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Review article

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# Global water resources and the role of groundwater in a resilient water future

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## Abstract

Water is a critical resource, but ensuring its availability faces challenges from climate extremes and human intervention. In this Review, we evaluate the current and historical evolution of water resources, considering surface water and groundwater as a single, interconnected resource. Total water storage trends have varied across regions over the past century. Satellite data from the Gravity Recovery and Climate Experiment (GRACE) show declining, stable and rising trends in total water storage over the past two decades in various regions globally. Groundwater monitoring provides longer-term context over the past century, showing rising water storage in northwest India, central Pakistan and the northwest United States, and declining water storage in the US High Plains and Central Valley. Climate variability causes some changes in water storage, but human intervention, particularly irrigation, is a major driver. Water-resource resilience can be increased by diversifying management strategies. These approaches include green solutions, such as forest and wetland preservation, and grey solutions, such as increasing supplies (desalination, wastewater reuse), enhancing storage in surface reservoirs and depleted aquifers, and transporting water. A diverse portfolio of these solutions, in tandem with managing groundwater and surface water as a single resource, can address human and ecosystem needs while building a resilient water system.

## Sections



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## Key points

- Net trends in total water storage data from the GRACE satellite mission range from −310 km$^3$ to 260 km$^3$ total over a 19-year record in different regions globally, caused by climate and human intervention.

- Groundwater and surface water are strongly linked, with 85% of groundwater withdrawals sourced from surface water capture and reduced evapotranspiration, and the remaining 15% derived from aquifer depletion.

- Climate and human interventions caused loss of ~90,000 km$^2$ of surface water area between 1984 and 2015, while 184,000 km$^2$ of new surface water area developed elsewhere, primarily through filling reservoirs.

- Human intervention affects water resources directly through water use, particularly irrigation, and indirectly through land-use change, such as agricultural expansion and urbanization.

- Strategies for increasing water-resource resilience include preserving and restoring forests and wetlands, and conjunctive surface water and groundwater management.

## Introduction

An estimated 80% of the world's population faces high threats to water security[1], with global water scarcity[2,3] increasing because of climate forcing, population growth and economic development. Increasing climate extremes (droughts and floods)[4] result in growing spatiotemporal disconnects between water supply and demand, making it increasingly difficult to ensure reliable water and food supplies across the planet. Owing to extreme droughts and almost complete reliance on surface water[5], some major cities have found themselves on the brink of running out of water, including São Paulo in 2015[6] and Cape Town in 2018[7]. These causes highlight the need to expand and diversify water supplies ahead of expected increases in climate extremes in the coming decades.

Water scarcity can be combatted through more sustainable water use, broadly defined as development that can be maintained indefinitely while minimizing adverse social, economic and environmental impacts[8]. Moving toward more sustainable water management requires integrating multiple dimensions of water security, including human and ecosystem health, environmental and social justice, climate adaptation and satisfying competing sectoral demands (agriculture, industry and energy). These dimensions have trade-offs. For example, investments in engineered water infrastructure (such as large surface reservoirs) have reduced threats to human water security by 95% in high-income countries[1] but adversely affected downstream communities and ecosystems. Trade-offs become even more apparent in the context of the UN sustainable development goals (SDGs), as some goals short-circuit progress on others. For example, SDG-1 focuses on alleviating poverty through rapid economic development, which historically threatens the integrity of water-resource systems emphasized under SDG-6.

Water is central in the response to climate change and socioeconomic development, particularly recognizing water as a central link within the climate system, the world economy and life support systems[9] (Fig. 1). As a result, the global dialogue surrounding water security has taken on new momentum. Water-security-related initiatives have materialized across the highest levels of government, business and civil society[10–14]. Between 2012 and 2020, water crisis appeared eight times among the top five high impact risks listed by the World Economic Forum[10], and the 77th UN General Assembly in 2022 has issued a red alert on climate and water supply[15].

In this Review, we discuss the status and trends in water development globally, emphasizing groundwater, the controls on water-resource systems, and solutions that can be adopted to increase the resilience of those systems. Previous reviews have primarily focused on either surface water or groundwater and have separated quantity and quality issues. Here, we provide a comprehensive assessment of current and historic freshwater supplies considering surface water and groundwater as a single resource. Additionally, we reanalyse global satellite data to assess water resources (Supplementary Information). We evaluate dominant controls of water-resource variability to inform effective solutions to pressing water-resource challenges. We conclude by reviewing complementary nature-based solutions and traditional engineered approaches, discussing their potential interactions, benefits and trade-offs in enhancing the resilience of water-resource systems.

## Water-resource status and evolution

The rising emphasis on the globalization of water-resource challenges has been aided by the proliferation of satellite data[16], global models[17], global datasets[1] and information on global food trade[18]. Increasing global data coverage of water uses and stressors, including watershed disturbance, pollution and biotic factors, highlights the importance of assessing contemporary water challenges[1]. For example, the availability of satellite data for global water-resource assessments has rapidly expanded in the past two decades[19]; however, hydrological systems have been evolving for over a century.

In this section, satellite data are reanalysed and placed in the context of the longer-term evolution of water resources by leveraging a large body of previous modelling and ground-based monitoring studies. In particular, the ability to study and quantify long-term trends in water storages and fluxes (Fig. 1) has been enabled by advances in global and regional modelling, including models that represent complex hydrological processes such as surface water and groundwater interactions.

### Multi-decadal variability in water resources

Gravity Recovery and Climate Experiment (GRACE) satellites provide a broad overview of water-resource change, as the data products have coarse resolution (~300 km or ~90,000 km$^2$)[20] and allow evaluation of changes in terrestrial total water storage (TWS) over time, integrating snow, surface water, soil moisture and groundwater, over the past two decades[19,21]. Early GRACE studies emphasized depletion trends in water storage and attributed most depletion to overexploitation of groundwater[22]. However, contemporary analyses emphasize dynamic TWS variability in many aquifers globally[23]. Thus, storage trends should be considered within the context of natural interannual variability, with trends exceeding three standard deviations of interannual variability (trend to interannual variability ratios) defined as exceptional in the original study[24].

Owing to difficulties in comparing GRACE TWS results from different studies because of varying time periods, GRACE data from April 2002 through August 2021 were reanalysed here (Fig. 2). Hotspots of TWS depletion (shown by large declining TWS trends) are found in

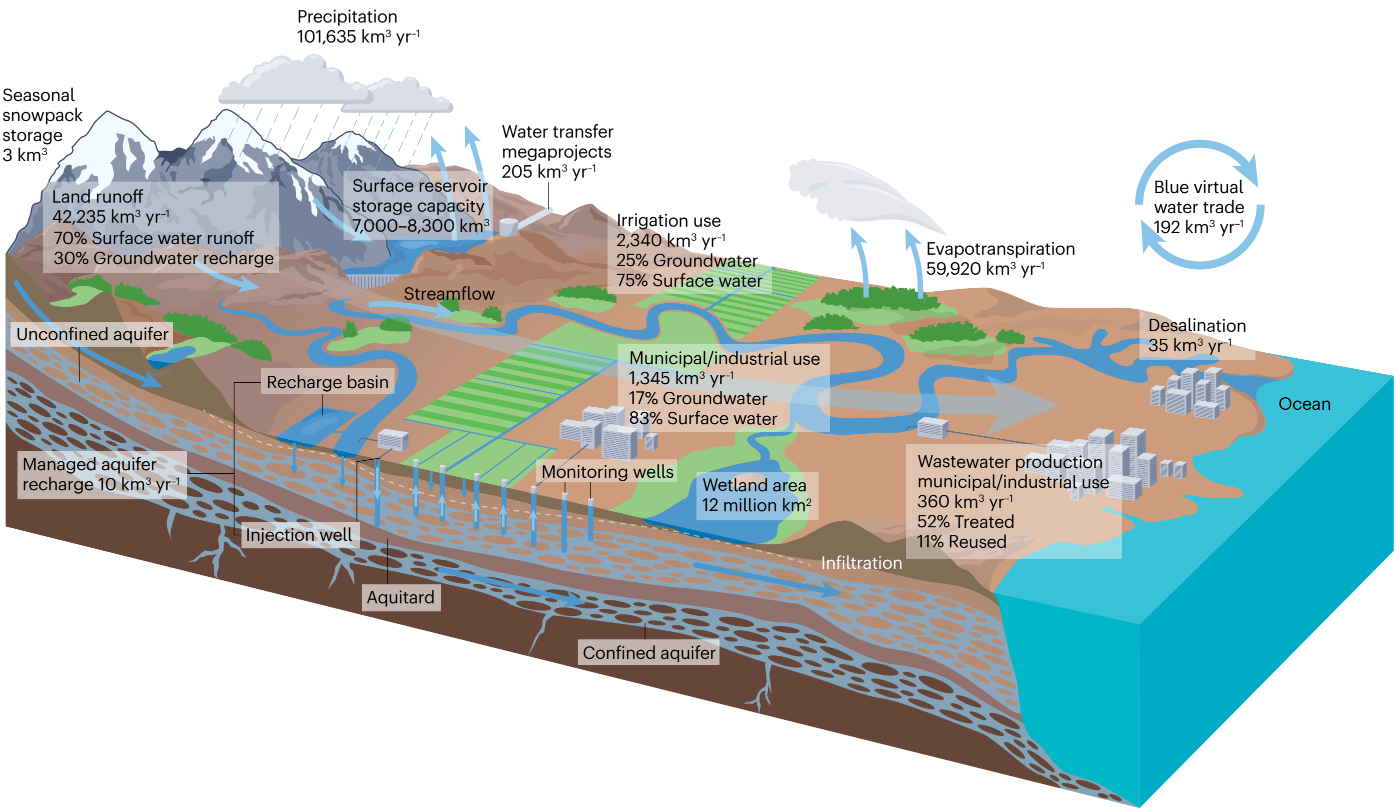


**Fig. 1 | The water cycle.** The figure includes representative values of global annual water storages and fluxes. Data and corresponding references are provided in Supplementary Tables 1 and 2.

northeast China (the Hai River Basin and North China Plain aquifer), northern India (Ganges–Brahmaputra aquifer), northeast South America (São Francisco basin), southwest and south-central United States (Central Valley and central plus south High Plains aquifers), eastern Europe (Don and Dnieper basins) and Middle East (Arabian Peninsula aquifer, Iran) with declines ranging from ~6–19 mm $yr^{-1}$ (Fig. 2, Table 1). Examples of areas with rising TWS trends include West Africa (Iullemeden aquifer, Niger and Volta basins), the Upper Kalahari aquifer, the northern United States (north High Plains aquifer, St Lawrence basin) and Central Canada. Net TWS trends range from −310 $km^3$ (Iran) to 261 $km^3$ (Niger Basin) over the 19-yr GRACE record. Decadal TWS trends dominate over interannual variability in many of these regions, whereas others exhibit more pronounced interannual variability relative to TWS trends (for example, the Upper Kalahari aquifer).

Variations in surface water bodies have been documented by Landsat satellite images. Between 1984 and 2015, ~90,000 $km^2$ of previously permanent surface water bodies disappeared, whereas 184,000 $km^2$ of new permanent water bodies developed elsewhere[25]. Most increases were linked to reservoir filling, with ~1,639 $km^3$ of large surface reservoir capacity built during this timeframe (calculated from GRanD database)[26]. Surface water losses occurred predominantly in the Middle East and Central Asia, related to drought and human intervention, including river diversions and unregulated water withdrawals.

Global hydrological models are also widely used to evaluate spatiotemporal variability in water resources. Advances in global modelling include calculation of TWS to allow direct comparisons with GRACE TWS data. Comparison of global models with GRACE TWS shows that the models underestimate declining and rising TWS changes, with opposite trends in land water storage between models (negative) and GRACE (positive) when all basins are considered[27].

## Surface water and groundwater as a single resource

Despite being commonly regulated and managed as separate resources, surface water and groundwater are inherently interconnected and intersect at streambeds, floodplains, wetlands and springs (Supplementary Information Section 3). Indeed, groundwater discharge accounts for ~50% of total annual streamflow (baseflow) in the United States[28], highlighting the need to conjunctively manage groundwater and surface water as a single resource.

Under natural (predevelopment) conditions, most aquifers are in a state of dynamic equilibrium, in which average inflows (recharge) balance average outflows, with no change in the average volume of water stored in the aquifer (Fig. 3). After development begins, this pumping or development creates an imbalance (Fig. 3b) that must be compensated by decreased groundwater discharge to surface water, increased recharge and/or a reduction (depletion) in groundwater storage, restoring the water balance (Fig. 3c). During early stages of aquifer development, pumping is balanced by storage depletion. Over time, pumping is increasingly derived from capture, including decreased discharge to streams and decreased groundwater-fed evapotranspiration[29], and/or increased recharge from surface water caused by declining water

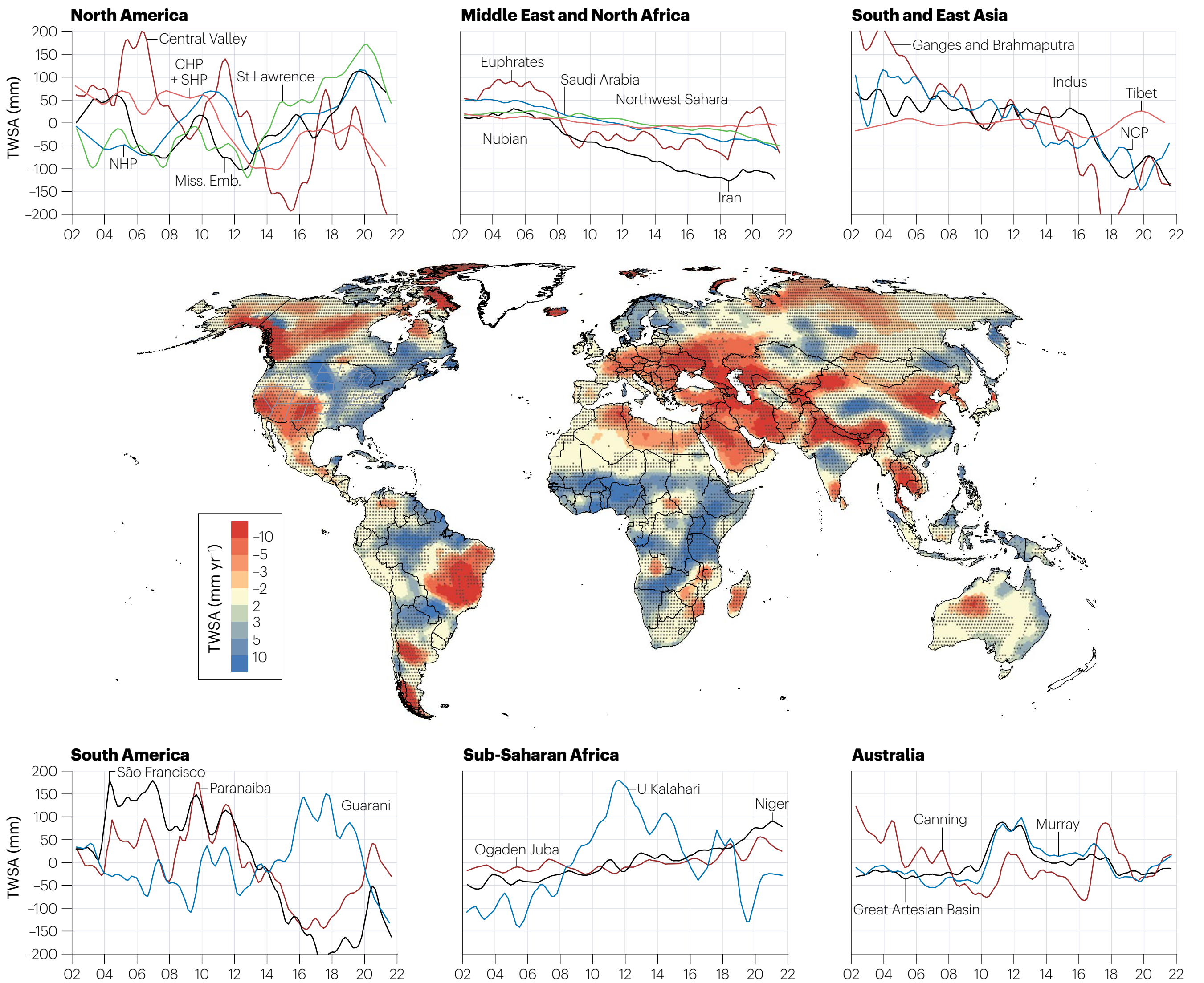


**Fig. 2 | GRACE satellite total water storage trends.** Trends in total water storage (TWS) anomalies from GRACE satellite data (2002–2022), based on the mean of mascon (mass concentration) solutions from the University of Texas Center for Space Research (CSR) and NASA Jet Propulsion Laboratory (JPL). Stippling reflects areas where TWS trends exceed three standard deviations of interannual variability. Time series of GRACE TWS anomalies are shown for representative aquifers and basins. Trends and related data are provided for aquifers in Supplementary Table 3a and for river basins in Supplementary Table 3b. TWS anomaly time series for selected aquifers and basins are in Supplementary Table 3c. Larger versions of time-series graphs for all regions plus Central Asia are in Supplementary Fig. 1a–c. CHP, NHP and SHP, central, northern and southern High Plains (USA); Miss. Emb., Mississippi Embayment; NCP, North China Plain; TWSA, total water storage anomaly.

tables. Therefore, the importance of groundwater storage depletion decreases with time, and capture of surface water increases as a result of decreased discharge to streams and ET. In fact, the magnitude of capture is much greater than storage loss.

In the United States, for example, an estimated 85% of groundwater abstraction was derived from capture of surface water or ET, with the remaining 15% derived from groundwater storage depletion[29]. The latter estimates are similar to values from previous studies, including global groundwater depletion estimates totalling 4,500 km$^3$ from 1900 to 2008 using calibrated groundwater models and volumetric budgets that consider capture[30]. This total groundwater depletion over the past century compares with annual groundwater use of ~800 km$^3$ yr$^{-1}$ (Fig. 1).

Groundwater depletion estimates from global hydrological models (such as the PCR-GLOBWB model)[3,31] were originally based on groundwater abstraction minus recharge (flux approach), which ignores the connection between groundwater and surface water. Advances in global models consider the strong interaction between groundwater and surface water. Simulated cumulative groundwater storage depletion (1960 to 2010) ranged from ~27,000 km$^3$ (flux approach; ignoring capture and evapotranspiration variations) to 4,200 km$^3$ (including capture and ET), equivalent to ~15% of the

**Table 1 | Water storage trends for major basins and aquifers**

| Basin or aquifer name | B/A | Area (km²) | TWS trend (mm yr⁻¹) | TWS trend (km³ in 19 yr) | TIVR | $R^2$ of TWS trend | *P* vs TWS |
|---|---|---|---|---|---|---|---|
| Hai | B | 163,076 | −19.3 | −60.9 | −13.83 | 0.94 | 0.26 |
| North China Plain | A | 437,748 | −11.5 | −97.1 | −7.6 | 0.82 | 0.30 |
| Ganges–Brahmaputra | A | 634,565 | −18.9 | −231.5 | −7.53 | 0.84 | 0.79 |
| São Francisco | B | 613,748 | −18.7 | −221.9 | −4.97 | 0.67 | 0.60 |
| Central Valley | A | 59,425 | −12.1 | −13.9 | −3.08 | 0.45 | 0.61 |
| North Caucasus | B | 295,468 | −10.6 | −60.4 | −4.86 | 0.64 | −0.02 |
| Dnieper | B | 513,997 | −9.6 | −95.0 | −6.09 | 0.76 | −0.27 |
| Don | B | 424,826 | −9.2 | −75.6 | −4.37 | 0.64 | −0.03 |
| Syr Darya | B | 417,802 | −6.2 | −50.2 | −4.22 | 0.57 | 0.63 |
| Euphrates | B | 761,713 | −6.1 | −90.2 | −3.01 | 0.45 | 0.74 |
| Iran | — | 1,648,268 | −9.9 | −309.9 | −12.05 | 0.93 | 0.72 |
| Arabian | B | 1,771,399 | −5.8 | −199.7 | −23.19 | 0.98 | 0.05 |
| Arizona Alluvial | A | 225,404 | −7.9 | −34.5 | −6.76 | 0.80 | 0.10 |
| Upper Colorado | A | 369,857 | −3.0 | −21.3 | −3.75 | 0.58 | 0.02 |
| C+S High Plains | A | 203,400 | −6.8 | −26.4 | −3.24 | 0.52 | −0.05 |
| Paranaiba | B | 319,365 | −7.3 | −45.1 | −2.12 | 0.27 | 0.77 |
| Danube | B | 806,131 | −4.8 | −74.6 | −2.83 | 0.43 | −0.25 |
| Paris | A | 171,968 | −3.6 | −11.8 | −3.58 | 0.52 | 0.22 |
| Russian Platform | B | 2,834,585 | −3.1 | −169.6 | −2.80 | 0.38 | −0.39 |
| Volga | B | 1,406,728 | −3.1 | −83.1 | −2.32 | 0.34 | −0.32 |
| Tarim | A | 468,333 | −2.6 | −28.8 | −10.84 | 0.91 | 0.11 |
| Lena | B | 2,346,188 | −2.3 | −105.4 | −2.61 | 0.33 | −0.69 |
| Nubian | A | 2,203,920 | −1.4 | −59.9 | −5.09 | 0.72 | 0.09 |
| Great Artesian | A | 1,727,400 | 0.9 | 30.4 | 0.55 | 0.03 | 0.41 |
| Murray | B | 1,069,567 | 1.1 | 23.3 | 0.60 | 0.06 | 0.26 |
| Ogaden–Juba | A | 1,035,211 | 2.2 | 44.8 | 3.38 | 0.57 | 0.28 |
| Guarani | A | 1,865,481 | 2.8 | 100.2 | 0.85 | 0.05 | 0.46 |
| Yangtze | B | 1,831,074 | 3.0 | 104.7 | 4.92 | 0.67 | 0.84 |
| Godavari | B | 327,126 | 3.6 | 22.5 | 1.49 | 0.14 | 0.83 |
| MERAS | A | 202,960 | 3.8 | 15.0 | 1.29 | 0.08 | 0.35 |
| Columbia Plateau | A | 114,178 | 4.3 | 9.5 | 4.14 | 0.58 | 0.70 |
| Iullemeden | A | 594,821 | 5.1 | 58.8 | 9.28 | 0.88 | 0.69 |
| Niger | B | 2,123,649 | 6.3 | 260.1 | 10.10 | 0.91 | 0.91 |
| Volta | B | 377,631 | 8.3 | 60.3 | 6.86 | 0.12 | −0.36 |
| Upper Kalahari | A | 989,348 | 5.7 | 108.6 | 1.33 | 0.10 | 0.67 |
| Northern High Plains | A | 250,965 | 6.1 | 29.2 | 2.84 | 0.37 | 0.42 |

A, aquifer; B, river basin; C+S, central and southern; *P*, precipitation; TIVR, trend to interannual variability ratio; TWS, total water storage. TWS trends based on Jet Propulsion Laboratory and Center for Space Research mascons solutions (2002–2022). *P* vs TWS, relationship between precipitation and long-term variability in TWS with a 2-month lag. Additional data are provided in Supplementary Information and Supplementary Table 3. Analysis for Iran is conducted at a country scale.

flux-approach estimate, similar to previous studies[29,30,32] (Fig. 4a). Hence, global models that ignore capture could greatly overestimate groundwater depletion, unless constrained by other data.

Overestimation of groundwater depletion was also found in comparisons of early global model outputs and regional models and in situ monitoring in many US aquifers[33]. Global models indicate that excessive groundwater pumping has greatly reduced discharge to streams, with 15–21% of watersheds reaching critical environmental flow thresholds (for the wettest to driest climate projections, respectively)[34]. Regional models and field studies also show large reductions in stream baseflows[35,36], corroborating the significant contribution of capture to groundwater abstraction and the need to include capture in depletion estimates.

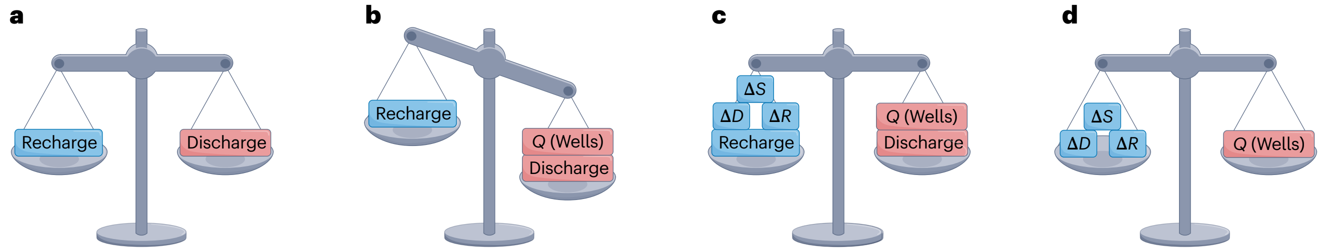


**Fig. 3 | Aquifer dynamics in response to development. a**, Predevelopment. **b**, Development with pumping. **c**, Pumping balanced by increased recharge, decreased discharge and/or a reduction (depletion) in groundwater storage. **d**, Removal of predevelopment recharge and discharge rates because unaffected by development. $Q$ is extraction from wells; $\Delta S$ is change in groundwater storage; $\Delta D$ is change in discharge from the aquifer; and $\Delta R$ is change in recharge to the aquifer. Figure adapted with permission from ref. [36], Groundwater Resource Development. Copyright © 2020 by The Authors.

### Long-term evolution of water resources

Although the GRACE period of record is now almost two decades (2002–2022), it is relatively short considering that water development began in many regions in the early 1900s. Analysis of groundwater observation wells in northwest India and central Pakistan shows century-long net increases in groundwater storage of at least 350 km$^3$ (1900–2010; Fig. 4b)[37]. Groundwater storage peaked in the 1970s due to recharge from canal irrigation from 1900 to 1960 but later declined by ~25–100 km$^3$ from 2000 to 2010, equivalent to ~20% of the net groundwater accumulation from 1900–2000 (450 km$^3$). This net increase in recharge since 1900 represents the largest anthropogenic unmanaged recharge globally.

Net increases in groundwater storage were modelled in aquifers in the northwest United States, including the Columbia Plateau, peaking at ~20 km$^3$ in the 1950s to 1970s and declining to 11 km$^3$ in 2009 (Fig. 4b)[38], and the East Snake River Plain aquifer, peaking at ~20 km$^3$ from the 1940s through early 1960s and declining to 9 km$^3$ in 2019[38]. The storage increases are attributed to surface water irrigation in the early to mid-1900s that enhanced spring discharge in the Snake River Plain, whereas groundwater irrigation increased in the late 1900s to 2000s. These data support modelled net increases in groundwater storage from surface water irrigation in northwest India, East Asia and northwest United States using the WaterGAP global model[2].

In contrast to rising trends in these regions, declining groundwater storage over the past century was found in the US central and southern High Plains and Central Valley based on groundwater level monitoring and regional modelling[33,39,40]. These storage declines are attributed to intensive groundwater irrigation in the High Plains, up to 10 times the aquifer recharge rates, and to switching from predominantly surface water irrigation to groundwater irrigation during drought in the Central Valley[38].

### Water scarcity

Water scarcity assessments have evolved to incorporate increasingly complex processes and refined scales (Supplementary Information Section 5). Populations affected by water scarcity increase from ~0.5 billion to 4 billion with increasing spatial resolution from country scale to ~50-km grid scale[41] and temporal resolution from annual to monthly[42]. Discrepancies arise as country-scale estimates use relative water demand (the ratio of withdrawal to long-term discharge) and do not account for the uneven spatial distribution of water supply and demand, both economic and population-driven[43].

Early studies of water scarcity focused on water supply from surface water only, including renewable groundwater discharge[41] whereas later studies incorporated groundwater abstraction[3]. Many people in water-insecure areas rely on groundwater storage to meet daily needs, highlighting the importance of groundwater in water availability metrics[28,44].

### Linkages between water quantity and quality

Degraded water quality further restricts the amount of water available for humans and ecosystems in many regions (Supplementary Information Section 4). When degraded surface water quality was included in water scarcity studies, the global population affected by water scarcity increased from 30% to 40% in one study[45] and from 33% to 65% in another[1], highlighting the substantial decrease in usable water supplies attributed to unsafe water quality. Degraded groundwater quality also exacerbates issues of water scarcity, with prominent examples including groundwater contamination by naturally occurring (geogenic) arsenic mobilization, exposing an estimated 94–220 million people to high arsenic concentrations (>10 μg l$^{-1}$, WHO guideline) in groundwater, with 94% residing in Asia[46].

Complex, interconnected processes controlling both water quantity and quality are increasingly being elucidated. For example, salinization of inland water bodies (such as lakes) in closed basins or endorheic basins reflects salt buildup from evaporative concentration over thousands of years[47,48]. Water abstraction exacerbates salinization in some of these basins, such as the Aral Sea[47]. In addition, with ~40% of global population living within 100 km of the coastline, seawater intrusion is a critical issue affecting ~30% of coastal metropolitan populations (cities with populations ≥1 million people)[49] and linked primarily to overexploitation of groundwater and also to sea level rise[50].

## Drivers of water-resource variability

Understanding drivers of spatiotemporal variability in water resources is essential to developing solutions to water-resource issues. Climate and human intervention are major drivers of spatiotemporal variability in water resources, with feedbacks between the two further affecting water resources. Human intervention affects water resources directly through water use and indirectly through land-use change. These controls are described separately in the following section. However, it is important to assess linkages and feedbacks between various controlling factors. For example, precipitation and human intervention can combine to amplify the effects on water storage, as shown by increased groundwater pumping in the US Central Valley during drought[38].

### Climate controls

Observed and projected increases in temperature from human-induced climate change cause increases in global atmospheric moisture holding capacity[51,52], intensifying the water cycle through rising evaporation and precipitation rates[53,54]. In addition, warming over land increases atmospheric evaporative demand, and subsequently, the severity of agricultural and ecological droughts[55].

Most hotspots of water scarcity and climate variability are found in semiarid regions globally (Fig. 2). Increases in drought frequency, duration and intensity have greatly affected water availability, such as the Millennium Drought in Australia[56], decadal drought in the Colorado River Basin[57], and droughts in the Middle East[58] and in eastern and southern Africa[59]. Many of these droughts and some drying in other regions, such as in Eurasia, have been linked to climate teleconnections related to El Niño Southern Oscillation (ENSO), North Atlantic Oscillation, Pacific Decadal Oscillation and others[60–62]. Droughts often end in floods caused by intense precipitation related to climate teleconnections[56] or atmospheric rivers (bands of concentrated atmospheric moisture). In the US West Coast, for example, an estimated 33–74% of droughts end in landfalling atmospheric rivers[63]. Extreme events related to ENSO have been shown to have a positive impact on groundwater resources through increased episodic recharge in semiarid regions[64,65]. Regions where irrigation is poorly developed and with high correlations between precipitation and GRACE-derived TWS variability reflect the importance of climate in controlling water storage, such as the Upper Kalahari aquifer.

In addition to varying precipitation extremes, projected trends in precipitation to 2100 also vary globally (based on 25 Coupled Model Intercomparison Project (CMIP) Phase 6 models with climate change under Shared Socioeconomic Pathway 8.5). Precipitation is projected to increase in much of sub-Saharan Africa, the northwest United States, northeast Brazil and parts of China, and to decrease in the Amazon and Mediterranean regions (Supplementary Fig. 2). In addition to isolated extremes and increasing variability in precipitation, predicted compound extreme events (such as concurrent heatwaves and droughts or storm surge and extreme rainfall)[55] will also affect water resources.

Additionally, climate change has accelerated glacier mass loss in the twenty-first century[66], which is concerning as mountain glaciers are critical for sustaining freshwater availability. Indeed, some of these losses are from regions that function as major 'water towers' (for example, the Brahmaputra in High Mountain Asia). Regional glacier mass loss is highly related to large-scale, decadal changes in temperature and precipitation[66]. Rapid glacier mass loss in the southeastern Tibetan Plateau, for example, is attributed mostly to rising temperatures[67–69].

Continued glacier loss under global warming could weaken the buffering capacity of glaciers to climate change, leading to temporal shifts in upstream hydrograph patterns, variations in runoff and water supply[70], and even droughts in downstream areas. This phenomenon is particularly true for densely populated water towers, such as the Indus and Ganges–Brahmaputra, with high glacier concentration, intensive irrigation and dense population[71]. In addition, warmer temperatures lead to earlier timing of glacial and snowmelt runoff, which decreases summer streamflow[72]. As a result, groundwater discharge to streams becomes increasingly important to maintain summer streamflow.

## Human intervention

Human intervention affects water resources directly through water use such as for irrigation, which accounts for 70% of global water withdrawal and 90% of water consumption[73]. Many hotspots of water scarcity globally correspond to intensively irrigated areas, mostly in semiarid regions (Fig. 2). In regions where water storage depletion is linked to human water use, primarily through groundwater-fed irrigation, correlations between climate forcing (precipitation) and TWS variability are generally low, such as in the US central and southern High Plains aquifer, Arabian aquifer and North China Plain aquifer (Table 1).

Various agricultural regions are in different phases of irrigation development, with reliance on surface water, groundwater or both. Limited surface water resources in some regions restrict irrigation to groundwater, such as the southern High Plains (Fig. 5a,b), resulting in steady groundwater depletion because demand greatly exceeds recharge rates[74]. However, many regions use surface water for irrigation during early stages of development, which increases groundwater recharge, as seen in the northwest United States, the Ganges–central Pakistan before the 1970s[37] (Fig. 5d) and the Po Valley, Italy (Supplementary Fig. 5d). Over time, increases in groundwater irrigation in these regions have reversed the initial rising storage trends. Irrigation impacts highlight the interconnections between groundwater and surface water and emphasize the importance of conjunctively managing both groundwater and surface water when available (for example, recent efforts to capture flood flows for managed aquifer recharge[75,76] in the US Central Valley[74]).

Human intervention indirectly affects water resources through land-use change. Examples of land-use change include cropland expansion, deforestation, wetland loss and urbanization[77] with many land-use changes inherently linked, such as cropland expansion with

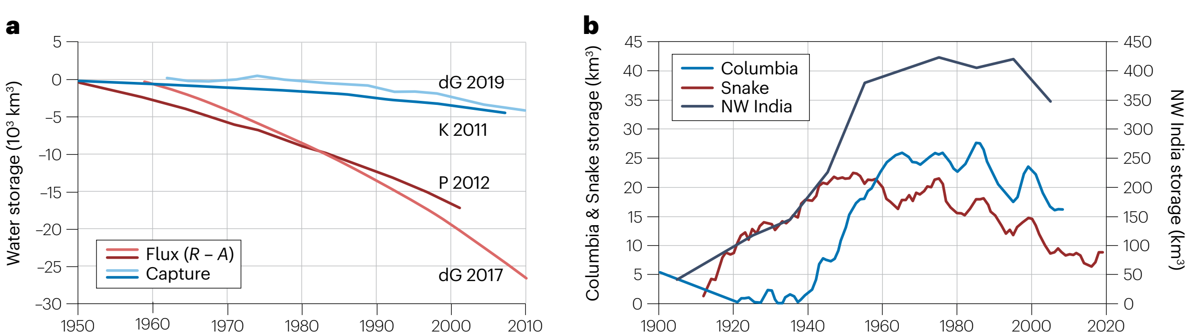


**Fig. 4 | Modelled global and regional groundwater storage over time.** **a**, Groundwater storage volumes based on global models that simulate fluxes (recharge *R* minus abstraction *A*) based on the work of Pokhrel et al. (P 2012)[31] and de Graaf 2017 (dG 2017)[32] versus global models that include capture of surface water and evapotranspiration (de Graaf et al. 2019 [dG 2019][34]; Konikow, 2011 [K 2011][30]). There is much lower groundwater storage depletion when capture is modelled. **b**, Long-term groundwater storage changes in northwest India and central Pakistan (only northwest India shown in key because of space restrictions) and in the northwest United States (Columbia Plateau and Snake River Plain). There has been a net increase in groundwater storage over the past century, followed by declines since the 1990s. Data are provided in Supplementary Table 4.

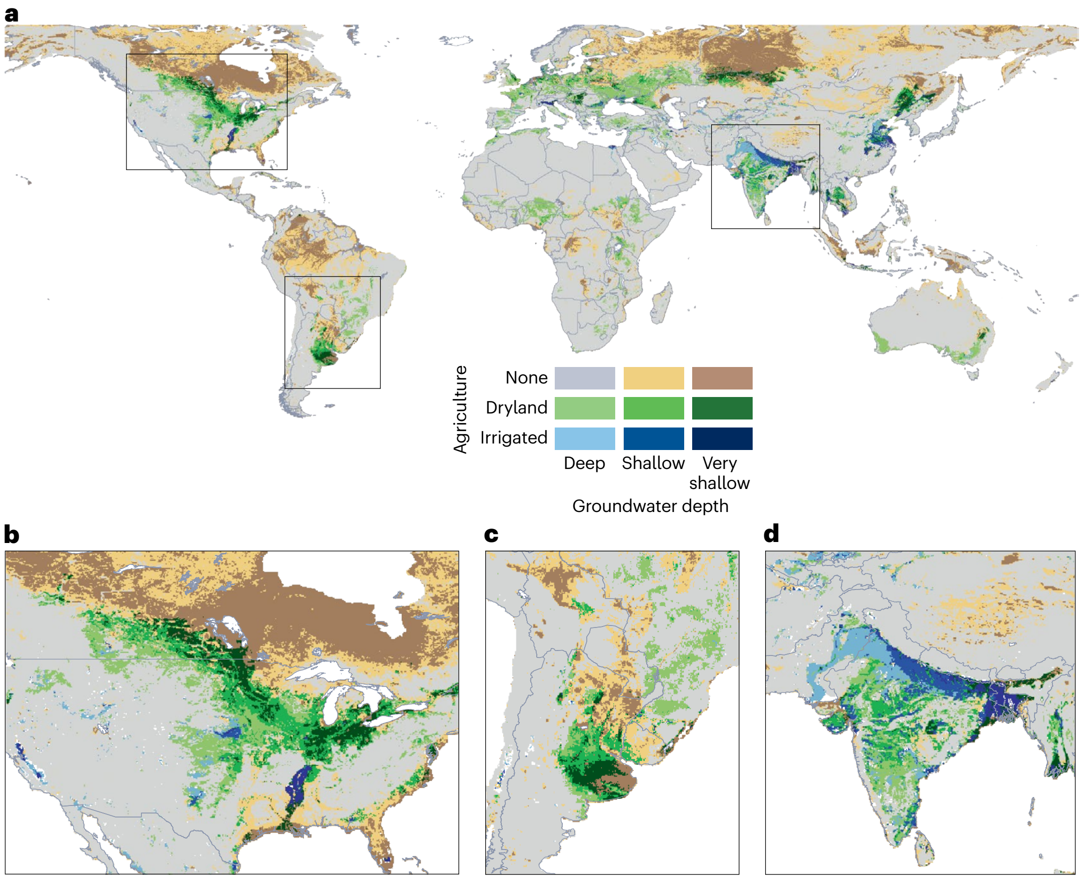


**Fig. 5 | Relationship between agricultural development and groundwater depth. a**, Global map. **b**, The United States and Canada. **c**, The Pampas and Chaco regions of South America. **d**, India and Pakistan. Cells are classified as agricultural if cropland areas exceed 30% of the cell area. Irrigated agriculture is estimated from cell areas with ≥30% irrigated area. Groundwater depths of ≤3 m, 3–10 m and >10 m correspond to very shallow, shallow and deep groundwater, respectively. Cropland data from ref. [175], water table depth from ref. [176]. Supplementary Fig. 3 includes additional panels for the Po valley in Western Europe, West Africa, and the border region of Kazakhstan and Russia.

deforestation. A recent analysis shows that doubling of cropland and tripling of ranchland over the past century in urban source watersheds resulted in a 40% increase in sediment yield and 120% increase in nitrogen yield[77]. Forested areas (406 billion hectares [Bha], 2018) represent ~31% of land surface but have been declining at a rate of 10 mha yr$^{-1}$ since 2015, primarily as a result of cropland expansion[78]. Wetlands (1.2 Bha) are also declining rapidly owing to land-use change and climate drivers, with an estimated 35% loss of natural wetlands from 1970–2015[79]. Wetland loss reduces water storage, flood attenuation and water quality maintenance.

Replacement of native vegetation with cropland increased groundwater supplies in many parts of Australia, South America, the southwest United States and West Africa, related to increased recharge associated with shallower rooting depths and shorter growing seasons in cropland relative to native vegetation[80–82] (Fig. 5). However, this land-use change led to trade-offs with dryland salinization in parts of Australia due to increasing evapotranspiration of shallower groundwater[83].

Irrigated agriculture causes both direct and indirect impacts on water resources, but the direct impacts of irrigation typically outweigh indirect effects (such as change in crop cover) on local water supplies. Additionally, the impact of agriculture development over time on water resources depends on the topographic and hydrologic setting. In flat topographic settings with shallow water tables, increasing recharge from dryland crop expansion may be linked to waterlogging, flooding and salinization (dark green belts, Fig. 5a). However, very shallow groundwater (≤3 m) can also supply water directly to crops, buffering the impact of droughts, as in parts of South America[84]. The semiarid Pampas and Chaco regions of South America have experienced rising water tables with doubling of dryland grain production over the past three decades, displacing perennial pastures, native grasslands and dry forests[81,85]. Other non-irrigated farming belts located in extremely flat settings with dry climates could be experiencing similar trends, as suggested for the Canadian prairies[86].

## Enhancing water-resource resilience

Resilience can be achieved by developing a diverse portfolio of solutions, such as increasing supplies, reducing demands, nature-based solutions, and storing and transporting water, to address multipronged issues, including water scarcity and excess, and water quality degradation. Conjunctive management of surface water and groundwater is an increasingly critical component of water management, where both are available, to holistically address climate and human stressors on water resources. Additionally, resilience will require multiple approaches with redundancies as backup, and therefore the most resilient systems will probably not be the most efficient systems.

In this section, approaches to enhance water-resource resilience are discussed.

### Increasing water access and supplies

Many people still lack access to water because of economic and/or governance constraints. Developing freshwater resources in regions with historically undeveloped groundwater can help to improve freshwater access for the 20 billion people without safely managed drinking water, with the most pronounced regions being sub-Saharan Africa and Oceania[87]. Although many countries in Africa appear water-stressed in global water scarcity indices, when factoring in groundwater storage[28] and the relatively small demand of drinking water, universal coverage for drinking water could be achieved with little impact on regional water stress[88]. Rainwater harvesting is also considered a valuable approach for increasing crop production because most agriculture is rainfed[89]. Therefore, exploration and use of groundwater resources, particularly in rural areas of Africa, can help meet SDG-6 and support economic development[88]. Boreholes equipped with hand pumps when maintained offer the most sustainable supply through long periods of drought in low-income settings[90].

Water security in rapidly urbanizing areas would benefit from improved water sanitation[91] and diversification of sources. Increasing groundwater development in regions that traditionally relied on surface water (for example, Cape Town, South Africa; São Paulo, Brazil) would greatly improve resilience in these rapidly urbanizing areas[92]. However, intensive groundwater development has also negatively affected some regions, such as by causing land subsidence in Mexico City (Mexico), Jakarta (Indonesia), Tehran (Iran) and the US Central Valley[93]. Conjunctively managing both surface water and groundwater, where possible, can reduce subsidence and enhance resilience, as in the US Arizona Basin and Range aquifers[94].

Many regions experiencing scarcity and overexploitation of water resources have traditionally responded with supply-side solutions, including developing conventional and unconventional water supplies such as wastewater reuse and desalination. Alternative water supplies, including wastewater reuse and recycling and desalination, constitute a relatively small (Fig. 1) but growing fraction of water supply portfolios. These approaches provide a water source that is effectively independent of climate variability.

Wastewater reuse and recycling can enhance water resources in urban and peri-urban areas where sufficient wastewater volumes are available and collected, with the added benefit of reducing pollutant discharges to the environment. Wastewater production scales with urbanization and economic development, and is projected to double by 2050[95]. However, infrastructure gaps for collecting and treating wastewater limit the potential for wastewater reuse, particularly in low- and middle-income countries[96]. Globally, ~360 km$^3$ yr$^{-1}$ of wastewater is produced from domestic and industrial sources[95,97] (Fig. 1) with ~63% of global production collected (227 km$^3$) and 52% treated (187 km$^3$), indicating that the remaining 48% is discharged without treatment into the environment[97]. Of this production, 11% of wastewater was reused (41 km$^3$ globally), mostly in the Middle East, North Africa and Europe[97,98]. Wastewater is primarily reused for irrigation, accounting for ~85% of irrigation water in Israel and ~70% in Spain[99,100].

Desalination is also increasingly promoted, particularly in coastal regions[101] and areas with brackish groundwater[102]. From 2000 to 2010, ~6,000 desalination plants were producing ~35 km$^3$ yr$^{-1}$ of water globally (Fig. 1), with about half of the capacity in the Middle East and North Africa[103]. Desalinated water is used primarily for domestic and industrial purposes[104] but also for irrigation in some Mediterranean countries[105]. Desalination of seawater and inland saline surface water could potentially be expanded to ~160 km$^3$ yr$^{-1}$, reducing water scarcity from severe to moderate levels[45]. The global population supported by desalination could triple from 2015 to 2050[101], with the highest potential for desalination in the United States, China, India and some European countries. However, managing brine waste from desalination is critical, as estimated global brine waste production from desalination (~52 km$^3$ yr$^{-1}$; 2000–2010) was ~50% greater than the global production of desalinated water[103]. Seawater desalination is considered more energy-intensive than recycling wastewater when the energy in wastewater is recovered[106], resulting in higher prices and greenhouse gas emissions compared with wastewater reuse. Desalination remains cost-prohibitive in most regions, and managing the resulting brine waste is difficult, limiting the applicability of desalination for increasing water resources at large scales.

### Reducing water demand

Because irrigation accounts for most water use[73] and is a large contributor to water quality degradation[107], there is an emphasis on reducing irrigation water demand. One approach for reducing irrigation would be to relocate crop production from semiarid to more humid regions, as suggested by a study exploring moving crop production from the semiarid Central Valley to the more humid Mississippi Basin, USA[108]. Additional approaches include switching to less water-intensive crops and fallowing cropland, particularly in drought years[109,110]. Increasing irrigation efficiency by shifting from flood to sprinkler and drip systems can also reduce water quality impacts from agricultural runoff. Getting 'more crop per drop' in both irrigated and rainfed crop production[111,112] can be supported by reducing non-beneficial evaporation through mulching, deficit irrigation and proper nutrient management. Deficit irrigation is practised in many regions and constitutes ~70% of irrigation in the southern High Plains[113]; however, it can cause soil salinization[114].

Increasing irrigation efficiency to reduce water demand is not straightforward, though. Increasing irrigation efficiency at a farm or field level in many regions has led to expansion of irrigated areas and reduced seepage and return flows, leading to net increase in water consumption at the basin or aquifer scale[110,115,116]. This observation is similar to the concept of the Jevons paradox, where increasing water supplies result in rising demands through a rebound effect[117]. The Jevons paradox illustrates the need for water managers and regulators to be aware of unintended consequences and behaviour of water users in order to consider policy changes or other approaches to ensure benefits to water resources.

Moreover, full water accounting of inflows, return flows and outflows, along with the change in groundwater storage, is necessary to determine whether irrigation efficiency can conserve water. For example, a detailed evaluation of irrigation in the Murray–Darling Basin in southeast Australia indicated that the $5.8-billion investment (USD) in infrastructure subsidies failed to deliver the projected environmental benefits, in part owing to a lack of comprehensive accounting of return flows at the basin level[116]. The source of irrigation water is also critical as losses (inefficiencies) of surface water irrigation could (beneficially) recharge underlying aquifers, as highlighted by recharge from canal irrigation in northwest India and central Pakistan[37].

Other approaches to reduce water demand are also implemented: for example, rationing of irrigation water in different regions in China by controlling water abstractions by farmers using smartcard systems with automated teller machines[118,119]. Other conservation tactics include promoting urban agriculture such as vertical indoor farms and aquaponics[120–123], reducing food loss and waste, and dietary changes that reduce water use. An estimated third of global food produced for human consumption is lost or wasted[124], accounting for ~23–24% of total water, cropland and fertilizer use[125], so reducing this wastage would reduce water usage.

### Nature-based solutions

There is increasing interest in nature-based or 'soft path' solutions (for example, decentralized systems, ecosystem services) to solve water-resource issues[126]. Because agricultural expansion represents the most widespread land-use change, various best management practices have been proposed such as conservation tillage, reduced fertilization, cover crops, terraces and agroforestry to reduce sediment and nutrient yield to downstream watersheds and cities[127].

Many large urban areas would greatly benefit from protecting their contributing watersheds, with conservation costs partially offset by the reduced water treatment costs in urban areas[128]. An excellent example is New York City, which works with landowners in the contributing watersheds to maintain the forested areas (75% of land area) and apply best management practices to agricultural areas to avoid building an 8–10 billion US$ treatment plant[92]. Local farmers are compensated to protect headwater catchments[129]. The city of São Paulo could reduce sediment and nutrient loading by restoring some forested areas, 70% of which were lost in the past[6,92]. Nature-based solutions are also proposed for the city of Cape Town[130] where partial removal of invasive species such as eucalyptus could greatly reduce water demand. Removal of invasive species is estimated to cost approximately one-tenth of other strategies for projected water supply increases, such as groundwater development, wastewater reuse and desalination.

Large-scale nature-based solution projects have been undertaken in China. Ecological restoration projects have been conducted throughout the Loess Plateau, including grassland development and forest and shrubland restoration, reducing runoff by ~70% and sediment loss by almost 100%[131]. Several wetland restoration projects have been developed, particularly in the middle reaches of the Yangtze to manage flooding and in parts of the Tibetan Plateau to protect the headwaters of the Yangtze, Yellow and Mekong rivers[132]. Other examples include restoration of ~200 water bodies in the city of Chennai to collect stormwater and recharge groundwater[133], emphasizing the importance of legally protecting areas, especially in urban watershed source areas[134]. The Sponge City Program, developed in China in 2013, has been deployed in 30 cities to mitigate urban flooding, purify stormwater and provide water storage for future use, and to enhance public amenities[135].

During planning, co-benefits associated with nature-based solutions should be considered, such as biodiversity, carbon sequestration and greenhouse gas impacts. Trade-offs also need to be considered, such as in forest restoration in parts of Africa, where flood risk and pollution are reduced but downstream water quantities are as well[136].

### Storing water

Managed water storage, including surface reservoirs and subsurface storage in aquifers, can resolve temporal disconnects between supply and demand caused by climate extremes (floods and droughts). Declining natural storage in snowpack as part of climate change[137] underscores the need to develop additional storage capacity to offset climate impacts.

Globally, an estimated 58,000 large dams (≥15 m high) provide an aggregated storage capacity of ~7,000–8,300 $km^3$ (ref. [138])(Fig. 1). Single-purpose dams are built for irrigation (~50%), hydropower (21%) and water supply (12%)[139]. However, mismanaged dams disrupt ecological connectivity of rivers and downstream water quantity and quality[26]. Although dam construction has already peaked in some (especially high-income) countries because suitable storage sites have been maximally developed, advances in the level of forecast skill foster efforts to optimize storage at existing sites using forecast-informed reservoir operations (FIRO), as demonstrated in Lake Mendocino, California[140,141]. FIRO involves transferring excess surface water prior to flooding from reservoirs to adjacent depleted aquifers to enhance water storage. The Ganges Water Machine provides another example of conjunctive management of surface water and groundwater to enhance water storage[142,143]. Expanded groundwater-fed irrigation during non-monsoon periods provides increased space to store flood waters from the 3-month monsoon period, enhancing surface–subsurface water exchange.

Dam construction is markedly increasing in low- and middle-income countries where there is still large potential for reservoirs[144]. About 3,700 hydroelectric dams are under construction or planned, mostly in South America, South and East Asia, and Africa[145,146]. There are drawbacks to using reservoirs to increase water shortages. For example, filling the Grand Ethiopian Renaissance Dam (GERD, 74 $km^3$ capacity) could greatly reduce reservoir levels in the High Aswan Dam reservoir downstream, and management of both reservoirs will be required to address multiyear droughts[147]. Based on the Jevons paradox, as with respect to irrigation efficiency, increasing water supplies can increase demands and make systems more vulnerable to shortages[148].

There is growing interest in storing water in depleted aquifers using managed aquifer recharge (MAR), the process of artificially infiltrating or injecting water into the subsurface for storage and later recovery. Moreover, with increasing climate extremes, there is rising interest in capturing flood flows and storm flows to recharge depleted aquifers[149,150]. The annual volume of water stored globally through MAR has increased to ~10 $km^3$ in 2015 (ref. [151])(Fig. 1). Although MAR storage volumes are low relative to surface reservoirs, MAR can be an extremely important local-scale strategy to help alleviate regional water stress. For example, in Orange County, California, MAR is an essential component of local water supply portfolio and provides enough water for 850,000 people, in addition to co-benefits such as preventing seawater intrusion and improving water quality[152]. Depletion of aquifer storage in the United States has been estimated at 1,000 $km^3$ between 1900 and 2008, exceeding the capacity of new surface reservoirs[30,153] (673 $km^3$) constructed during that period. This legacy of aquifer depletion represents a large potential subsurface reservoir capacity to support MAR, even considering permanent aquifer storage loss from compaction (for example, ~20% in California)[40]. MAR projects can further expand local storage options through conjunctive management of traditional surface reservoirs with co-located MAR facilities. Although MAR can have multiple benefits, including mitigating land subsidence and ecosystem restoration, there can also be adverse impacts on the environment, including waterlogging, soil salinization and water quality degradation[154].

### Transporting water

Water can be transported physically through aqueducts and pipelines or virtually through trade of food and other commodities. Physical water transfers over large distances are energy-intensive and often

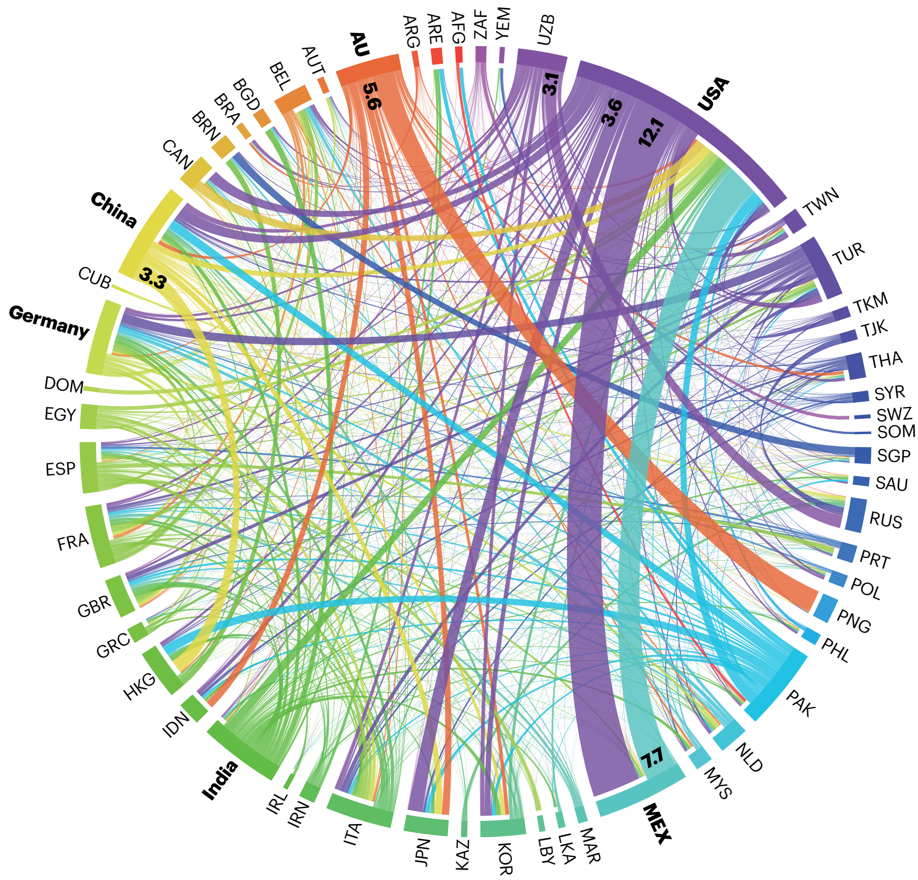


**Fig. 6 | Global virtual water flows linked to food trade.** Blue virtual water flows between countries (in km$^3$ yr$^{-1}$) based on 1996–2005 period of analysis[8]. The colours of the links correspond to the exporting country. The corresponding plot for virtual groundwater flows is in Supplementary Fig. 3a. Data and country acronyms are provided in Supplementary Table 8a.

not considered a viable option because of high cost. Global water transfer megaprojects are those with construction costs ≥US$1 billion, distances ≥190 km and volumes ≥0.23 km$^3$ yr$^{-1}$ (ref. [155]). Currently, there are 34 existing megaprojects and 76 planned or proposed with primary uses for irrigation followed by hydroelectricity and multipurpose projects. Most of these megaprojects transfer surface water and have affiliated impoundment infrastructures. Existing megaprojects transfer a total of 204 km$^3$ yr$^{-1}$ (Fig. 1) across distances up to 2,820 km (Great Manmade River, Libya) and 1,128 km (California State Water Project, USA). Projected transfer volumes for planned or proposed projects (1,910 km$^3$ yr$^{-1}$) are an order of magnitude greater than existing projects, with total pipeline distances exceeding twice the Earth's circumference (80,000 km). For example, the South to North Water Diversion Project (SNWDP) megaproject in China was designed to supply water for municipal and industrial uses, transporting up to 45 km$^3$ yr$^{-1}$ via diversions from the Yangtze River in the humid south to cities in the semiarid north. Part of the transferred water through the SNWDP is used to recharge depleted aquifers through MAR[156]. Potential benefits of megaprojects include ecological restoration (as seen, for example, in the Florida Everglades), flood control in donor basins, wetland restoration in receiving basins (for instance, the Mississippi Transfer Project; Peacock River Transfer Project), and subsidence reduction (as in the Central Arizona Project). Negative environmental trade-offs include runoff reduction in river discharge in donor basins and increased pollution, salinization, and drying of lakes (for example, Aral Sea)[157].

In addition to physical water transfers, countries have for decades relieved water stress by importing virtual water from more water-abundant countries[158–160] (Fig. 6). Virtual water is defined as the water consumed to generate products, such as food, but not reflected in the water content of the final product, and is transported virtually with the traded products[18]. If water-intensive products are traded from nations of high to low water productivity, virtual water imports result in water savings for importing countries as well as global water savings.

Globally, blue virtual water flows (sourced from surface water and groundwater) totalled ~301 km$^3$ yr$^{-1}$ from 1996 to 2005[161]. Crops and derived crop products account for the largest share of virtual blue water flows between countries (Fig. 6). However, groundwater depletion is widespread in major exporting regions. Substantial non-renewable groundwater flows embodied in international trade were found from South Asia to West Asia and from North America to East Asia[162,163]. Major virtual groundwater exporters include the United States (31% of global total), India (15%) and Pakistan (13%). Major crops responsible for unsustainable virtual groundwater flows are wheat, maize, rice, sugarcane, cotton and fodder[164].

Water transfers can also be facilitated through water markets, allowing temporary lease or permanent transfer of groundwater or surface water rights. Water markets can regulate supply and demand by reallocating water from low to high-value users. Large differences in the marginal values of water across users result in inefficiencies in water use and, conversely, opportunities to reallocate water among users. In 2019–2020, water markets in the southern Murray Darling Basin were worth ~US$750 million in traded permanent water rights[165]. An independent review of these water markets concluded that economic impacts were net positive, particularly in times of drought when some high-value uses of water would, otherwise, not be possible without water trades from lower value uses.

With few exceptions[166], water markets do not account for non-market values that can include small-scale water users and ecosystem services of streamflows or groundwater. Definitions of *de minimis* groundwater use typically focus on negligible extraction for domestic purposes. For example, California's Sustainable Groundwater Management Act defines *de minimis* groundwater use as extraction for domestic purposes of 2 acre feet per year or less. Therefore, checks are required to ensure that trading does not affect ecosystems and water supply for marginalized or low-income groups[167]. Some markets do implement trading rules (such as zonal trading restrictions) to limit impacts on ecosystems or de minimis users. For example, the Murray–Darling Basin developed one of the world's largest water markets by unbundling land from water rights to promote trade[168].

## Summary and future perspectives

Recognizing that surface water and groundwater behave as a single resource is essential for managing water resources. Surface reservoirs store large volumes of water (7,000–8,300 $km^3$), but these reservoirs are more vulnerable to long-term droughts than aquifers. Groundwater plays a critical role in resilience, as it buffers spatiotemporal variability in surface water[169]. However, aquifers are far more difficult to manage on a large-scale basis, rendering them more susceptible to overexploitation.

Where available, using both surface water and groundwater through conjunctive management, in addition to developing alternative supplies, builds redundancy and resiliency into water-resource systems. For example, excess surface water during wet periods can be temporally reallocated by storing water in depleted aquifers for use during dry periods. Water policies often hamper conjunctive management, owing to separate legal frameworks for surface water and groundwater that persist in most countries. There is growing recognition of the need to close the governance gap of groundwater in integrated water resources management, with both UN-Water World Water Day and the World Water Development Report dedicated to groundwater in 2022.

Increased satellite data, advances in global and regional modelling, and expanded ground-based monitoring networks have revolutionized understanding of water quantity. In contrast, the overall lack of global water quality monitoring continues to constitute a major shortcoming in assessing water resources[170]. Most countries do not routinely collect water quality data, placing >3 billion people at risk for health concerns because they rely on water of unknown chemical or biological character[171]. To improve data coverage, the UNEP Water Programme is building a global water quality database, GEMStat, including surface water (73% of stations) and groundwater (27% of stations)[172].

An improved understanding of climate drivers and overexploitation of water resources highlights the severity of water-resource challenges, and also enables and informs more effective future water management that can account for complex feedbacks in water systems (such as trade-offs in water quantity and quality). More than ever, water managers have a large body of operational and scientific knowledge that can be used to develop diverse water management portfolios. These strategies extend beyond traditional engineered approaches that have been relied on historically and emphasize nature-based solutions, economic incentives and the importance of water governance[173,174] to achieve desirable outcomes. Not only does a portfolio of solutions build resilience to climate and human stressors on water supplies, but it also provides an opportunity to maximize co-benefits for ecosystems and human water users.

In summary, there is no single, one-size-fits-all solution for achieving resilient water resources; rather, resilience will be achieved through a portfolio of water management options. The relative importance of specific approaches (such as increasing water supplies, reducing water demand, storing water and/or transferring water) will vary in a given region depending on local stressors and water uses. Regardless of the specific approach, it is critical for groundwater and surface water to be managed as a single resource.

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## References

1. Vorosmarty, C. J. et al. Global threats to human water security and river biodiversity. *Nature* **467**, 555–561 (2010).
2. Doell, P., Mueller Schmied, H., Schuh, C., Portmann, F. T. & Eicker, A. Global-scale assessment of groundwater depletion and related groundwater abstractions: combining hydrological modeling with information from well observations and GRACE satellites. *Water Resour. Res.* **50**, 5698–5720 (2014).
3. Wada, Y. et al. Global depletion of groundwater resources. *Geophys. Res. Lett.* **37**, L20402 (2010).
4. Douville, H. et al. in *Climate Change 2021: The Physical Science Basis* (eds Masson-Delmotte, V. et al.) 1055–1210 (IPCC, Cambridge Univ. Press, 2021).
5. Olivier, D. W. & Xu, Y. X. Making effective use of groundwater to avoid another water supply crisis in Cape Town, South Africa. *Hydrogeol. J.* **27**, 823–826 (2019).
6. Ozment, S. et al. Natural infrastructure in Sao Paulo's water system. *World Resources Institute Report 2013–2014: Interim Findings* (2018).
7. Pascale, S., Kapnick, S. B., Delworth, T. L. & Cooke, W. F. Increasing risk of another Cape Town 'Day Zero' drought in the 21st century. *Proc. Natl Acad. Sci. USA* **117**, 29495 (2020).
8. Alley, W. M., Reilly, T. E. & Franke, O. L. Sustainability of ground-water resources. *US Geological Survey Circular* 1186 (1999).
9. Breslin, S. COP26 has 4 goals. Water is central to all of them. *SIWI News* https://siwi.org/latest/cop26-has-4-goals-water-is-central-to-all-of-them/ (2021).
10. *Global Risks 2021* 16th edition (World Economic Forum, 2021); https://www.weforum.org/reports/the-global-risks-report-2021/
11. *The Water Challenge: The Roundtable on Water Financing* (OECD, 2022); https://www.oecd.org/water/roundtable-on-financing-water.htm
12. *The United Nations World Water Development Report 2018: Nature-Based Solutions for Water* (United Nations World Water Assessment Program/UNESCO, 2018).
13. Browder, G., Ozment, S., Rehberger-Bescos, I., Gartner, T. & Lange, G. M. *Integrating Green and Gray: Creating Next Generation Infrastructure* (World Bank and World Resources Institute, 2019); https://openknowledge.worldbank.org/handle/10986/31430
14. *Making Every Drop Count: Agenda for Water Action* (High Level Panel on Water, United Nations and World Bank, 2018).
15. Lederer, E. M. Next UN assembly president warns world in dangerous crisis. *Washington Post* https://www.washingtonpost.com/world/next-un-assembly-president-warns-world-in-dangerous-crisis/2022/06/07/55075dce-e6b6-11ec-a422-11bbb91db30b_story.html (7 June 2022).
16. Tapley, B. D. et al. Contributions of GRACE to understanding climate change. *Nat. Clim. Change* **9**, 358–369 (2019).
17. Wada, Y. & Bierkens, M. F. P. Sustainability of global water use: past reconstruction and future projections. *Environ. Res. Lett.* https://doi.org/10.1088/1748-9326/9/10/104003 (2014).
18. Mekonnen, M. M. & Hoekstra, A. Y. Blue water footprint linked to national consumption and international trade is unsustainable. *Nat. Food* **1**, 792–800 (2020).
19. Rodell, M. et al. Emerging trends in global freshwater availability. *Nature* **557**, 651–659 (2018).
20. Save, H., Bettadpur, S. & Tapley, B. D. High-resolution CSR GRACE RL05 mascons. *J. Geophys. Res. Solid Earth* **121**, 7547–7569 (2016).
21. Tapley, B. D., Bettadpur, S., Watkins, M. & Reigber, C. The Gravity Recovery And Climate Experiment: mission overview and early results. *Geophys. Res. Lett.* https://doi.org/10.1029/2004gl019920 (2004).

22. Richey, A. S. et al. Quantifying renewable groundwater stress with GRACE. *Water Resour. Res.* **51**, 5217–5238 (2015).
23. Shamsudduha, M. & Taylor, R. G. Groundwater storage dynamics in the world's large aquifer systems from GRACE: uncertainty and role of extreme precipitation. *Earth Syst. Dyn.* **11**, 755–774 (2020).
24. Vishwakarma, B. D., Bates, P., Sneeuw, N., Westaway, R. M. & Bamber, J. L. Re-assessing global water storage trends from GRACE time series. *Environ. Res. Lett.* **16**, 034005 (2021).
25. Pekel, J. F., Cottam, A., Gorelick, N. & Belward, A. S. High-resolution mapping of global surface water and its long-term changes. *Nature* **540**, 418–436 (2016).
26. Lehner, B. et al. High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. *Front. Ecol. Environ.* **9**, 494–502 (2011).
27. Scanlon, B. R. et al. Global models underestimate large decadal declining and rising water storage trends relative to GRACE satellite data. *Proc. Natl Acad. Sci. USA* **115**, E1080–E1089 (2018).
28. Winter, T. C., Harvey, J. W., Franke, O. L. & Alley, W. M. *Ground Water and Surface Water: A Single Resource*. Circular 1139 (United States Geological Survey, 1998).
29. Konikow, L. F. Overestimated water storage. *Nat. Geosci.* **6**, 3 (2013).
30. Konikow, L. F. Contribution of global groundwater depletion since 1900 to sea-level rise. *Geophys. Res. Lett.* https://doi.org/10.1029/2011gl048604 (2011).
31. Pokhrel, Y. N. et al. Model estimates of sea-level change due to anthropogenic impacts on terrestrial water storage. *Nat. Geosci.* **5**, 389–392 (2012).
32. de Graaf, I. E. M. et al. A global-scale two-layer transient groundwater model: development and application to groundwater depletion. *Adv. Water Resour.* **102**, 53–67 (2017).
33. Rateb, A. et al. Comparison of groundwater storage changes from GRACE satellites with monitoring and modeling of major U.S. aquifers. *Water Resour. Res.* https://doi.org/10.1029/2020WR027556 (2020).
34. de Graaf, I. E. M., Gleeson, T., van Beek, L. P. H., Sutanudjaja, E. H. & Bierkens, M. F. P. Environmental flow limits to global groundwater pumping. *Nature* **574**, 90–94 (2019).
35. Sophocleous, M. From safe yield to sustainable development of water resources — the Kansas experience. *J. Hydrol.* **235**, 27–43 (2000).
36. Konikow, L. F. & Bredehoeft, J. D. *Groundwater Resource Development: Effects and Sustainability* (The Groundwater Project, 2020).
37. MacAllister, D. J., Krishan, G., Basharat, M., Cuba, D. & MacDonald, A. M. A century of groundwater accumulation in Pakistan and northwest India. *Nat. Geosci.* https://doi.org/10.1038/s41561-022-00926-1 (2022).
38. Scanlon, B. R. et al. Effects of climate and irrigation on GRACE-based estimates of water storage changes in major US aquifers. *Environ. Res. Lett.* https://doi.org/10.1088/1748-9326/ac16ff (2021).
39. McGuire, V. L. *Water-Level and Recoverable Water In Storage Changes, High Plains Aquifer, Predevelopment to 2015 and 2013–15*. US Geological Survey Scientific Investigations Report 2017–5040 (2017); https://doi.org/10.3133/sir20175040
40. Faunt, C. C. Groundwater availability of the Central Valley Aquifer, California. *US Geol. Surv. Prof. Pap*. 1766 (2009).
41. Vorosmarty, C. J., Green, P., Salisbury, J. & Lammers, R. B. Global water resources: vulnerability from climate change and population growth. *Science* **289**, 284–288 (2000).
42. Mekonnen, M. M. & Hoekstra, A. Y. Four billion people facing severe water scarcity. *Sci. Adv.* https://doi.org/10.1126/sciadv.1500323 (2016).
43. Vorosmarty, C. J. & Sahagian, D. Anthropogenic disturbance of the terrestrial water cycle. *Bioscience* **50**, 753–765 (2000).
44. Gronwall, J. & Danert, K. Regarding groundwater and drinking water access through a human rights lens: self-supply as a norm. *Water* https://doi.org/10.3390/w12020419 (2020).
45. van Vliet, M. T. H. et al. Global water scarcity including surface water quality and expansions of clean water technologies. *Environ. Res. Lett.* https://doi.org/10.1088/1748-9326/abbfc3 (2021).
46. Podgorski, J. & Berg, M. Global threat of arsenic in groundwater. *Science* **368**, 845–850 (2020).
47. Yapiyev, V., Sagintayev, Z., Inglezakis, V. J., Samarkhanov, K. & Verhoef, A. Essentials of endorheic basins and lakes: a review in the context of current and future water resource management and mitigation activities in Central Asia. *Water* https://doi.org/10.3390/w9100798 (2017).
48. Pauloo, R. A., Fogg, G. E., Guo, Z. L. & Harter, T. Anthropogenic basin closure and groundwater salinization (ABCSAL). *J. Hydrol.* https://doi.org/10.1016/j.jhydrol.2020.125787 (2021).
49. Cao, T. Z., Han, D. M. & Song, X. F. Past, present, and future of global seawater intrusion research: a bibliometric analysis. *J. Hydrol.* https://doi.org/10.1016/j.jhydrol.2021.126844 (2021).
50. Werner, A. D. et al. Seawater intrusion processes, investigation and management: recent advances and future challenges. *Adv. Water Resour.* **51**, 3–26 (2013).
51. Held, I. M. & Soden, B. J. Robust responses of the hydrological cycle to global warming. *J. Clim.* **19**, 5686–5699 (2006).
52. Fan, X., Duan, Q. Y., Shen, C. P., Wu, Y. & Xing, C. Global surface air temperatures in CMIP6: historical performance and future changes. *Environ. Res. Lett.* **15**, 104056 (2020).
53. Tabari, H. Climate change impact on flood and extreme precipitation increases with water availability. *Sci. Rep.* https://doi.org/10.1038/s41598-020-70816-2 (2020).
54. Williams, A. P. et al. Large contribution from anthropogenic warming to an emerging North American megadrought. *Science* **368**, 314 (2020).
55. Arias, P. A. et al. in *Climate Change 2021: The Physical Science Basis* (eds Masson-Delmotte, V. et al.) 33–144 (IPCC, Cambridge Univ. Press, 2021).
56. van Dijk, A. et al. The Millennium Drought in southeast Australia (2001–2009): natural and human causes and implications for water resources, ecosystems, economy, and society. *Water Resour. Res.* **49**, 1040–1057 (2013).
57. Scanlon, B. R. et al. Hydrologic implications of GRACE satellite data in the Colorado River Basin. *Water Resour. Res.* **51**, 9891–9903 (2015).
58. Rateb, A., Scanlon, B. R. & Kuo, C. Y. Multi-decadal assessment of water budget and hydrological extremes in the Tigris-Euphrates Basin using satellites, modeling, and in-situ data. *Sci. Total Environ.* https://doi.org/10.1016/j.scitotenv.2020.144337 (2021).
59. Anyamba, A., Glennie, E. & Small, J. Teleconnections and interannual transitions as observed in African vegetation: 2015–2017. *Remote Sens.* https://doi.org/10.3390/rs10071038 (2018).
60. Scanlon, B. R. et al. Linkages between GRACE water storage, hydrologic extremes, and climate teleconnections in major African aquifers. *Environ. Res. Lett.* https://doi.org/10.1088/1748-9326/ac3bfc (2022).
61. Ul Hassan, W. & Nayak, M. A. Global teleconnections in droughts caused by oceanic and atmospheric circulation patterns. *Environ. Res. Lett.* https://doi.org/10.1088/1748-9326/abc9e2 (2021).
62. Shen, Z. X. et al. Drying in the low-latitude Atlantic Ocean contributed to terrestrial water storage depletion across Eurasia. *Nat. Commun.* **13**, 1849 (2022).
63. Dettinger, M. D. Atmospheric rivers as drought busters on the US West Coast. *J. Hydrometeorol.* **14**, 1721–1732 (2013).
64. Taylor, R. G. et al. Ground water and climate change. *Nat. Clim. Change* **3**, 322–329 (2013).
65. Cuthbert, M. O. et al. Observed controls on resilience of groundwater to climate variability in sub-Saharan Africa. *Nature* **572**, 230 (2019).
66. Hugonnet, R. et al. Accelerated global glacier mass loss in the early twenty-first century. *Nature* **592**, 726 (2021).
67. Zhao, F., Long, D., Li, X., Huang, Q. & Han, P. Rapid glacier mass loss in the Southeastern Tibetan Plateau since the year 2000 from satellite observations. *Remote. Sens. Environ.* **270**, 112853 (2022).
68. Li, X. Y. et al. Climate change threatens terrestrial water storage over the Tibetan Plateau. *Nat. Clim. Change* https://doi.org/10.1038/s41558-022-01443-0 (2022).
69. Yao, T. D. et al. The imbalance of the Asian water tower. *Nat. Rev. Earth Environ.* https://doi.org/10.1038/s43017-022-00299-4 (2022).
70. Immerzeel, W. W., van Beek, L. P. H. & Bierkens, M. F. P. Climate change will affect the Asian water towers. *Science* **328**, 1382–1385 (2010).
71. Immerzeel, W. W. et al. Importance and vulnerability of the world's water towers. *Nature* **577**, 364 (2020).
72. Dery, S. J. et al. Detection of runoff timing changes in pluvial, nival, and glacial rivers of western Canada. *Water Resour. Res.* https://doi.org/10.1029/2008wr006975 (2009).
73. Siebert, S. et al. Groundwater use for irrigation – a global inventory. *Hydrol. Earth Syst. Sci.* **7**, 3977–4021 (2010).
74. Scanlon, B. R. et al. Groundwater depletion and sustainability of irrigation in the US High Plains and Central Valley. *Proc. Natl Acad. Sci. USA* **109**, 9320–9325 (2012).
75. Dahlke, H. E. et al. in *Advanced Tools for Integrated Water Resources Management* Vol. 3 (eds Friesen, J. & Rodriguez-Sinobas, L.) 215–275 (Elsevier, 2018).
76. Reddy, V. R., Pavelic, P. & Hanjra, M. A. Underground taming of floods for irrigation (UTFI) in the river basins of South Asia: institutionalising approaches and policies for sustainable water management and livelihood enhancement. *Water Policy* **20**, 369–387 (2018).
77. McDonald, R. I., Weber, K. F., Padowski, J., Boucher, T. & Shemie, D. Estimating watershed degradation over the last century and its impact on water-treatment costs for the world's large cities. *Proc. Natl Acad. Sci. USA* **113**, 9117–9122 (2016).
78. *The State of the World's Forests 2020. Forests, Biodiversity, and People* (FAO/UNEP, 2020).
79. Convention on Wetlands. *Global Wetland Outlook: Special Edition 2021* (Secretariat of the Convention on Wetlands, 2021).
80. Scanlon, B. R., Jolly, I., Sophocleous, M. & Zhang, L. Global impacts of conversions from natural to agricultural ecosystems on water resources: quantity versus quality. *Water Resour. Res.* https://doi.org/10.1029/2006WR005486 (2007).
81. Nosetto, M. D., Paez, R. A., Ballesteros, S. I. & Jobbagy, E. G. Higher water-table levels and flooding risk under grain vs. livestock production systems in the subhumid plains of the Pampas. *Agric. Ecosyst. Environ.* **206**, 60–70 (2015).
82. Favreau, G. et al. Land clearing, climate variability, and water resources increase in semiarid southwest Niger: a review. *Water Resour. Res.* https://doi.org/10.1029/2007wr006785 (2009).
83. Walker, C. D., Zhang, l, Ellis, T. W., Hatton, T. J. & Petheram, C. Estimating impacts of changed land use on recharge: review of modelling and other approaches appropriate for management of dryland salinity. *Hydrogeol. J.* **10**, 68–90 (2002).
84. Nosetto, M. D., Jobbagy, E. G., Jackson, R. B. & Sznaider, G. A. Reciprocal influence of crops and shallow ground water in sandy landscapes of the Inland Pampas. *Field Crops Res.* **113**, 138–148 (2009).
85. Gimenez, R., Mercau, J., Nosetto, M., Paez, R. & Jobbagy, E. The ecohydrological imprint of deforestation in the semiarid Chaco: insights from the last forest remnants of a highly cultivated landscape. *Hydrol. Process.* **30**, 2603–2616 (2016).
86. Eilers, R. G., Eilers, W. D. & Fitzgerald, M. M. A salinity risk index for soils of the Canadian prairies. *Hydrogeol. J.* **5**, 68–79 (1997).

87. *Progress on Household Drinking Water, Sanitation and Hygiene 2000–2020: Five Years into the SDGs* (WHO/UNICEF, 2021).
88. Cobbing, J. & Hiller, B. Waking a sleeping giant: realizing the potential of groundwater in sub-Saharan Africa. *World Dev.* **122**, 597–613 (2019).
89. Rockström, J. & Falkenmark, M. Agriculture: increase water harvesting in Africa. *Nature* **519**, 283–285 (2015).
90. MacAllister, D. J., MacDonald, A. M., Kebede, S., Godfrey, S. & Calow, R. Comparative performance of rural water supplies during drought. *Nat. Commun.* **11**, 1099 (2020).
91. Aboah, M. & Miyittah, M. K. Estimating global water, sanitation, and hygiene levels and related risks on human health, using global indicators data from 1990 to 2020. *J. Water Health* **20**, 1091–1101 (2022).
92. Abell, R. et al. *Beyond the Source: The Environmental, Economic and Community Benefits of Source Water Protection* (The Nature Conservancy, 2017).
93. Herrera-Garcia, G. et al. Mapping the global threat of land subsidence. *Science* **371**, 34–36 (2021).
94. Scanlon, B. R., Reedy, R. C., Faunt, C. C., Pool, D. & Uhlman, K. Enhancing drought resilience with conjunctive use and managed aquifer recharge in California and Arizona. *Environ. Res. Lett.* **11**, 035013 (2016).
95. Qadir, M. et al. Global and regional potential of wastewater as a water, nutrient and energy source. *Nat. Resour. Forum* **44**, 40–51 (2020).
96. *Water Reuse within a Circular Economy Context*. Global Water Security Issues Series 2 (UNESCO, 2020).
97. Jones, E. R., van Vliet, M. T. H., Qadir, M. & Bierkens, M. F. P. Country-level and gridded estimates of wastewater production, collection, treatment and reuse. *Earth Syst. Sci. Data* **13**, 237–254 (2021).
98. Jeuland, M. Challenges to wastewater reuse in the Middle East and North Africa. *Middle East. Dev. J.* **7**, 1–25 (2015).
99. Zhang, Y. & Shen, Y. Wastewater irrigation: past, present, and future. *WIREs Water* **6**, e1234 (2019).
100. Fito, J. & Van Hulle, S. W. H. Wastewater reclamation and reuse potentials in agriculture: towards environmental sustainability. *Environ. Dev. Sust.* **23**, 2949–2972 (2021).
101. Gao, L., Yoshikawa, S., Iseri, Y., Fujimori, S. & Kanae, S. An economic assessment of the global potential for seawater desalination to 2050. *Water* https://doi.org/10.3390/w9100763 (2017).
102. Ahdab, Y. D., Thiel, G. P., Bohlke, J. K., Stanton, J. & Lienhard, J. H. Minimum energy requirements for desalination of brackish groundwater in the United States with comparison to international datasets. *Water Res.* **141**, 387–404 (2018).
103. Jones, E., Qadir, M., van Vliet, M. T. H., Smakhtin, V. & Kang, S. M. The state of desalination and brine production: a global outlook. *Sci. Total Environ.* **657**, 1343–1356 (2019).
104. Lin, S. S. et al. Seawater desalination technology and engineering in China: a review. *Desalination* https://doi.org/10.1016/j.desal.2020.114728 (2021).
105. Martinez-Alvarez, V., Martin-Gorriz, B. & Soto-Garcia, M. Seawater desalination for crop irrigation — a review of current experiences and revealed key issues. *Desalination* **381**, 58–70 (2016).
106. Smith, K., Liu, S. M., Hu, H. Y., Dong, X. & Wen, X. H. Water and energy recovery: the future of wastewater in China. *Sci. Total Environ.* **637**, 1466–1470 (2018).
107. Pulido-Bosch, A. et al. Impacts of agricultural irrigation on groundwater salinity. *Environ/Earth Sci.* https://doi.org/10.1007/s12665-018-7386-6 (2018).
108. Kurnik, J. *The Next California: Phase 1: Investigating Potential in the Mid-Mississippi Delta River Region* (The Markets Institute at WWF, 2020); https://www.worldwildlife.org/publications/the-next-california-phase-1-investigating-potential-in-the-mid-mississippi-delta-river-region
109. Senay, G. B., Schauer, M., Friedrichs, M., Velpuri, N. M. & Singh, R. K. Satellite-based water use dynamics using historical Landsat data (1984–2014) in the southwestern United States. *Remote Sens. Environ.* **202**, 98–112 (2017).
110. Gebremichael, M., Krishnamurthy, P. K., Ghebremichael, L. T. & Alam, S. What drives crop land use change during multi-year droughts in California's Central Valley? Prices or concern for water? *Remote Sens.* https://doi.org/10.3390/rs13040650 (2021).
111. Brauman, K. A., Siebert, S. & Foley, J. A. Improvements in crop water productivity increase water sustainability and food security — a global analysis. *Environ. Res. Lett.* **8**, 024030 (2013).
112. Mekonnen, M. M., Hoekstra, A. Y., Neale, C. M. U., Ray, C. & Yang, H. S. Water productivity benchmarks: the case of maize and soybean in Nebraska. *Agric. Water Manag.* https://doi.org/10.1016/j.agwat.2020.106122 (2020).
113. Colaizzi, P. D., Gowda, P. H., Marek, T. H. & Porter, D. O. Irrigation in the Texas High Plains: a brief history and potential reductions in demand. *Irrig. Drain.* **58**, 257–274 (2008).
114. Scanlon, B. R., Gates, J. B., Reedy, R. C., Jackson, A. & Bordovsky, J. Effects of irrigated agroecosystems: (2). Quality of soil water and groundwater in the southern High Plains, Texas. *Water Resour. Res.* **46**, W09538 (2010).
115. Ward, F. A. & Pulido-Velazquez, M. Water conservation in irrigation can increase water use. *Proc. Natl Acad. Sci. USA* **105**, 18215–18220 (2008).
116. Grafton, R. Q. et al. The paradox of irrigation efficiency. *Science* **361**, 748–750 (2018).
117. Alcott, B. in *The Jevons Paradox and the Myth of Resource Efficiency Improvements* (eds Polimeni, J. M., Mayumi, K., & Giampetro, M.) 7–78 (Earthscan, 2008).
118. Aarnoudse, E. & Bluemling, B. *Controlling Groundwater Through Smart Card Machines: The Case of Water Quotas and Pricing Mechanisms in Gansu Province, China*. Groundwater Solutions Initiative for Policy and Practice (GRIPP) Case Profile Series 02 (International Water Management Institute, 2017); https://doi.org/10.5337/2016.224
119. Kinzelbach, W., Wang, H., Li, Y., Wang, L. & Li, N. *Groundwater Overexploitation in the North China Plain: A Path to Sustainability* (Springer, 2021).
120. McDougall, R., Kristiansen, P. & Rader, R. Small-scale urban agriculture results in high yields but requires judicious management of inputs to achieve sustainability. *Proc. Natl Acad. Sci. USA* **116**, 129–134 (2019).
121. Langemeyer, J., Madrid-Lopez, C., Mendoza Beltran, A. & Villalba Mendez, G. Urban agriculture — a necessary pathway towards urban resilience and global sustainability? *Landsc. Urban Plan.* **210**, 104055 (2021).
122. Palmer, L. Urban agriculture growth in US cities. *Nat. Sust.* **1**, 5–7 (2018).
123. Grafius, D. R. et al. Estimating food production in an urban landscape. *Sci. Rep.* **10**, 5141 (2020).
124. *The State of Food Insecurity in the World 2015* (FAO/IFAD/WFP, 2015).
125. Kummu, M. et al. Lost food, wasted resources: global food supply chain losses and their impacts on freshwater, cropland, and fertiliser use. *Sci. Total Environ.* **438**, 477–489 (2012).
126. Gleick, P. H. Global freshwater resources: soft-path solutions for the 21st century. *Science* **302**, 1524–1528 (2003).
127. Miralles-Wilhelm, F. *Nature-Based Solutions in Agriculture — Sustainable Management and Conservation of Land, Water, and Biodiversity* (FAO/The Nature Conservancy, 2021).
128. McDonald, R. I. & Shemie, D. *Urban Water Blueprint: Mapping Conservation Solutions to the Global Water Challenge* (The Nature Conservancy, 2014); http://water.nature.org/waterblueprint
129. Kane, M. & Erickson, J. D. Urban metabolism and payment for ecosystem services: history and policy analysis of the New York city water supply. *Adv. Econ. Environ. Resour.* **7**, 307–328 (2007).
130. *Greater Cape Town Water Fund: Business Case: Assessing the Return on Investment for Ecological Infrastructure Restoration* (The Nature Conservancy, 2019).
131. Hu, J., Lu, Y. H., Fu, B. J., Comber, A. J. & Harris, P. Quantifying the effect of ecological restoration on runoff and sediment yields: a meta-analysis for the Loess Plateau of China. *Prog. Phys. Geogr. Earth Environ.* **41**, 753–774 (2017).
132. Liu, W. W. et al. Improving wetland ecosystem health in China. *Ecol. Indic.* https://doi.org/10.1016/j.ecolind.2020.106184 (2020).
133. *Cities100: Chennai Is Restoring Waterbodies to Protect Against Flooding and Drought*. C40 Knowledge Hub: Nordic Sustainability, South and West Asia, Chennai, Case Studies and Best Practice Examples https://www.c40knowledgehub.org/s/article/Cities100-Chennai-is-restoring-waterbodies-to-protect-against-flooding-and-drought?language=en_US (2019).
134. Chung, M. G., Frank, K. A., Pokhrel, Y., Dietz, T. & Liu, J. G. Natural infrastructure in sustaining global urban freshwater ecosystem services. *Nat. Sust.* **4**, 1068 (2021).
135. Qi, Y. F. et al. Addressing challenges of urban water management in Chinese sponge cities via nature-based solutions. *Water* https://doi.org/10.3390/w12102788 (2020).
136. Acreman, M. et al. Evidence for the effectiveness of nature-based solutions to water issues in Africa. *Environ. Res. Lett.* https://doi.org/10.1088/1748-9326/ac0210 (2021).
137. Livneh, B. & Badger, A. M. Drought less predictable under declining future snowpack. *Nat. Clim. Change* **10**, 452–458 (2020).
138. Mulligan, M., van Soesbergen, A. & Sáenz, L. GOODD, a global dataset of more than 38,000 georeferenced dams. *Sci. Data* **7**, 31 (2020).
139. International Commission on Large Dams https://www.icold-cigb.org/ (2022).
140. Yang, G., Guo, S., Liu, P. & Block, P. Integration and evaluation of forecast-informed multiobjective reservoir operations. *J. Water Resour. Plan. Manag.* **146**, 04020038 (2020).
141. Delaney, C. J. et al. Forecast informed reservoir operations using ensemble streamflow predictions for a multipurpose reservoir in northern California. *Water Resour. Res.* https://doi.org/10.1029/2019wr026604 (2020).
142. Amarasinghe, U. A., Muthuwatta, L., Surinaidu, L., Anand, S. & Jain, S. K. Reviving the Ganges water machine: potential. *Hydrol. Earth Syst. Sci.* **20**, 1085–1101 (2016).
143. Shamsudduha, M. et al. The Bengal water machine: quantified freshwater capture in Bangladesh. *Science* **377**, 1315–1319 (2022).
144. Chao, B. F., Wu, Y. H. & Li, Y. S. Impact of artificial reservoir water impoundment on global sea level. *Science* **320**, 212–214 (2008).
145. Zarfl, C., Lumsdon, A. E., Berlekamp, J., Tydecks, L. & Tockner, K. A global boom in hydropower dam construction. *Aquat. Sci.* **77**, 161–170 (2015).
146. Zarfl, C. et al. Future large hydropower dams impact global freshwater megafauna. *Sci. Rep.* https://doi.org/10.1038/s41598-019-54980-8 (2019).
147. Wheeler, K. G., Jeuland, M., Hall, J. W., Zagona, E. & Whittington, D. Understanding and managing new risks on the Nile with the Grand Ethiopian Renaissance Dam. *Nat. Commun.* https://doi.org/10.1038/s41467-020-19089-x (2020).
148. Di Baldassarre, G. et al. Water shortages worsened by reservoir effects. *Nat. Sust.* **1**, 617–622 (2018).
149. Dahlke, H. E., Brown, A. G., Orloff, S., Putnam, D. & O'Geen, T. Managed winter flooding of alfalfa recharges groundwater with minimal crop damage. *Calif. Agric.* **72**, 65–75 (2018).
150. Yang, Q. & Scanlon, B. R. How much water can be captured from flood flows to store in depleted aquifers for mitigating floods and droughts? A case study from Texas, US. *Environ. Res. Lett.* **14**, 054011 (2019).
151. Dillon, P. et al. Sixty years of global progress in managed aquifer recharge. *Hydrogeol. J.* https://doi.org/10.1007/s10040-018-1841-z. (2018).
152. Groundwater Replenishment System Technical Brochure, https://www.ocwd.com/media/10443/gwrs-technical-brochure-2021.pdf (2021).

153. Konikow, L. F. *Groundwater Depletion in the United States (1900–2008)*. US Geological Survey Scientific Investigation Report 2013–5079, http://pubs.usgs.gov/sir/2013/5079 (2013).
154. Hartog, N. & Stuyfzand, P. J. Water quality donsiderations on the rise as the use of managed aquifer recharge systems widens. *Water* **9**, 808 (2017).
155. Shumilova, O., Tockner, K., Thieme, M., Koska, A. & Zarfl, C. Global water transfer megaprojects: a potential solution for the water–food–energy nexus? *Front. Environ. Sci.* https://doi.org/10.3389/fenvs.2018.00150 (2018).
156. Long, D. et al. South-to-north water diversion stabilizing Beijing's groundwater levels. *Nat. Commun.* https://doi.org/10.1038/s41467-020-17428-6 (2020).
157. Zhuang, W. Eco-environmental impact of inter-basin water transfer projects: a review. *Environ. Sci. Pollut. Res.* **23**, 12867–12879 (2016).
158. Hoekstra, A. Y. *Virtual Water Trade: Proceedings of the International Expert Meeting on Virtual Water Trade* (UNESCO-IHE, 2003).
159. Oki, T. & Kanae, S. Virtual water trade and world water resources. *Water Sci. Technol.* **49**, 203–209 (2004).
160. Dolan, F. et al. Evaluating the economic impact of water scarcity in a changing world. *Nat. Commun.* https://doi.org/10.1038/s41467-021-22194-0 (2021).
161. Hoekstra, A. Y. & Mekonnen, M. M. The water footprint of humanity. *Proc. Natl Acad. Sci. USA* **109**, 3232–3237 (2012).
162. Dalin, C., Wada, Y., Kastner, T. & Puma, M. J. Groundwater depletion embedded in international food trade. *Nature* **543**, 700–704 (2017).
163. Hanasaki, N., Inuzuka, T., Kanae, S. & Oki, T. An estimation of global virtual water flow and sources of water withdrawal for major crops and livestock products using a global hydrological model. *J. Hydrol.* **384**, 232–244 (2010).
164. Mekonnen, M. M. & Gerbens-Leenes, W. The water footprint of global food production. *Water* https://doi.org/10.3390/w12102696 (2020).
165. *Australian Water Markets Report: 2019-20 Review and 2020-21 Outlook* (Aither, 2020); https://aither.com.au/wp-content/uploads/2020/08/2020-Water-Markets-Report.pdf
166. Grafton, R. Q. & Wheeler, S. A. Economics of water recovery in the Murray–Darling Basin, Australia. *Annu. Rev. Resour. Econ.* **10**, 487–510 (2018).
167. Moench, M. Water and the potential for social instability: livelihoods, migration and the building of society. *Nat. Resour. Forum* **26**, 195–204 (2002).
168. *Water Markets in Australia: A Short History* (National Water Commission, 2011).
169. Kundzewicz, Z. W. & Döll, P. Will groundwater ease freshwater stress under climate change? *Hydrol. Sci. J.* **54**, 665–675 (2009).
170. *A Snapshot of the World's Water Quality: Towards a Global Assessment* (UNEP, 2016).
171. *Summary Progress Update 2021: SDG 6 — Water and Sanitation for All* (UN-Water, 2021).
172. GEMStat: Global Environmental Monitoring System, https://gemstat.org/ (UNEP, 2022).
173. Akhmouch, A. & Correia, F. N. The 12 OECD principles on water governance — when science meets policy. *Util. Policy* **43**, 14–20 (2016).
174. Lankford, B., Bakker, K., Zeitoun, M. & Conway, B. D. *Water Security: Principles, Perspectives, and Practices* (Routledge, 2013).
175. Potapov, P. et al. Global maps of cropland extent and change show accelerated cropland expansion in the twenty-first century. *Nat. Food* **3**, 19 (2022).
176. Fan, Y., Li, H. & Miguez-Macho, G. Global patterns of groundwater table depth. *Science* **339**, 940–943 (2013).

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B.R.S. conceptualized the review and coordinated input. S.F. reviewed many of the topics and developed some of the figures. A.R. analysed GRACE satellite data and M.S. reviewed this output. Q.G. provided input on water economics. E.J. reviewed impacts of land-use change. S.R.K. provided data on future precipitation changes. L.F.K. provided detailed information on surface water/groundwater interactions. M.M. provided data on water trade. C.J.V. provided input on green and grey solutions. All authors reviewed the paper and provided edits.

## Competing interests
The authors declare no competing interests.

## Additional information
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