associated with nature. Different people and societies may hold different values, rooted in different worldviews, cultural traditions and knowledge systems. Consequently, the choice of valuation method is itself consequential: it determines which dimensions of value are made visible and which may remain underrepresented (IPBES, 2016, 2022). IPBES therefore calls for greater integration and, where appropriate, bridging among different valuation approaches rather than the exclusive use of monetary metrics.

The distinction between instrumental, relational, and intrinsic values is central to this argument. Economic valuation is particularly well-suited to expressing instrumental values because it assesses how changes in nature affect human well-being and the trade-offs people are willing to make. Some relational values may also enter economic valuation indirectly when people express concerns about identity, responsibility, community or future generations through their preferences. However, relational values may not be fully represented by individual willingness-to-pay measures because their significance can depend on relationships, social practices and shared understandings rather than on individual utility alone (Díaz et al., 2018; IPBES, 2022; Kenter et al., 2015).

Intrinsic values present an even more fundamental boundary. Economic valuation can capture existence value, but existence value should not be equated with intrinsic value. Existence value describes the value that people derive from knowing that an environmental entity continues to exist. Intrinsic value, by contrast, refers to the proposition that nature, species or ecosystems may possess value independently of their usefulness to or valuation by human beings. Such value cannot simply be added as another category within a conventional preference-based framework because its normative basis differs fundamentally from anthropocentric valuation (Díaz et al., 2015).

The IPBES framework therefore provides an important qualification to the conventional TEV perspective. TEV remains useful for organizing a broad range of economic values, including direct and indirect use values and non-use values. However, the broader IPBES framework shows that the diversity of values of nature cannot be reduced to the categories of economic valuation. Different valuation approaches may illuminate different dimensions of nature's importance, and some values may be better expressed through deliberation, participatory processes, qualitative assessment, biophysical indicators or other forms of value articulation (IPBES, 2022).

5.5 Policy recommendations from an economic perspective

Economic analyses of water stress, ecosystem degradation, and declining ecosystem services indicate that addressing the depletion of natural capital requires a fundamental transformation of economic systems. The challenge is not only to increase investment in ecosystem protection and restoration, but also to reduce and ultimately phase out the economic activities, incentives, and financial flows that drive or contribute to natural capital degradation.

From an economic perspective, this transformation requires action at two closely interconnected levels: First, economic and public policies need to reshape incentives, prices, regulations, and investment frameworks so that economic decisions by businesses and private households better reflect the value and scarcity of natural capital. Second, the financial sector needs to integrate nature-related dependencies and risks into capital allocation, lending, investment, and risk-management decisions. These two dimensions are mutually reinforcing: public policy shapes the risk-return conditions under which financial decisions are made, while financial institutions influence the allocation of capital across economic activities and therefore affect the pace and direction of the transition.

5.5.1 Transforming economic incentives and public policy

A useful conceptual framework for this transformation is provided by the Nature Transition X-Curve (Loorbach et al., 2017; Wittmer et al., 2021; UNEP, 2026; ELD, 2026). The framework emphasises that a successful transition towards nature-positive economies requires two complementary processes: (1) phasing out nature-negative activities and (2) phasing in nature-positive solutions (Fig. 5.7). To put this relationship into perspective, it is worth recalling the relative scale of nature-negative and nature-positive financial flows (Chapter 5.2), which are in the magnitude of more than 30 to one.

On the one hand, the transition requires the phasing-out of activities and incentives that generate persistent pressures on ecosystems, including unsustainable resource extraction, inefficient water use, ecosystem conversion, and environmentally harmful subsidies. These activities often persist because markets fail to reflect the full social and ecological values and costs associated with the depletion of natural capital (see Chapter 5.4). Without addressing these underlying drivers, additional investments

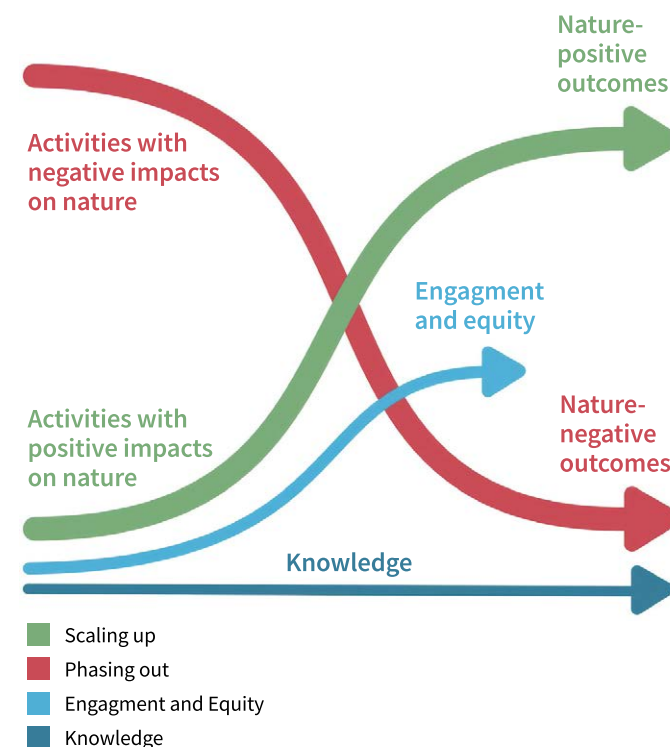


Fig. 5.7 The Nature Transition X-Curve, demonstrating a prototypical pathway towards a future where people and the environment live in harmony. The core principle is a simultaneous phasing-in of nature-positive outcomes (green line) and phasing-out of nature-negative ones (red line). (Source: UNEP, 2026)

in conservation and restoration risk being insufficient to reverse current trends. On the other hand, the transition requires the rapid phasing-in and scaling-up of investments, technologies, and business models that restore ecosystem functions and services and strengthen economic resilience. Nature-based solutions (NbS), sustainable agriculture, circular production systems, water-efficiency measures, and ecosystem restoration can generate multiple economic benefits by reducing risks, enhancing productivity, and supporting long-term value creation. This dual policy approach differs from a conventional environmental policy perspective that focuses primarily on additional conservation spending. Instead, it recognises that the transition requires a broader transformation of economic incentives, financial systems, and decision-making frameworks. Building on the findings of this report, several policy recommendations emerge.

The economic considerations presented in this chapter demonstrate that the value of freshwater ecosystems and biodiversity extends far beyond what is captured by conventional economic indicators and market transactions. A fundamental prerequisite for addressing nature-related economic risks is to make natural capital visible in economic and financial decision-making. Traditional economic indicators, including GDP, do not adequately capture the

depletion of natural assets or the degradation of ecosystem services. As highlighted by TEEB (2010) and the Dasgupta Review (2021), economic systems have historically treated nature as an unlimited asset rather than as a form of capital underpinning economic activity (Dasgupta, 2021; TEEB, 2010).

Governments should therefore move from recognizing natural capital as an environmental concern to systematically incorporating it into economic planning and public decision-making. This includes accelerating the development and implementation of natural capital accounts within national statistical systems and using these accounts alongside conventional economic indicators (Bellinghen et al., 2024; UNEP, 2024c). Consistent accounting frameworks should make these effects measurable and comparable across sectors and over time (Nature Restoration Regulation, 2024; United Nations et al., 2021). This would allow governments to identify where economic activity generates value while depleting natural assets, as well as where investments in ecosystem restoration or protection generate long-term economic benefits (UNEP, 2024c). Such information could provide an evidence base for budget allocation, infrastructure priorities, agricultural and water policies, and, where appropriately developed, fiscal risk assessments. Consistent accounting frameworks should make these effects

measurable and comparable across sectors and over time. This would allow governments to identify where economic activity generates value while depleting natural assets, as well as where investments in ecosystem restoration or protection generate long-term economic benefits. Such information could then inform budget allocation, infrastructure priorities, agricultural and water policies, and fiscal risk assessments. The objective should not be to replace GDP with a single alternative indicator, but to complement conventional economic measures with information on changes in the natural asset base on which future economic activity depends.

For companies and financial institutions, standardised approaches for identifying, assessing, managing, and disclosing nature-related dependencies, impacts, risks, and opportunities should be further developed and implemented. Frameworks such as the Taskforce on Nature-related Financial Disclosures (TNFD) provide an important foundation by establishing a structured approach for organisations to identify, assess, manage, and disclose their nature-related dependencies, impacts, risks, and opportunities (TNFD, 2023a).

One of the central economic drivers of water stress and ecosystem degradation is the existence of incentives that encourage environmentally harmful activities. Underpriced

water resources, subsidies for water-intensive agricultural production, and public support for environmentally damaging activities create economic signals that are inconsistent with long-term resource availability (OECD, 2021b, 2022; UNEP, 2023c). A key policy priority should therefore be the reform of environmentally harmful subsidies and the redesign of incentives in sectors with high impacts on water and biodiversity. This includes, in particular, agricultural subsidies that encourage excessive water consumption, inefficient irrigation practices, or land-use changes leading to ecosystem degradation (OECD, 2021a). Water pricing mechanisms should better reflect scarcity, environmental costs, and long-term resource availability while ensuring affordability and access for vulnerable households. Properly designed pricing systems can encourage efficiency improvements and reduce over-extraction without undermining social objectives (Leflaive & Hjort, 2020; OECD, 2021b).

Water and biodiversity risks are generally widespread across the economy, but highly sector-specific in detail. Agriculture, food production, energy, manufacturing, mining, textiles, and digital infrastructure face different dependencies and impacts. Effective policy responses therefore require more granular approaches rather than relying exclusively on broad environmental targets. [Chapter 5.3](#) illustrates how

nature-related dependencies and impacts can be assessed at the sector level. In agriculture, priorities include improving irrigation efficiency, promoting water-saving production systems, reducing food waste, and encouraging sustainable dietary patterns. In industrial sectors, policies should support water efficiency, circular production models, and technological innovation. For energy and digital infrastructure, water requirements should become an explicit consideration in investment planning.

Freshwater systems are often common-pool resources that cross administrative and national boundaries. Their sustainable management requires cooperation among governments, businesses, communities, and financial actors. Policies should strengthen integrated water-resource management, improve monitoring systems, and establish governance mechanisms that prevent over-extraction and ensure fair allocation among competing users (OECD, 2015; UN Water, 2024). International cooperation is particularly important because virtual water trade transfers water-related pressures across borders through global supply chains (Mekonnen et al., 2024; Tamea et al., 2021). Development finance and international cooperation should support countries facing high vulnerability due to water stress, ecosystem degradation, and limited financial capacity (UN Water, 2024; World Bank, 2024), particularly where declining natural

capital threatens economic development, fiscal stability, and sovereign creditworthiness.

5.5.2 Mobilising finance for the nature transition

Economic policy reform alone cannot deliver the required transition without sufficient investment. A fundamental challenge is the substantial mismatch between the scale of required investment and current financial flows. While public finance remains essential for ecosystem conservation, restoration, and water security, the magnitude of the challenge exceeds the capacity of governments alone. Mobilising private capital is therefore a central element of an economically viable transition towards a nature-positive economy.

The economic rationale for public intervention is particularly strong because many ecosystem services generate substantial societal benefits but limited direct financial returns for investors. Water regulation, flood protection, soil restoration, biodiversity conservation, and ecosystem resilience often represent public goods and common-pool resources whose economic value is not fully captured in market transactions ([Chapter 5.1](#)). As a result, projects that generate significant social and environmental benefits may not be financially attractive under

conventional investment criteria. This creates an investment gap despite the potentially high economic returns of ecosystem restoration and nature-based solutions.

According to estimates by the ELD group (ELD, 2026), global commitments to combat desertification, land degradation, and drought, to restore ecosystems, and to achieve land degradation neutrality require cumulative investments of approximately US\$ 2.6 trillion over the period 2016–2030 (UNCCD, 2024) (Chapter 5.1). Despite the critical role of land and nature as foundations of economies and societies, current finance remains far below required levels. Investment in land- and drought-related objectives increased from approximately US\$ 37 billion in 2016 to nearly US\$ 66 billion in 2022; however, an estimated increase to approximately US\$ 355 billion annually during 2025–2030 is required, implying an annual financing gap of around US\$ 278 billion (UNCCD, 2024). At the same time, financial flows that negatively affect nature remain vastly larger, with around US\$ 7.3 trillion annually directed towards activities that contribute to ecosystem degradation. Closing the investment gap therefore requires not only mobilising additional capital (‘financing green’), but also redirecting existing financial flows away from harmful activities (‘greening finance’).

Scaling investment in restoration also represents a major economic opportunity. Restoring degraded ecosystems could unlock an estimated US\$ 24.6 trillion in additional annual ecosystem-service value, highlighting that land restoration can be viewed as an investment opportunity rather than merely a cost.

The limited involvement of private investors reflects several structural barriers. Nature-positive investments may generate substantial societal benefits while providing uncertain, delayed, or insufficient private financial returns. Policy frameworks should therefore aim to improve the risk–return profile of nature-positive investments and create conditions that enable private capital mobilisation. This requires a combination of regulatory reforms, improved risk assessment, innovative financial instruments, and public-sector de-risking mechanisms. Blended finance models are particularly relevant because they combine public, philanthropic, and private capital to overcome investment barriers (ELD, 2026). Public resources can be used strategically to absorb initial risks, provide guarantees, offer concessional finance, or support project preparation, thereby making nature-positive investments more attractive to institutional investors such as pension funds, insurers, and asset managers. Such approaches are particularly relevant in emerging and developing economies, where high political,

regulatory, and market risks can prevent private investment despite substantial ecological and economic benefits.

Private capital can be mobilised through a range of financial instruments, including guarantee mechanisms and risk-sharing instruments that reduce investment risks for private financiers; concessional finance and first-loss capital provided by development banks or public institutions to improve project viability; nature-based infrastructure funds that pool investments in ecosystem restoration, watershed protection, and sustainable land management; and payments for ecosystem services (PES) that create revenue streams for landowners and communities protecting water-related ecosystem functions. Green, blue, and sustainability-linked bonds can additionally mobilise capital for projects contributing to water security, ecosystem restoration, and climate resilience. The suitability of individual instruments depends on the characteristics of the underlying projects, their revenue-generating capacity, risk profile, and institutional context.

The OECD report *Financing Water Security: Models and Approaches for Investments that Last* (OECD, 2026b) examines how the substantial investment gap in the water sector can be addressed through sustainable, long-term financing models. In light of growing challenges such as climate change, water scarcity, ageing infrastructure, and increasing water-quality requirements, public budgets alone are insufficient to meet future investment needs. The report therefore emphasises the importance of strategically combining public and private sources of finance. It highlights a range of financing instruments, including bonds, public-private partnerships, results-based financing mechanisms, and Islamic finance. Their effectiveness depends on appropriate risk allocation, predictable regulatory frameworks, transparent governance, and long-term project planning. Governments and financial institutions should therefore create enabling conditions that attract private capital for water infrastructure while safeguarding public interests and ensuring long-term water security. Public finance should not simply substitute for private capital but should be used strategically to address market failures, reduce risks, and create investable opportunities.

5.5.3 The role of the financial sector in redirecting capital

While public policy establishes the broader economic framework, financial institutions play a central role in determining how capital is allocated within that framework. Banks, insurers, asset managers, pension funds, and other investors increasingly need to consider ecosystem degradation and water insecurity as material financial risks affecting asset values, creditworthiness, cash flows, and long-term investment performance.

Financial institutions should integrate nature-related dependencies, impacts, risks, and opportunities into lending decisions, investment strategies, and risk-management frameworks. Banks and investors should assess their exposure to sectors and regions with high dependencies on ecosystem services, particularly freshwater resources. Nature-related risks include physical risks arising from ecosystem degradation and resource scarcity as well as transition risks associated with regulatory changes, technological transformation, and changing market expectations. Scenario analysis and stress-testing approaches should increasingly consider compound risks arising from interactions between climate change, biodiversity loss, and water insecurity. As demonstrated by recent studies on financial exposure

in the euro area (see [Chapter 5.3](#)), nature-related dependencies can affect substantial shares of corporate lending portfolios. This illustrates that nature-related risks are not peripheral environmental concerns but can become material financial risks.

Standardised approaches for assessing and disclosing nature-related dependencies, impacts, and risks should be further developed and implemented. Frameworks such as the Taskforce on Nature-related Financial Disclosures (TNFD) provide an important foundation by encouraging financial institutions and companies to identify, assess, manage, and disclose their exposure to nature-related risks. Improved disclosure can strengthen transparency and allow investors and financial institutions to differentiate more effectively between activities that contribute to nature degradation and those that support ecosystem resilience. However, disclosure alone is insufficient. Information must ultimately be incorporated into capital-allocation decisions, risk assessment, pricing, and engagement strategies.

The financial sector also has an important role in redirecting capital flows. Financial institutions should increasingly distinguish between activities that generate persistent pressures on natural capital and activities that contribute to ecosystem restoration, resource efficiency, and

resilience. This requires moving beyond the mobilisation of additional “green” finance. The transition also requires the gradual withdrawal or repricing of finance for activities that contribute significantly to ecosystem degradation. In this sense, the financial sector is directly connected to both the phasing-in and phasing-out. Economic policy and financial-sector action should not be viewed as separate policy domains. They are mutually dependent components of the processes described by the Nature Transition X-Curve.

5.5.4 Linking economic policy and financial-sector action

Economic policy and financial-sector action should not be viewed as separate policy domains. They are mutually dependent components of the same transition process. Public policy determines many of the conditions under which economic and financial decisions are made. Environmental regulation, water pricing, subsidy reform, natural capital accounting, taxation, public investment, and risk-sharing mechanisms influence the relative attractiveness and risk profile of different economic activities. By correcting market failures and internalising environmental externalities, governments can improve the economic incentives for nature-positive activities and reduce incentives

for nature-negative ones. At the same time, financial institutions translate these economic conditions into capital-allocation decisions. The cost and availability of finance influence corporate investment, infrastructure development, technological innovation, and business-model transformation. If nature-related risks are appropriately reflected in lending and investment decisions, capital costs can increasingly reflect the long-term risks associated with ecosystem degradation and resource scarcity.

This creates an important feedback mechanism. Effective public policy can improve the risk-return profile of nature-positive investments and increase the attractiveness of private capital. In turn, financial-sector risk assessment and capital allocation can reinforce policy objectives by directing investment towards sustainable activities and increasing the financial pressure on activities that are incompatible with long-term environmental sustainability. The relationship can therefore be understood as a mutually reinforcing cycle that can lead to economic transformation and a state of reduced pressure on natural capital. Breaking this cycle at either level can weaken the overall transition. Public policies without corresponding changes in capital allocation may fail to redirect sufficient resources, while financial-sector initiatives without supportive policy frameworks may struggle

to generate adequate risk-adjusted returns for nature-positive investments.

An effective transition towards a nature-positive economy therefore requires coordinated action across both spheres. And public policy needs to establish credible long-term policy frameworks and stable mechanisms to correct market failures and harmful incentives—and use public finance strategically. Financial institutions will only then integrate nature-related risks effectively into their decision-making and redirect capital towards resilient and sustainable economic activities. The interaction between these two levels is essential for closing the financing gap, reducing systemic risks, and creating the conditions for long-term economic resilience.

5.6 Key messages

► **Natural capital is economic capital.**

The degradation of freshwater resources, ecosystems, and biodiversity represents a depletion of natural capital and undermines the ecological foundations of economic activity, prosperity, and long-term economic resilience.

► **The water and biodiversity crises are mutually reinforcing and driven by economic activity.**

Water stress, ecosystem degradation, and biodiversity loss are closely interconnected and can reinforce one another. Agriculture, industry, energy production, land-use change, pollution, and resource extraction are major drivers of these pressures, while economic activity itself depends on functioning ecosystems and reliable water resources.

► **Nature-related dependencies create economic and financial risks.**

Economic sectors depend directly and indirectly on ecosystem services such as water supply and purification, pollination, soil quality, flood regulation, and climate regulation. These dependencies are transmitted

through supply chains and can translate into physical, transition, and systemic risks for companies, governments, and financial institutions.

► **Environmental degradation imposes substantial economic costs that can propagate across economic sectors.**

Droughts, floods, water scarcity, land degradation, and declining ecosystem services can reduce productivity, disrupt supply chains, increase commodity prices, damage assets, and place pressure on public budgets and sovereign creditworthiness. Local environmental shocks can therefore propagate across sectors and borders, generating wider national and international economic and financial consequences.

► **Economic incentives matter.**

Market failures, externalities, the underpricing of water and ecosystem services, harmful subsidies, common-pool resource problems, and short-term decision-making can contribute to the overuse and degradation of natural resources. Conventional economic indicators such as GDP do not

adequately capture the depletion of natural capital.

► **Economic valuation of nature can improve decision-making—but has limits.**

Valuation methods can capture a broad range of use and non-use values and improve cost-benefit analysis and investment decisions. However, monetary valuation remains anthropocentric and context-dependent and cannot fully represent the relational and intrinsic values of nature.

► **The transition requires both phasing in nature-positive activities and phasing out nature-negative activities.**

Nature-based solutions, ecosystem restoration, sustainable water management, and resilient infrastructure can simultaneously reduce risks, strengthen ecosystem services, and generate economic benefits. However, increasing investment in nature-positive activities alone will not be sufficient: harmful subsidies, unsustainable resource use, and financial flows that contribute to ecosystem degradation must also be reduced and finally redirected.

6 From competition to synergy

Klement Tockner

Decades of mismanagement and undervaluation of water have degraded freshwater ecosystems and depleted and contaminated freshwater resources worldwide. These physical, chemical and biological alterations of freshwaters as well as the increases in human water withdrawals can be considered a ‘pandemic array’ of alterations – or a pandemic transformation of the water cycle – that characterise the Anthropocene (Alcamo et al., 2008). Overcoming the competition between human water use and freshwater biodiversity requires a fundamental shift from fragmented, sectoral water management towards a systemic approach that recognises freshwater resources and freshwater ecosystems as mutually dependent foundations of nature, economies and human well-being.

Freshwater connects climate, biodiversity, food, energy, health and economic development and therefore provides an integrating “currency”

across otherwise separately governed domains. Indeed, water is directly or indirectly relevant to all 17 Sustainable Development Goals (GCEW, 2024).

Conventional water management and economy have focused predominantly on blue water in rivers, lakes, wetlands and groundwaters, while paying less attention to green water (i. e., soil moisture and water in vegetation) and its critical role in sustaining terrestrial ecosystems, agriculture, atmospheric moisture recycling and climate regulation (GCEW, 2024; Gleeson et al., 2020; Wang-Erlandsson et al., 2022). Human alteration of both blue and green water has now reached a scale at which freshwater change must be considered a fundamental threat to the Earth system rather than merely a local resource-management issue (GCEW, 2024; Gleeson et al., 2020; Richardson et al., 2023; Rockström J, J Gupta, D Qin, SJ Lade, JF Abrams,

LS Andersen, et al. (2023). Safe and just Earth system boundaries. *Nature* 619 (7968), 102-111).

Competition for freshwater increases because the same finite and highly variable resource must sustain freshwater ecosystems while meeting demands for irrigation, cities, industry and energy. Increasing human appropriation can therefore generate linked conflicts: among human users and between human use and the environmental flows required to maintain freshwater biodiversity and ecosystem functions (Pastor et al., 2014). The Colorado River provides a striking example: it supports more than 40 million people and over two million hectares of cropland, yet extensive water consumption has contributed to a river that rarely reaches its delta (Richter et al., 2024). The 2014 binational release of about 132 million m³ as environmental flow to the largely dry delta illustrates the fundamental trade-off as well as the potential

for deliberate reallocation of water from consumptive uses to ecosystem needs (Nelson et al., 2017).

Moving beyond competition among different water uses towards synergy requires a new approach to water governance that values not only water withdrawn for human use, but also water retained in rivers, lakes, wetlands, soils, and groundwater, together with the multiple benefits generated by functioning ecosystems. Wetland degradation and deforestation disrupt water and carbon cycles simultaneously, illustrating why climate, biodiversity and water policies cannot be pursued independently. Indeed, a paradigm shift in landscape hydrology is already under way: whereas water was historically drained from landscapes as quickly as possible, the emphasis is increasingly on retaining water in the landscape, for example by rewetting drained peatlands, restoring more diverse and resilient forests, and reconnecting rivers with their floodplains. Such measures can simultaneously support water retention, biodiversity and carbon storage and mitigate floods and droughts.

Freshwater ecosystems and the hydrological processes sustaining them should be recognised simultaneously as Natural Capital, life-supporting systems and critical natural infrastructure, with their protection and restoration treated

as investments in societal and ecological resilience rather than being reduced to constraints on economic development (Chung et al., 2021; Dasgupta, 2021). The objective should be to create synergies and reduce trade-offs with lopsided negative effects across biodiversity, water, food, health and climate, as emphasised by the IPBES Nexus Assessment, while recognising the hydrological cycle as a global common good, as proposed by the Global Commission on the Economics of Water (GCEW, 2024). In this sense, water is not simply one environmental sector among many: it is the fundamental connecting medium through which climate stability, biodiversity, ecosystem services, and human well-being are maintained (GCEW, 2024; IPBES, 2024).

We identify nine domains that we consider key for achieving synergies in the integrated management of freshwater resources, biodiversity, and the services they provide, for the benefit of nature and people alike. These nine domains are not intended to be exhaustive; rather, they highlight key areas in which fundamental changes in policy, governance, investment and management could help shift the relationship between water use and freshwater biodiversity from competition towards synergy. Together, they provide a framework for developing a more integrated approach to safeguarding freshwater ecosystems and water security in a rapidly changing world.

6.1 Recognise the hydrological cycle and freshwater ecosystems as global common goods and natural capital

In 2017, Aotearoa aka New Zealand fundamentally changed the legal status of the Whanganui River. The Te Awa Tupua Act recognised the river not as water to be owned, allocated or used, but as ‘*an indivisible and living whole*’ extending from the mountains to the sea and incorporating its physical and metaphysical elements. The act granted the river legal personhood and established guardians to represent its interests. Rooted in Māori law and the *Ko au te Awa, ko te Awa ko au* principle–‘I am the River and the River is me’–the approach places the integrity of the freshwater ecosystem and reciprocal responsibilities between people and nature at the centre of governance (New Zealand Parliament, 2017).

Indeed, freshwater cannot adequately be treated as an ordinary consumption commodity. It is essential and largely non-substitutable for life, ecosystems and economies; it moves across political and property boundaries and it generates benefits and costs for people that markets systematically fail to capture. The broader lesson from Te Awa Tupua is therefore not that its particular legal model should necessarily be scaled up globally, but that water is more than a resource: it is part of an interconnected

living system whose stewardship entails shared responsibilities.

Moving from a single river basin to the global scale, freshwater is an essential part of the Earth system. Depletion of terrestrial water storages contributes to global sea-level rise (Horwath et al., 2022), while large-scale irrigation can modify regional climate by cooling the land surface and altering atmospheric circulation and precipitation, as demonstrated for the middle and lower Yangtze River basin (Wang et al., 2021). Virtual water connects consumers with distant water resources and freshwater ecosystems. These multiple interdependencies have contributed to the proposition that the hydrological cycle should be treated as a global common good (GCEW, 2024; Porcher, 2024, 2026).

This perspective, too, shifts water governance from allocating a resource among competing users to safeguarding the integrity and regenerative capacity of the hydrological cycle and its ecosystems as a shared responsibility. This requires governing economic activities within hydrological and ecological limits, while ensuring equitable access and sustaining benefits for

both societies and ecosystems. To guide a fair and just governance, approaches based on sufficiency criteria may offer an important complement to utilitarian or egalitarian perspectives by asking which minimum ecological and societal conditions must be secured before additional benefits are distributed (Wallimann-Helmer et al., 2023).

Porcher (2026) further argues that water can be understood as a global common good because it creates (i) biophysical interdependence through the hydrological cycle and atmospheric moisture flows, (ii) economic interdependencies generated through international virtual water trade, and (iii) functional interdependencies as currency across all 17 SDGs (Porcher, 2026). These interconnections create externalities, but also missing property rights and regulatory leakage, that cannot be addressed through local or regional governance alone (Porcher, 2024).

Moreover, global goods may be defined not only by their material properties or individual service provision but also by the benefits they generate globally. For water, this means safeguarding system integrity across interconnected,

place-based components, from atmospheric moisture circulation to environmental flows in shared rivers. At the catchment scale, however, water is typically a rivalrous common-pool resource.

This perspective also changes how prosperity is measured. In an *inclusive wealth* framework, freshwater resources and biodiversity are critical natural capital: they underpin health, food and energy production, economic activity and ecosystem functioning. Their degradation can therefore increase GDP today while depleting the natural assets on which future prosperity depends (UNEP, 2023a).

Ostrom's (1990) institutional analysis of common-pool resources demonstrated both the potential and the conditions for collective action in governing shared and scarce natural resources. Her work challenged the assumption that common resources can be sustainably managed only through either central state control or privatisation and identified institutional principles associated with successful collective governance. The question is to what extent these principles can facilitate governance at larger and up to international scales. This requires cooperation among governments, the private sector and civil society, but also much more coherent integration of trade, climate, water and biodiversity policies within international and

supranational institutions such as the European Union.

A major institutional gap remains in synthesising scientific knowledge on freshwater change and translating it into coordinated international action. We therefore propose establishing an Intergovernmental Panel on Water (IPW) – analogous in its science-assessment function to the IPCC for climate and IPBES for biodiversity – to provide regular, authoritative assessments of the state of the global water cycle, freshwater resources and ecosystems, emerging risks, and options for action. Proposals for an IPCC-like scientific assessment mechanism for water have been discussed previously, including within WMO and UNESCO-IHP, but no equivalent global science-policy assessment body for water currently exists.

An IPW should complement, not duplicate, UN-Water. While UN-Water coordinates water-related activities across the UN system, an IPW would deliver the independent scientific assessment function currently missing and thereby provide UN-Water and governments a common evidence base for coordinated action.

6.2 Prioritise nature-based and hybrid solutions alongside damage-minimising technical innovations

There is growing reliance on large-scale engineering solutions to address increasing challenges in the water sector, including the construction of dams, inter-basin water transfers and desalination plants (Shumilova et al., 2018; Zarfl et al., 2015). Desalination, for example, is expanding rapidly in water-stressed regions: global capacity grew by around 7 % annually during the 2010s and could increase approximately threefold by 2050, creating challenges related to energy demand, greenhouse-gas emissions, seawater abstraction and hypersaline brine discharge (Andrijevic et al., 2026; Jones et al., 2019). Such megaprojects can entail high financial, social and ecological risks because of their scale, long planning horizons and far-reaching impacts. They can also create path dependencies and technological lock-ins that constrain alternative solutions for decades.

Greater emphasis is therefore needed on green infrastructure, nature-based solutions (NbS), hybrid natural-engineered systems, and local and decentralised approaches. Particularly effective hybrid solutions combine restored floodplains and retention areas with dikes for river-flood protection, wetlands with engineered

wastewater treatment, and coastal wetlands with dikes and breakwaters (e. g., Huynh et al., 2024; Wu et al., 2023). Such combinations can provide high levels of protection while reducing costs and environmental impacts and delivering major benefits for biodiversity and climate resilience. In coastal protection, for example, a global meta-analysis found hybrid measures to provide the highest hazard reduction and greater cost-effectiveness than conventional hard infrastructure alone (Perricone et al., 2023).

A prominent example is the Dutch *Room for the River* programme, which marked a shift from repeatedly raising dykes towards giving rivers more space. Across 39 initially selected locations, measures included dyke relocation, floodplain lowering and bypass channels, with the dual objectives of increasing flood safety and improving spatial quality (Klijn et al., 2013). The programme demonstrates how engineered flood protection can be combined with more natural hydrological processes and multifunctional river landscapes.

Germany is pursuing related approaches at landscape scale. The Federal Blue Belt

Programme (*Bundesprogramm Blaues Band*) combines river and floodplain restoration with ecological connectivity, flood-risk reduction and the development of multifunctional river landscapes. The Federal Action Plan on Nature-based Solutions for Climate and Biodiversity (*Aktionsprogramm Natürlicher Klimaschutz*, ANK) similarly promotes the rewetting of peatlands and restoration of other ecosystems, with potential benefits for landscape hydrology, biodiversity, climate mitigation and adaptation. Such transformations, however, require more than individual restoration projects. They depend on viable business and financing models, long-term governance arrangements and incentives for landowners and other stakeholders that make landscape-scale changes economically and socially feasible. New value chains for paludiculture on rewetted peatlands illustrate this shift from financing restoration itself towards creating viable economic activities compatible with restored ecosystem functions.

6.3 Make water diplomacy an instrument for cooperation, stability and shared prosperity

Water diplomacy is becoming increasingly important as freshwater resources and biodiversity face mounting pressures from climate change, pollution, major infrastructure development and growing human demands across sectors and national boundaries. Transboundary rivers, lakes and groundwaters account for more than 60 % of global freshwater flows, and 153 countries share transboundary waters. However, only 43 countries have operational cooperation arrangements covering at least 90 % of their shared waters (UNESCO & UNECE, 2024).

Effective water diplomacy requires moving beyond negotiations over water allocation towards the joint management of entire river basins and groundwaters, linking Integrated Water Resources Management (IWRM), ecosystem protection, climate adaptation and conflict prevention. The IPBES Nexus Assessment provides a broader policy framework for this shift, emphasising the interdependencies among biodiversity, water, food, health and climate and the need to address synergies and trade-offs among policy choices rather than managing these domains separately (IPBES, 2024). Water diplomacy should therefore increasingly become ‘nexus diplomacy’, addressing

freshwater as part of an interconnected social-ecological system.

The urgency of progress in this domain is reinforced by a sharp increase in water-related violence in recent decades, particularly in the Middle East, southern Asia, and the Sahel. Water can be a trigger, weapon, or casualty of conflict: scarcity and unequal access can exacerbate tensions; dams and water infrastructure can be strategically manipulated, and water-supply systems are increasingly damaged or deliberately targeted during armed conflicts (Gleick & Shimabuku, 2023). The Russia–Ukraine war provides a striking recent example. The destruction of the Kakhovka Dam in 2023 caused massive and long-lasting hydrological and ecological changes along the Dnipro River and the Black Sea coast (Shumilova et al., 2025).

At the same time, shared water can become a catalyst for cooperation and peace. Because upstream and downstream countries are hydrologically interdependent, cooperation can generate benefits that unilateral action cannot provide, including more efficient water and energy use, coordinated flood and drought management, ecosystem restoration, shared

monitoring and early-warning systems, and increased regional stability (Krause et al., 2024; Sadoff & Grey, 2002; Wolf, 2007).

Water diplomacy should shift the focus from dividing water volumes towards sharing benefits, risks, information and responsibilities across the entire river basin. Transboundary River Basin Organisations (RBOs) can provide the institutional backbone for such cooperation. Successful organisations can create continuity beyond short-term political cycles, provide platforms for negotiation and dispute resolution, coordinate infrastructure and environmental management, and generate trusted scientific information for joint decision-making (Kittikhoun & Schmeier, 2021; Schmeier, 2013).

Several RBOs already demonstrate what effective water diplomacy can achieve. Institutionalised international cooperation and coordination in the Rhine River Basin transformed a heavily polluted international river into a model of coordinated water-quality improvement and ecological restoration, including the return of migratory fishes such as Atlantic salmon (*Salmo salar*) and allis shad (*Alosa alosa*). In the Senegal River Basin, cooperation

through the Organisation pour la mise en valeur du fleuve Sénégal (OMVS; Mali, Mauritania, Senegal) has gone further, with jointly managed infrastructure and shared benefits from hydropower, irrigation and navigation, illustrating how water diplomacy can shift from allocating water towards sharing benefits.

The effectiveness of RBO varies considerably, however. Power asymmetries between upstream and downstream countries, weak mandates, insufficient financing, limited data sharing and geopolitical interests can constrain cooperation. Moreover, maintaining the status quo through non-decision, non-participation, or delayed implementation can itself become an expression of power in transboundary basins (Vij et al., 2024). Likewise, whether countries participate in international water conventions is shaped not only by environmental needs but by sovereignty, geopolitical interests and power relations (Nagheeb, 2026). The Mekong illustrates these constraints: although the Mekong River Commission (MRC) provide an important platform for exchange, the effectiveness of cooperation and coordination is limited by national sovereignty, hydropower interests and power asymmetries, with China remaining a dialogue partner rather than a full member (Pham, 2025). Lastly, the example of the Zambezi Watercourse Commission illustrates the importance of water diplomacy and deliberation in informal learning

spaces under conditions of distrust and power struggles (Lumosi et al., 2019, 2020).

Strengthening water diplomacy therefore requires stronger and more adaptive RBOs, supported by international water law, transparent data sharing, joint monitoring and modelling, early-warning systems, stakeholder participation and sustainable financing (Pahl-Wostl, 2020; UNESCO & UNECE, 2024). Importantly, ecological objectives need to become an integral part of water diplomacy rather than an environmental add-on: maintaining environmental flows, sediment connectivity, wetlands, migratory corridors, good water quality and groundwater-surface-water interactions can simultaneously strengthen biodiversity, water security and climate resilience. International water cooperation also is a particularly important mechanism for implementing the Global Biodiversity Framework (GBF), particularly its commitments to maintain or restore ecological integrity and connectivity. Ultimately, water diplomacy offers an opportunity to transform shared freshwater from a potential source of conflicts into a medium facilitating cooperation, ecological resilience and peace (Krause et al., 2024).

6.4 Safeguard biocultural diversity while preparing for ecological novelty

Beyond providing water, food, fertile soils, and transport corridors, freshwaters contribute to recreation, aesthetic and spiritual values, cultural heritage, collective memory, identity, and sense of place, freshwaters are not only ecological systems; they are also cultural scenes. They have shaped where people live, how societies use resources, and how cultures develop. Human societies have therefore developed in continuous interaction with freshwater ecosystems and, in particular, with their hydrological rhythms (Wantzen, 2022; Wantzen et al., 2016). Indigenous and traditional knowledge systems are particularly important manifestations of biocultural diversity (i.e., linkage between biological and cultural diversity), but the concept can also encompass the ecological knowledge, practices, identities and stewardship of anglers, conservation volunteers, birdwatchers, river associations and other nature-engaged communities.

This relationship is especially apparent in dynamic river–floodplain systems. Seasonal expansion and contraction, floods and low-flow periods, and less predictable hydrological events create alternating windows of opportunity and risk. These dynamics select for

ecological strategies to cope with variability, while also shaping human practices, institutions and cultural traditions. The River Culture concept captures this coupling: adaptation to recurring hydrological change can be embedded both in biological diversity and in cultural knowledge and practices (Wantzen et al., 2016). Hence, in many freshwater systems, hydrological variability is a fundamental process sustaining both ecological and cultural diversity.

Migratory fishes provide striking examples of these coupled relationships. Pacific salmon (*Oncorhynchus* spp.) have shaped the cultures of Indigenous Peoples around the North Pacific for millennia. The decline of these fish populations represents not only a loss of biodiversity and ecosystem functioning, but also a loss of cultural practices, food sovereignty, ecological knowledge and sense of place (Atlas et al., 2021; Garibaldi & Turner, 2004; Schindler et al., 2010). Similarly, relationships between Stóo Peoples and White Sturgeon (*Acipenser transmontanus*) in the Fraser River are embedded in oral histories, spiritual beliefs, customary laws and ecological knowledge (Oloriz & Parlee, 2020). Such examples demonstrate that freshwater biodiversity and biocultural diversity are deeply

intertwined: degrading ecological relationships can simultaneously erode cultural relationships, knowledge, and identity.

This interdependence has major implications in an era of rapid global change. Climate change, biological invasions, altered flow regimes, pollution, and other pressures push many freshwater ecosystems beyond historical reference conditions. Restoration to former states is consequently becoming more difficult and, in some cases, may no longer be feasible. New combinations of environmental conditions, species, ecological processes, and human–nature interactions without historical analogues are emerging. Consequently, ecological novelty will therefore increasingly become part of the new normal (Heger et al., 2019, 2026; Kerr et al., 2026).

The Rhine River exemplifies the emergence of novel aquatic communities following environmental recovery and biological invasions. Although non-native macroinvertebrates represent only about 30–40 % of benthic taxa, they account for more than 80–90 % of macroinvertebrate biomass in some sections of the river (Leuven et al., 2009). More generally, the rapid spread of non-native and translocated

species across river basins has led to the homogenisation of historically distinct freshwater communities. (Dudgeon & Liew, 2026; Sommerwerk et al., 2017).

This creates a fundamental challenge for conservation and restoration. Protecting present biodiversity and restoring degraded ecosystems remains essential wherever this is feasible. However, conservation can no longer be concerned exclusively with recovering the past – or not even protecting the present state. It must increasingly combine protection and restoration with adaptation to unavoidable ecological change. The central question is how ecological functions, biodiversity, and human–nature relationships can be sustained or renewed under rapidly changing conditions.

These transformations have profound cultural consequences. The degradation or disappearance of familiar environments can generate *solastalgia*, which means distress associated with environmental change in one's home environment as well as feelings of loss, injustice, and powerlessness (Albrecht et al., 2007). Changes in freshwater ecosystems can alter cultural identity, belief systems, social norms, livelihoods, and sense of place. Ecological novelty is therefore not simply about new species assemblages or altered ecosystem processes. It can create biocultural novelty by transforming the

relationships through which people understand, value and interact with freshwater ecosystems.

At the same time, cultural diversity can itself provide an important source of adaptive capacity. Indigenous and local knowledge, cultural memory, customary practices, and long experience of living with hydrological variability contain multiple ways of responding to environmental change. Maintaining these relationships is therefore not merely about preserving cultural heritage; it can contribute directly to social–ecological resilience. Reconnecting people with freshwater ecosystems can strengthen a sense of place and belonging, sustain knowledge across generations and potentially increase societal engagement in conservation, restoration, and adaptation.

Such reconnection can take many forms. Buried urban streams can be daylighted, rivers reconnected with their floodplains, wetlands rewetted, and opportunities created for people to directly experience near-natural waters. Particularly in cities, where direct relationships with rivers and wetlands have often been lost, accessible human–river encounter spaces can reconnect ecological restoration with everyday experience (Zingraff-Hamed et al., 2021). Cultural practices associated with rivers, from fishing and swimming to seasonal celebrations and collective stewardship, can similarly reconnect ecological

processes with social life. These approaches should draw on indigenous and local knowledge, and involve practitioners, communities, and site managers in the co-production of knowledge and restoration strategies.

As mentioned, reconnection does not necessarily mean recreating either an ecological or a cultural past. Under conditions of ecological novelty, historical reference states remain important guides, but they cannot be the sole endpoint of restoration. Where environmental change is irreversible, societies may also need to develop even *de novo* relationships with transformed ecosystems, including new practices, meanings, institutions and forms of stewardship that maintain ecological integrity while enabling people to live with change.

The challenge is thus twofold: to conserve and restore the ecological and biocultural diversity that can still be safeguarded, while building the capacity to navigate ecological novelty where historical conditions cannot be recovered. Freshwater restoration in the Anthropocene must consequently be understood as more than the repair of ecosystems. It is also about maintaining, restoring and where necessary creating the relationships between people and water that enable both nature and societies to persist through change.

6.5 Integrate water governance across biodiversity, climate, food, energy and health sectors

Water connects a wide range of crucial sectors, including food production, energy generation, climate regulation, biodiversity and human health (IPBES, 2024). Agriculture accounts for around 70 % of global freshwater withdrawals; energy infrastructure, particularly dams, can fragment river systems and alter flow regimes; degraded peatlands release substantial quantities of greenhouse gases; and inadequate drinking water, sanitation and hygiene contributed to an estimated 1.4 million preventable deaths and 74 million disability-adjusted life years (DALYs) in 2019 (WHO, 2023b). The challenge is, therefore, not simply to allocate water more efficiently among sectors but to develop interventions that generate multiple benefits while avoiding the transfer of environmental costs from one sector to another. Peatland rewetting, treated wastewater re-use, nutrient recovery, diversified renewable-energy portfolios (with emphasis on solar and wind), and landscape water retention illustrate how conventional trade-offs can be converted into synergies.

6.5.1 Water–Food Nexus

Agricultural production, including feed and fibre, is the largest direct human use of freshwater. Livestock production accounts for about 29 % of the global agricultural water footprint, mainly through feed production, although most of it is rain-fed green water rather than irrigation-derived blue water (Mekonnen & Hoekstra, 2012). Changing human diets would therefore massively reduce the overall footprint. In France, Germany and the United Kingdom, nutritionally recommended diets would reduce food-related blue and green water footprints by 11–35 % for diets containing meat, 33–55 % for pescetarian diets and 35–55 % for vegetarian diets (Vanham et al., 2018). However, reducing animal products does not necessarily reduce scarce blue-water use because several plant-based foods important in healthy diets, particularly cereals, fruits, nuts, and oils, can make substantial contributions to blue-water footprints. Production location and local water scarcity therefore matter alongside dietary composition (Gerten et al., 2020; Harris et al., 2020).

Water efficiency improvements also require careful assessment. Drip irrigation, precision

farming, and solar-powered pumping can reduce water or energy use per unit of production, but lower costs can encourage the expansion of irrigated areas and thereby increase total abstraction. Innovative incentive systems can counteract this rebound effect; for example, grid-connected solar irrigation can allow farmers to sell surplus electricity rather than use it for additional groundwater pumping.

Circular production systems offer further opportunities. Recirculating Aquaculture Systems (RAS) can reuse more than 90 % of their water, substantially reducing freshwater demand, although their energy requirements need to be considered. Aquaponic systems such as the “Tomato-Fish” concept developed at IGB even recycle nutrients from fish production into plant production (Kloas et al., 2015). Similarly, treated wastewater can substitute for freshwater and return nutrients to productive use rather than allowing them to contribute to eutrophication.

An integrative global study showed that it is possible to feed ten billion people without exceeding the planetary boundaries of freshwater use, nitrogen pollution, forest loss, and biosphere integrity if challenging but feasible changes in

food production and consumption are implemented (Gerten et al., 2020). These changes would lead to an overall reduction in cropland and irrigation water use and encompass, in increasing order of effectiveness: 1) sustainable expansion of croplands, nutrient use and irrigation, 2) improved water, nutrient and land management, 3) halving food loss, and 4) a diet supplying everyone with 2,355 kcal per capita per day, with only 25 % of protein from animal sources (Gerten et al., 2020).

Cotton accounts for about 10 % of the global blue-water footprint of crop production and can therefore place considerable pressure on scarce water resources (Mekonnen & Hoekstra, 2020). While water use per kg of produced cotton has decreased over time due to improved cultivars, nutrient input and water management, increased production led to an increase in the global blue-water footprint of cotton by 17 % between 1972 and 2018 (Demeke et al., 2024). More than two thirds (71 %) of this blue-water use became part of the international virtual water trade (Demeke et al., 2024). Efficiency gains therefore need to be accompanied by demand-side measures, including more durable and less water-intensive patterns of textile consumption.

The way forward is to increase the nutritional and economic value generated per unit of

scarce water through healthier plant-rich diets, water-sensitive crop choices, limits on ground-water use, efficient irrigation, wastewater reuse, circular nutrient systems and lower demand for irrigation-intensive products.

6.5.2 Water–Energy Nexus

The energy transition reduces greenhouse-gas emissions while simultaneously decreasing water withdrawals for cooling in thermal power plants. However, increased hydropower production can increase pressures on freshwater ecosystems. Zarfl et al., (2025) identified at least 3,700 major hydropower dams that were planned or under construction at the time of their assessment, which, if implemented, would have increased global hydropower capacity by approximately 73 %. Dams alter flows, interrupt sediment transport and fish migration pathways, and fragment river networks. Globally, only 37 % of rivers longer than 1,000 km remain free-flowing over their entire length (Grill et al., 2019).

For Africa, existing and proposed renewable-energy plants could, if fully implemented and operated at assumed capacities, provide around 76 % of projected continental electricity demand in 2040 (Peters et al., 2024). Moreover, all 36 African countries with proposed hydropower development have sufficient technically

feasible solar and wind potential to replace the projected hydroelectricity generation, although realising this potential would require storage, expanded grids and regional electricity exchange (Peters, 2024). This is of particular importance, as 102 of the 673 currently proposed large hydropower plants across Africa fall within the top quarter of projects with the greatest ecological and socioeconomic impacts (Peters et al., 2026).

Other climate-mitigation technologies can also increase pressure on water resources. In scenarios with widespread irrigation of biomass plantations for Bioenergy with Carbon Capture and Storage (BECCS), both the global area and population exposed to severe water stress could approximately double compared with today (Stenzel et al., 2021).

Conversely, hybrid approaches can generate synergies. A global assessment of 114,555 reservoirs estimated that floating solar photovoltaics covering 30 % of suitable reservoir surfaces could theoretically generate approximately 9,434 TWh/yr while reducing evaporation by around 106 km³/yr (Jin et al., 2023). Such estimates represent technical potential rather than deployment targets, and potential ecological impacts need to be considered.

The way forward is to shift from project-by-project decisions to catchment-scale energy portfolios that jointly consider electricity generation, water demand, river connectivity, sediment transport and biodiversity, while expanding solar, wind, storage, regional grids and better use of existing infrastructure to reduce pressure to further fragment rivers through new hydropower plants.

6.5.3 Water–Climate–Biodiversity Nexus

Nature-based solutions can simultaneously strengthen water security, climate mitigation and biodiversity. Peatlands provide a particularly clear example. Although they occupy only around 3–4 % of the global land surface, they store up to one-third of the world's soil carbon. Approximately 12 % of the global peatlands are drained or degraded, accounting for around 4 % of global anthropogenic greenhouse-gas emissions (UNEP, 2022).

In Germany, peat soils released around 52.2 million tonnes of CO₂-equivalents in 2024, corresponding to about 8 % of the country's national greenhouse-gas emissions (UBA, 2026). Raising water tables can substantially reduce peat degradation and associated CO₂ emissions while increasing water retention and restoring wetland habitats. Productive use can continue

through paludiculture, including cultivation of reed, cattail, and other wetland-plants.

However, hydrological restoration alone does not automatically restore biodiversity. A European comparison of 320 rewetted and 243 near-natural fens demonstrated persistent differences in vegetation composition and ecosystem functioning (Kreyling et al., 2021). Therefore, rewetting must be combined with nutrient reduction, vegetation management, and catchment-scale restoration where necessary to restore broader ecosystem functions.

Increasing water retention in the landscape can also substantially reduce climate risks. European-scale modelling indicates that strategically implemented retention areas could, under approximately 3 °C global warming, reduce projected annual flood losses by the year 2100 from around € 44 billion to € 8.1 billion per year, and reduce the exposed population by approximately 84 % (Dottori et al., 2023). While such retention areas are not synonymous with floodplain restoration, the results illustrate the considerable economic value of creating space to store water.

The way forward is to shift landscape management from rapid drainage towards water retention by protecting and restoring wetlands, peatlands, floodplains and riparian corridors,

with success measured by improvements in hydrology, greenhouse-gas emissions, biodiversity, and avoided economic losses rather than hectares restored alone.

6.5.4 Water–Health Nexus

Water management is also a fundamental component of public health. Inadequate WASH (water, sanitation and hygiene) contributed to an estimated 1.4 million preventable deaths and 74 million DALYs (disability-adjusted life years) in 2019 (WHO, 2023b). Health risks extend beyond drinking-water supply and sanitation: nutrient enrichment can promote harmful algal blooms; wastewater transports pathogens, pharmaceuticals, and antimicrobial-resistance determinants; floods can mobilise sewage and contaminants; and droughts reduce dilution and concentrate pollutants.

Wastewater simultaneously represents a major opportunity for improving water security and health. In 2022, approximately 42 % of household wastewater was not safely treated before discharge, corresponding to around 113 billion m³ released to the environment with inadequate or no treatment (UN Water, 2024). Advanced treatment and resource recovery can instead produce reusable water and recover phosphorus, nitrogen, organic carbon, and energy

while reducing risks to aquatic ecosystems and human health.

More broadly, healthy catchments can function as preventive health infrastructure. Protecting drinking-water source areas, wetlands and riparian vegetation can reduce pollutants before they reach water-treatment plants, while wastewater surveillance can provide early warning of pathogens and antimicrobial resistance.

The way forward is to embed universal WASH within a One Health approach that integrates human, animal and environmental health through source-water protection, pollution prevention, wastewater treatment and reuse, resource recovery, health surveillance and ecosystem restoration.

6.5.5 Conclusion: towards a water-positive economy

The water–food–energy–climate–biodiversity–health nexus demonstrates that sectoral efficiency alone is insufficient. More efficient irrigation can increase total consumptive water use if irrigated areas expand; inexpensive solar pumping can accelerate groundwater depletion; hydropower can reduce fossil-carbon emissions while fragmenting rivers; and Bioenergy with Carbon Capture and Storage (BECCS) can

remove atmospheric carbon while increasing water stress. Solutions therefore need to be evaluated at the scale at which their wider hydrological, ecological, and socioeconomic consequences become visible.

The most promising interventions generate multiple benefits from the same investment. Rewetting peatlands can reduce greenhouse-gas emissions while retaining water and restoring wetland habitat; floodplain reconnection can reduce flood risk while rebuilding ecological connectivity; floating solar can generate electricity while reducing reservoir evaporation; diversified renewable-energy portfolios can expand electricity access while avoiding additional river fragmentation; and circular wastewater systems can recover water, nutrients, and energy while reducing pollution and health risks. These supply-side measures need to be complemented by shifts towards more sustainable consumption patterns, particularly among high-consuming populations.

These examples point towards what could be termed a water-positive economy: an economy in which investments are assessed not only by their efficiency within an individual sector, but by their contribution to justice, water security, food production, clean energy, climate mitigation and adaptation, biodiversity, and human health within hydrological and ecological limits.

The guiding question therefore shifts from how scarce water can be divided among competing sectors to how the same water can generate multiple societal, economic, and ecological benefits.

6.6 Harness AI for freshwater stewardship while curbing AI's water footprint.

Artificial intelligence (AI) is emerging as a transformative tool for freshwater research, conservation and water management. Applications range from automated species identification and water-quality assessment to forecasting droughts, floods, harmful algal blooms, invasive species, and ecological change. More fundamentally, AI could shift freshwater management from retrospective assessment towards predictive, near-real-time and adaptive decision-making. Combining Earth observation, environmental sensors, eDNA, and other biodiversity observations with hydrological and ecological models could identify biodiversity hotspots and climate refugia, prioritise conservation actions, and provide early warning of emerging risks. Digital twins, which are continuously updated computer replicas of real-world ecosystems, could further integrate observations and models to test alternative management scenarios before interventions are implemented (Ma et al., 2025; Reynolds et al., 2025).

Moving towards multi-objective decision support at the catchment scale provides a particularly important opportunity. AI could optimise environmental flows and reservoir operations while evaluating trade-offs among water supply,

food production, hydropower, flood protection, and biodiversity (Okola et al., 2025; Razavi et al., 2026). Rather than simply maximizing hydropower production, for example, reservoir operations could simultaneously consider environmental flows, water temperature and other ecological requirements, as demonstrated by recent applications integrating deep learning and multi-objective optimisation (Qiu et al., 2024; Zhu et al., 2026). However, many such applications remain at an early stage and are constrained by data gaps, limited model transferability and transparency, as well as institutional capacity.

Agriculture offers major opportunities because it accounts for roughly 70 % of global water abstractions. AI can integrate soil moisture, weather forecasts, crop status, satellite observations, and evapotranspiration to optimise irrigation. A recent synthesis of 135 studies reported 30–50 % water savings and 20–30 % yield improvements from AI-supported irrigation, although results vary substantially among crops, climates, and technologies (Oğuztürk, 2025). Importantly, greater efficiency does not necessarily mean more water for nature: reduced water use per unit of production can stimulate

expansion of irrigated agriculture or reallocation of saved water. Efficiency gains therefore need to be accompanied by limits on total abstraction volumes and a minimum level of environmental flows, which are essential for safeguarding freshwater biodiversity (Ruffing & Freed, 2026).

AI can also increase the scale and speed of biodiversity monitoring and enforcement. A multimodal machine-learning approach screened 86,321 YouTube posts and detected 3,517 associated with wildlife trade, involving 226 species, including 51 Critically Endangered and 62 Endangered species (Momeny et al., 2026). Similar approaches could provide early warning of trade in freshwater turtles, sturgeons, crocodilians, and ornamental fishes. More broadly, a horizon scan involving 27 conservation and AI experts identified 21 potentially transformative AI applications for biodiversity conservation, including species identification, ecosystem digital twins, wildlife-trade monitoring and improved biodiversity-loss predictions (Reynolds et al., 2025).

Yet, these capabilities create new ecological and societal risks. Increasingly accessible biodiversity data and powerful models could make rare

species easier to locate and exploit. ‘AI colonialism’ could reinforce inequalities over who collects and owns biodiversity data, builds models, defines conservation priorities, and benefits from the resulting knowledge. Increasing dependence on automated identification, modelling and decision support could also erode taxonomic and field-based expertise and create false confidence in opaque model outputs. AI should therefore augment rather than replace ecological expertise, requiring human oversight, transparency, equitable data governance and protection of sensitive biodiversity information (Reynolds et al., 2025).

At the same time, AI itself has a rapidly growing water, energy, land and material footprint. Data centres consume water directly for cooling and indirectly through electricity generation, while semiconductor production creates additional upstream demand. One scenario projects global AI-related water abstraction of 4.2–6.6 billion m³ annually by 2027, with 0.38–0.60 billion m³ consumed, although these estimates remain uncertain (Li et al., 2025).

Where and when this water is used matters as much as how much is used. Water abstraction from a water-stressed river or groundwater during drought has fundamentally different ecological consequences than equivalent use in a water-abundant catchment. Water consumption

per computing workload can vary by more than 10,000-fold, depending particularly on server efficiency, electricity source, server utilisation, cooling technology, and climate (Lei et al., 2025). This variability also demonstrates a substantial potential to reduce AI’s freshwater footprint through improved technology, siting, timing, and energy sourcing.

Further impacts arise from electricity generation, including cooling-water abstraction and thermal pollution, as well as from hydropower operations. Data-centre construction, transmission infrastructure and mineral extraction add land, water and biodiversity pressures. Assessing AI therefore requires a life-cycle and nexus perspective integrating water, energy, climate, land, materials and biodiversity, rather than focusing on carbon emissions alone.

The challenge is to make AI net-positive for freshwater systems by maximizing its capacity to monitor ecosystems, anticipate risks and improve decisions while minimizing the environmental footprint of its infrastructure. Efficiency gains should translate into absolute reductions in water abstraction; data centres should be located and operated according to catchment-level water availability and ecological risk assessments; and environmental impacts should be reported transparently across the life cycle.

Ultimately, AI should not simply help us exploit water and ecosystems more efficiently; it should help us remain within planetary boundaries. Used accordingly, AI could become a powerful tool for integrated, predictive and adaptive stewardship of freshwater resources and biodiversity.

6.7 Build a globally coordinated, near-real-time freshwater observatory

Reliable, long-term monitoring is a prerequisite for sustainably managing freshwater resources and halting biodiversity loss; yet, major observational gaps persist and have increased for some variables. Hydrological gauging networks remain indispensable because they provide the ground observations needed to detect long-term trends, calibrate models, and validate satellite products. However, the number of active river gauges contributing data to the Global Runoff Data Centre (GRDC) has declined substantially since the 1980s; by 2020, only 14 % of GRDC gauges were classified as active (Elmi et al., 2024). Monitoring coverage remains particularly sparse in many parts of Africa, Asia, and South America.

Freshwater biodiversity monitoring is similarly incomplete and geographically biased. A recent assessment based on respondents from 67 countries found systematic freshwater biodiversity monitoring to be lacking at national and subnational levels in many countries (Simaika et al., 2024), while long-term biodiversity observations remain disproportionately concentrated in Europe and North America (Proença et al., 2017). Even in comparatively well-monitored regions, sampling frequency and duration can

be insufficient to reliably assess changes in biodiversity. An analysis of 1,353 river-invertebrate time series from 18 European countries demonstrated that reducing either sampling frequency or monitoring duration substantially decreases the reliability of estimated biodiversity trends, particularly their magnitude (Cortés-Guzmán et al., 2026). Water-quality monitoring faces related limitations, as conventional periodic grab sampling can fail to capture short-term pollution events and rapid fluctuations in water chemistry. High-frequency sensors, automated and event-triggered sampling, and emerging analytical approaches can increasingly capture such temporally dynamic conditions.

Paradoxically, these gaps and the erosion of parts of the conventional monitoring infrastructure coincide with unprecedented opportunities to observe the water cycle and freshwater biodiversity. Earth observation, autonomous in-situ sensors, environmental DNA (eDNA), automated biological observations, artificial intelligence and hydrological and ecological modelling can increasingly complement conventional field monitoring across scales from individual water bodies and catchments to the entire globe. Satellite archives already allow reconstruction

of more than three decades of changes in global surface-water extent (Pekel et al., 2016), while the Surface Water and Ocean Topography (SWOT) mission substantially expands observations of water levels and surface areas of lakes, reservoirs, and large rivers. Remote sensing can also partially fill gaps in conventional discharge records: Elmi et al. (2024), for example, developed remote sensing-based extensions for 3,377 of 6,015 GRDC stations on rivers with mean discharge $>10 \text{ m}^3/\text{s}$. Such approaches can greatly expand spatial and temporal coverage, but they remain dependent on high-quality ground observations for calibration and validation and should therefore complement rather than replace gauging networks (Elmi et al., 2024).

At regional scales, operational systems demonstrate how observations and models can already be translated into actionable information. The UFZ German Drought Monitor, for example, provides spatially continuous, daily updated information on soil moisture, plant-available water, and drought conditions across Germany, while its historical dataset provides information at approximately 1.2-km spatial resolution. Integrating such observations with hydrological and ecological models and AI offers the

potential to shift monitoring progressively from retrospective assessment towards forecasting and early warning of droughts, floods, pollution events, and ecological risks.

A similar transformation is becoming possible for freshwater biodiversity. eDNA and metabarcoding can detect a wide range of taxa from water samples without conventional capture or direct observation and offer an increasingly scalable and standardisable approach to biodiversity assessment (Altermatt et al., 2025; Macher et al., 2026). High-frequency eDNA monitoring has recently demonstrated the capacity to detect more than 1,000 species across multiple trophic levels in a single freshwater system. In the same study, eDNA monitoring provided more than twice the biodiversity information of many conventional surveys at approximately one-sixth of the cost (Macher et al., 2026). Recent assessments conclude that many technical components are already sufficiently mature to contribute to a global measurement network for aquatic biodiversity, although method standardisation, taxonomic reference libraries, quality assurance, data interoperability and robust long-term sampling designs remain important constraints (Altermatt et al., 2025). Such approaches could be particularly valuable in biodiversity-rich but poorly monitored regions. Automated imaging, acoustic monitoring and AI-assisted taxonomic identification can further

complement eDNA and conventional field surveys and increase the spatial, temporal and taxonomic resolution of biodiversity observations.

Importantly, these new technologies should complement rather than replace long-term field observations. Their greatest potential lies in integrating gauging stations, ecological field surveys, eDNA, automated sensors, satellite observations, and models into interoperable observation systems. Such integration is also essential for linking changes in water quantity and quality directly to ecological responses rather than monitoring each independently.

Existing research infrastructures demonstrate important elements of such an integrated approach. In Europe, the Integrated European Long-Term Ecosystem, Critical Zone and Socio-Ecological Research Infrastructure (eLTER RI) applies a whole-system approach that integrates observations across the geosphere, hydrosphere, biosphere, atmosphere, and sociosphere. Its recently developed framework of Standard Observations combines harmonised variables, measurement methods and protocols across environmental compartments and provides an important model for integrated long-term ecosystem observation (Zacharias et al., 2026). Other major infrastructures, including the U. S. National Ecological Observatory Network (NEON) and the Chinese Ecosystem Research

Network (CERN), provide complementary long-term observational capacities.

The next step should be to connect such infrastructures and emerging technologies into a globally coordinated, near-real-time freshwater observation system that simultaneously tracks water quantity, water quality, ecological condition, and biodiversity and translates these observations into forecasting, early warning and decision support. Such a system could identify emerging droughts and floods, pollution events, alterations of ecological flow regimes, biological invasions, and biodiversity declines early enough for management to respond, while simultaneously providing the long-term observations required to distinguish directional environmental change from short-term variability. The objective should therefore not be real-time observation of every variable everywhere, but an integrated system in which high-frequency and near-real-time measurements are combined with long-term observations and repeated biological surveys at frequencies appropriate to the processes being monitored.

Globally interconnected infrastructures, sustained long-term investment, internationally agreed standards, open and interoperable data, and equitable access to monitoring technologies will be crucial. These need to be aligned with open-science principles, transparent

governance, ethical applications of AI, and equitable benefit-sharing. Data sovereignty requires particular attention: countries, local communities, and Indigenous Peoples should retain appropriate rights and authority over the collection, governance, sharing, and application of environmental and biodiversity data generated within their territories and ecosystems.

Rather than creating an entirely new institutional architecture, such investments should build on and connect established global initiatives, particularly GEO BON's Freshwater Biodiversity Observation Network (Freshwater BON). Essential Biodiversity Variables (EBVs) already provide an established framework for harmonising heterogeneous biodiversity observations and translating them into comparable measures of biodiversity state and change from local to global scales (Navarro et al., 2017; Pereira et al., 2013; Proença et al., 2017; Turak et al., 2017). Freshwater BON provides an existing global community for developing harmonised freshwater monitoring, mobilising and integrating data, strengthening monitoring capacity, and translating observations into EBVs and policy-relevant indicators.

Ultimately, a globally coordinated freshwater observation system should link hydrological, water-quality and biodiversity observations within a common framework and be explicitly

aligned with the monitoring frameworks of the Sustainable Development Goals and the Global Biodiversity Framework. This would provide the harmonised evidence needed to assess changes in the extent and condition of freshwater ecosystems, ecological connectivity, pollution, restoration, and biodiversity, while enabling emerging risks to be detected early enough for societies to respond. Such an integrated system would transform freshwater monitoring from a fragmented record of past change into an operational infrastructure for anticipating change and guiding water and biodiversity management.

6.8 Create the economic and financial conditions ensuring a water-positive transformation

The transition towards a nature-positive and resilient economy and society requires a fundamental shift in economic incentives, investment conditions, and financial flows. Increasing investment in conservation and restoration will not be sufficient if economic activities and financial flows continue to contribute to water stress and ecosystem degradation. The transition therefore needs to combine the phasing out of nature-negative activities and incentives with the simultaneous phasing in and scaling-up of nature-positive solutions, investments, and business models (see [Chapter 4.5](#)).

This requires action at two interconnected levels: (a) economic policy, which establishes the incentives and long-term framework conditions for the transition, and (b) the financial sector, which translates these conditions into investment, lending, and capital-allocation decisions. Public finance remains a central element, particularly where nature-positive activities generate substantial societal benefits but insufficient or uncertain private financial returns.

6.8.1 Economic policy

Economic policy should focus on creating the framework conditions for the transition and, in particular, on reviewing and reforming existing policies and incentives that contribute to nature degradation and water stress. This requires a gradual, credible, and predictable pathway for phasing out environmentally harmful subsidies, incentives, and activities that contribute to inefficient water use, unsustainable land use, ecosystem degradation, and resource over-extraction. This should include reviewing agricultural and energy-related subsidies, reforming incentives for water-intensive production, and progressively aligning water abstraction charges and other resource prices with scarcity and environmental costs (see [Chapter 5.4](#)). Clearly communicated transition periods should allow businesses, landowners, and land users to adjust their investments and business models.

At the same time, economic policy should actively accelerate the phasing in of nature-positive activities, including water-efficient agriculture, sustainable land management, nature-based solutions, ecosystem restoration, circular production, and resilient water

infrastructure. Public procurement, targeted investment programmes, technical assistance, and outcome-based support can help bring these solutions to scale.

Clear, consistent, and long-term policy frameworks are decisive. Governments should establish predictable rules for water management, land use, environmental standards, permitting, resource pricing, and public support schemes. Where possible, support programmes should provide multi-year or long-term certainty, particularly for investments whose returns depend on ecological processes that may take many years. Policy frameworks should be reviewed periodically, and changes should be transparent and predictable so as not to undermine existing private investments.

Those managing land and water resources should have clear economic incentives to provide ecosystem services. Long-term Payments for Ecosystem Services (PES) and results-based contracts should compensate landowners and users for measurable ecosystem services outcomes such as water retention, groundwater recharge, wetland restoration, soil protection, and biodiversity conservation. Where benefits

accrue to water utilities, municipalities, businesses, or communities, appropriate cost-sharing mechanisms should ensure that those benefiting from ecosystem services contribute to their provision.

Natural capital accounting (e. g., GDP accounting) should provide an important basis for these policies. Natural capital and ecosystem services should be systematically reflected in public planning and economic decision-making alongside conventional economic indicators. Natural capital accounts should directly inform decision-makers in infrastructure investment, agricultural and water policies, and assessments of long-term economic and fiscal risks.

6.8.2 Financial sector and capital mobilisation

Clear and consistent public policies are also a prerequisite for mobilising private financial capital. However, many investments in water security, ecosystem restoration, and nature-based infrastructure generate substantial public benefits but cannot generate sufficient private revenues on their own. Public funding will always be necessary to finance public goods and ecosystem services and cannot be substantially replaced by private capital. Large parts of Nature are not investible.

Public resources should nevertheless be used more strategically and extensively to crowd-in additional private investment where it is possible (see [Chapter 5.5](#)). This includes funding project preparation and technical assistance, providing grants and concessional finance, and using guarantees, first-loss structures, and other risk-sharing instruments where these can unlock private investment and blended finance mechanisms. The objective should be to deploy public finance where it can have the greatest impact, while maintaining public oversight and appropriate risk allocation.

Nature-positive investments should increasingly be structured around predictable revenue streams and appropriate risk allocation. Relevant models include long-term ecosystem-service contracts, nature-based infrastructure funds, Green and Blue Bonds, sustainability-linked finance, Public-Private Partnerships, results-based financing, and Payments for Ecosystem Services. As mentioned above, for projects that generate substantial public benefits but cannot achieve sufficient market revenues, long-term public or regulated payment mechanisms may be required to make investment viable.

Banks, insurers, pension funds, asset managers, and other financial institutions should systematically assess dependencies and risks related to

water and ecosystems and integrate them into their lending, investment, pricing, and risk-management decisions. Standardised approaches for identifying, assessing, managing, and disclosing nature-related dependencies, impacts, risks, and opportunities, such as the TFND, should be further developed and implemented (see [Chapter 5.5](#)). Disclosure should ultimately translate into changes in capital allocation rather than remain a reporting exercise.

Economic policy and financial-sector action should be understood as mutually reinforcing components of the same transformation. Economic policy establishes the incentives and predictable framework conditions, while the financial sector translates these conditions into capital-allocation decisions. Together, these elements can support the simultaneous phasing out of nature-negative activities and financial flows and the phasing in of nature-positive investments and business models.

6.9 Establish a Blueprint for Freshwater Life: prioritise protection, enhance restoration

Despite decades of national, European and global legislation, restoration programmes and political commitments, freshwater biodiversity continues to decline at an alarming rate, demonstrating a persistent gap between political ambition and implementation (Cooke et al., 2026; Dudgeon & Strayer, 2025; Tickner et al., 2020).

A Blueprint for Freshwater Life is therefore urgently needed to provide a coherent, science-based agenda for protecting and restoring freshwater ecosystems, their biodiversity, and processes and services. Building on the Emergency Recovery Plan for Freshwater Biodiversity (Tickner et al., 2020) and its implementation roadmap (Cooke et al., 2026), the Blueprint should accelerate six priority actions: (1) implementing environmental flows; (2) improving water quality; (3) protecting and restoring critical habitats; (4) managing the exploitation of freshwater species and riverine aggregates; (5) preventing and controlling invasive non-native species; and (6) safeguarding and restoring freshwater connectivity.

Most importantly, the Blueprint should prioritise protecting what remains intact: free-flowing rivers, unpolluted and near-natural lakes, intact

wetlands, and high-quality groundwater systems. Preventing degradation is more effective and less costly than restoring ecosystems after their structure, connectivity, and ecological functions have been degraded or lost.

The Blueprint should not create another layer of policy. Rather, it should provide the scientific framework, spatial priorities, measurable targets, implementation pathways and monitoring needed to translate existing commitments into coordinated action. In Europe, it could directly support implementation of the Water Framework Directive, Habitats Directive, EU Biodiversity Strategy for 2030, and Nature Restoration Regulation, while globally contributing to the implementation of the Kunming–Montreal Global Biodiversity Framework.

The potential for synergy is substantial: the EU has committed to restoring at least 25,000 km of rivers to a free-flowing state by 2030, and the Nature Restoration Regulation requires Member States to identify and remove barriers and restore river connectivity to contribute to this objective. These measures can simultaneously advance the ecological objectives of the Water Framework Directive. The Blueprint should

ensure that success is measured by improvements in ecological conditions and biodiversity, not simply by measures implemented, and place freshwater biodiversity at the core of water management rather than treating it as a secondary consideration.

We propose two ambitious activities that could form core responsibilities of research institutions benefiting from stable, long-term public funding. Such institutions are uniquely positioned to provide the continuity, scientific independence, infrastructure and long-term observations needed to move from short-term projects towards sustained freshwater biodiversity recovery.

6.9.1 A Synthesis Centre for Freshwater Life

Freshwater biodiversity has been monitored for decades, yet much of the resulting baseline and long-term data remains fragmented, difficult to access or unavailable for synthesis. This limits our ability to quantify biodiversity change, identify its drivers, and develop targeted conservation and restoration actions. A Synthesis Centre

for Freshwater Life is therefore needed to mobilise, aggregate, curate, and integrate long-term data on genes, species, communities, traits and ecosystem processes across German, European, and global freshwater systems. By linking these data with major environmental drivers, including climate change, pollution, hydrological alteration, habitat loss and fragmentation, and invasive species, the Centre would enable state-of-the-art analyses of biodiversity status, trends and drivers, as well as predictive models and scenarios. Virtual Research Environments and AI-supported workflows would make complex and heterogeneous datasets increasingly accessible and interoperable, while supporting transparent and reproducible analyses.

The Centre would serve as a hub for transforming fragmented data into knowledge, and knowledge into action. It would bring together leading international experts and establish long-term partnerships with water managers, policymakers, public authorities, NGOs, industry, and civil society. Through scientific syntheses, accessible data infrastructures, decision-support tools, and actionable recommendations, the Centre would strengthen the evidence base for freshwater conservation, restoration, and sustainable water management, while supporting the implementation and evaluation of European and global biodiversity and sustainability targets. In the longer term, it should evolve into a leading

international platform for understanding, anticipating and responding to freshwater biodiversity change.

6.9.2 Towards a Global Oasis Alliance

Oases are among the most important biocultural systems of the world's drylands. In Asia and North Africa alone, they cover approximately 400,000 km², corresponding to only about 1.5 % of the region's arid and hyper-arid lands. However, around 150 million people live within these oases, and approximately 418 million live in or within 10 km of them, many of whom depend on oasis resources and ecosystem services. Oases also support exceptional biological and cultural diversity: more than 8,200 vertebrate species have been recorded in oases globally, more than 13 % of which are threatened. Despite their importance, only 0.32 % of their total area is currently protected, placing oases among the least protected ecosystems worldwide (Hernández-Agüero et al., 2025).

At the same time, oases are increasingly threatened by groundwater depletion, salinisation, climate change, land-use transformation, urbanisation and the erosion of traditional knowledge. As water scarcity, salinisation and land degradation undermine agriculture and local livelihoods, human migration can become an important

coping and adaptation strategy, contributing to population decline and abandonment in severely affected oases (e.g., Sobczak-Szelc, 2024). At the same time, recent assessments highlight the close interdependence of their biological and cultural diversity and the urgent need to safeguard both simultaneously (Bretzke et al., 2025; Hernández-Agüero et al., 2025).

We therefore call for the establishment of a Global Oasis Alliance that brings together oasis countries and communities, UN agencies, research institutions, and other partners around a shared framework and agenda for protecting, restoring, and sustainably managing oases. Priorities should include a Global Oasis Observatory and a continuously updated knowledge platform, a network of flagship pilot sites and long-term observatories, common indicators for assessing oasis resilience, and the systematic integration of scientific, local, and traditional knowledge. Particular attention should be given to sustainable groundwater management, as the long-term persistence of many oasis systems ultimately depends on maintaining their underlying water resources.

The Alliance could position oases much more prominently within global water, biodiversity, climate, land, and cultural-heritage agendas and provide a platform for translating knowledge into coordinated action. It could build

on existing international frameworks and partnerships and potentially operate under, or in close cooperation with, the United Nations Convention to Combat Desertification (UNCCD). Ultimately, a Global Oasis Alliance should recognise oases not simply as isolated green spaces in drylands, but as coupled social-ecological systems whose future depends on safeguarding water resources, biodiversity, and cultural heritage together.

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Supplement: water stress indicators

The most widely used Indicator of water scarcity for humans is the water abstraction-to-renewable water resources ratio I_{was} .

$$I_{was} = WA/RWR$$

where WA: water abstractions, RWR: renewable water resources

I_{was} considers that any abstractions lead to stress, mainly because returned water is likely to have lower quality and because it may not return to the same place/compartiment where it was abstracted. An alternative water stress indicator is I_{cus} (referred to as WS_{phys} in the main text), with

$$I_{cus} = CU/RWR$$

where CU: consumptive water use, the fraction of the abstracted water that evapotranspirate during use.

I_{cus} only considers the part of the water that, after use, is no longer in liquid form (focusing strongly on irrigation water use). Neither water scarcity indicator accounts for the possibility that the local situation may not be stressed because of water inflow from upstream.

While I_{was} and I_{cus} are mostly aggregated to countries or river basins, [Figs. S1](#) and [S2](#) show the indicators for 0.5 arc-deg x 0.5 arc-deg grid cells, where, to take into account short-range water transfers, the average water abstractions and renewable water resources over the grid cell and its surrounding eight grid cells (if existing) are taken into account to compute the grid cell indicator value. This provides a spatially more

resolved indicator of water scarcity compared to basin-average or country-average values.

Renewable water resources (RWR): 30-year average (1990–2019) of net cell runoff under naturalized conditions (data from WaterGAP 2.3, climate forcing W5E5). Net cell runoff of each 0.5 arc-deg grid cell is computed as the difference between the outflow from the cell and the inflow(s) to the cell.

Water abstractions (WA) and consumptive use (CU) in 2015 (but with irrigation water use not for 2015 and its weather but considering irrigation water demand for the irrigated areas of 2015 under long-term climatic condition 1990–2019) (output of GWSWUSE): Irrigation water use: based on irrigated area in 2015 (run GIM with constant AAI of 2015) and the climate 1990–2019, then average over the 30 years to obtain one value for irrigation water abstractions and one

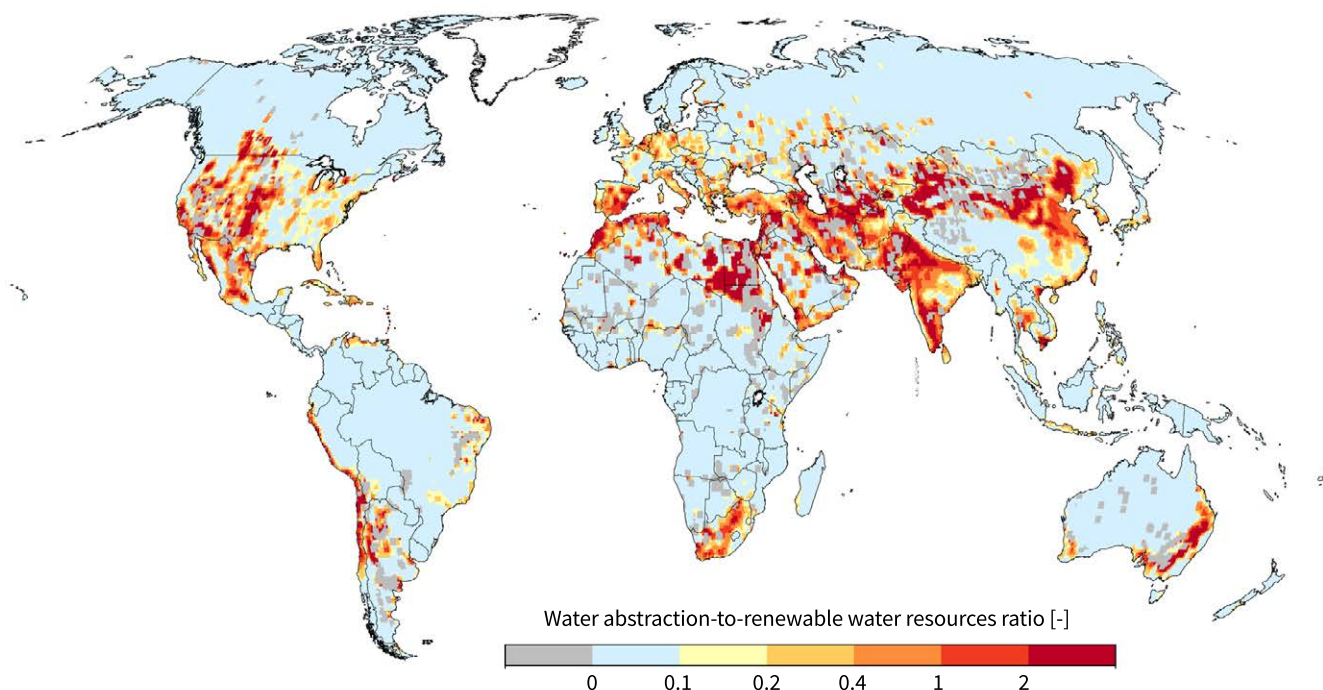


Fig. Supplement 1 Water stress for humans as indicated by the water abstraction-to-renewable water resources ratio. Water abstractions in 2015 and renewable water resources in 1990–2019 as computed by WaterGAP 2.3. Renewable water resources are defined as the long-term mean runoff, the difference between long-term mean precipitation and evapotranspiration. Negative values indicate that water inflow from upstream is the source of water use.

value for irrigation consumptive use; then add this values to the other sectoral uses for 2015 (data from WaterGAP 2.3, climate forcing W5E5).

In humid areas, the temporal variability of runoff and streamflow is generally lower than in semi-arid and arid regions, both seasonally and interannually. It can be assumed that for the same values of renewable water resources, a higher water variability makes the provisioning of water for human use more difficult, unless all the water is sourced from deep groundwater. Two indicators show this vulnerability. Ivar is only based on the variability of monthly net cell runoff. Ivar_sw combines the variability with the relative dependence of the human water supply on surface water abstractions.

Both indicators are based on monthly time series of net cell runoff NCR under anthropogenic conditions in the period 1990–2019 as computed by WaterGAP 2.3 (climate forcing W5E5) from the difference between outflow from each 0.5 arc-deg grid cell and inflow(s) to the cell.

$$\text{Ivar} = (\text{NCR}_{10} - \text{NCR}_{90}) / \text{NCR}_{50}$$

where NCR₁₀: monthly NCR that is exceeded in 1 out of 10 months

NCR₅₀: median NCR

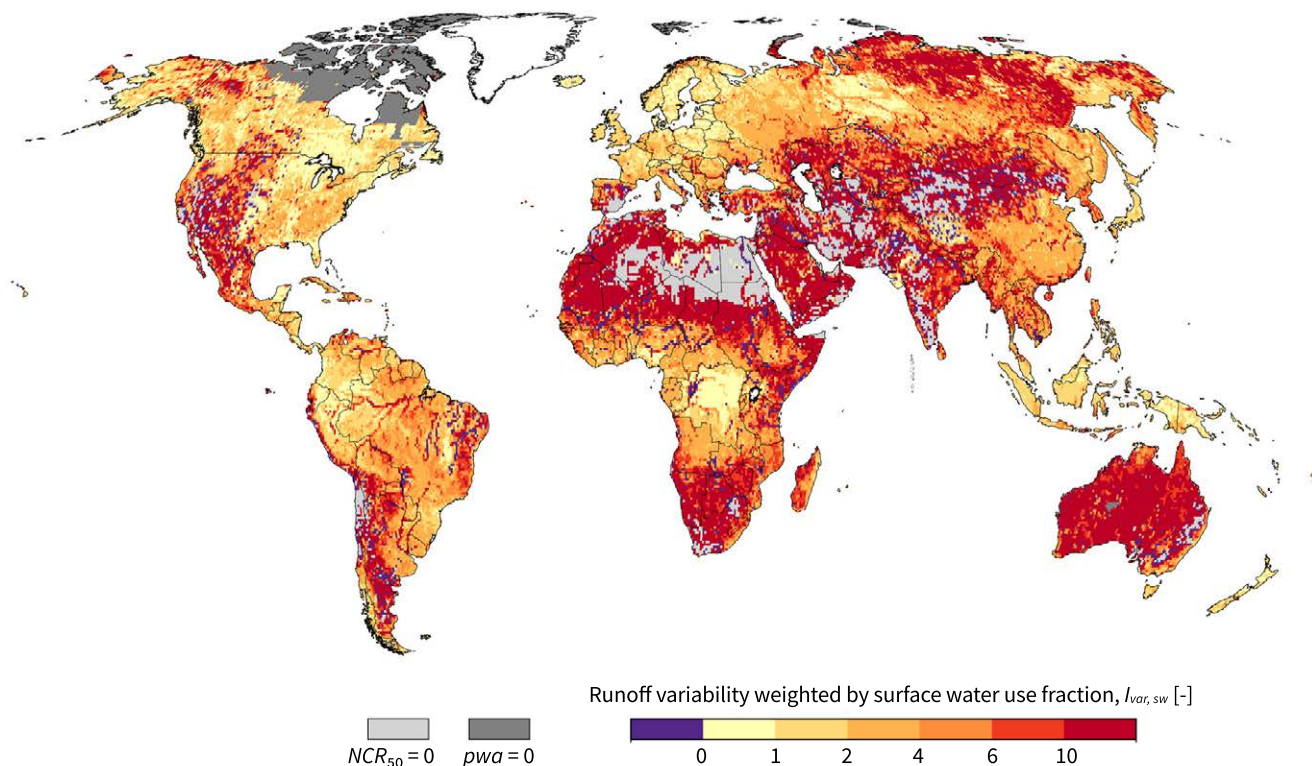


Fig. Supplement 2 Vulnerability of human water supply due to the temporal variability of runoff, taking into account the source of water supply. The higher the fraction of total water abstractions taken from groundwater, the lower the vulnerability. The larger I_{var_sw} is, the higher the vulnerability. If I_{var_sw} is negative (due to negative NCR_{50}), there is a particularly high vulnerability, as people and freshwater ecosystems depend on water inflow from upstream. (Computed for the time period 1990-2019 by WaterGAP 2.3)

NCR_{90} : monthly NCR that is exceeded in 9 out of 10 months.

$$I_{var_sw} = I_{var} * \left(\frac{\text{pot_water_abstractions_from_sw}}{\text{pot_water_abstractions_total}} \right)$$

The higher the indicator values, the more vulnerable human water users are. To compute I_{var_sw} , I_{var} is multiplied by the fraction of water abstractions from surface water, indicating that lower reliance on surface water for water supply (and a concurrent higher reliance on groundwater, which provides substantial natural water storage) decreases vulnerability. Where both indicators are negative (due to negative NCR_{50}), there is a particularly high vulnerability, as people and freshwater ecosystems depend on water inflow from upstream.

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